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人类活动对坦桑尼亚热带沿海生态系统结构的干扰和服务价值的影响

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Human Activity Disturbances on the Structure and Impacts on the Service Values of the Coastal Ecosystems in Tanzania

By

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(Supervisor)

Date...2018年 11月11日...

Declaration

I, **Elly Josephat Ligate** declare to the Graduate College of Fujian Agriculture and Forestry University that this thesis entitled; **"Human Activity Disturbances on the Structure and Impacts on the Service Values of the Coastal Ecosystems in Tanzania"** accomplished by me under the directions of my supervisor and that it has not been presented and will not be presented to any other University for a similar or any other degree award. This is an independent study, all the reviewed works from other researchers scientifically cited, hence, this Doctoral Degree thesis submitted with no violation of academic ethics and standards.

Signature: Elly Ligate

Date: 2018.11.11

Dedication

This thesis dedicated to Alidika Kaseke Mahenge (my mother) who passed away in 2011. Her inspiring legacy on my academic foundation is exemplary. Unfortunately, there is no any platform for Alidika to eyewitness this Doctor of Science-Ecology achievement!

May God rest her soul in eternal peace!

Amen!

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Acronyms and Abbreviations

ADS	Agriculture Disturbed Sites
CEC	Cation Exchange Capacity
CFS	Closed Forest Sites
DCCA	Detrended Canonical Correspondence Analysis
DGS	Disturbed by Livestock Grazing Sites
EC	Electrical Conductivity
ESV	Ecosystem Services Values
FDG	Focus Group Discussions
FAFU	Fujian Agriculture and Forest University
FAO	Food and Agriculture Organization
FRA	Forest Resources Assessment
H-ESV	Human to Ecosystem Services Value
Kha/yr	1000 hectors per year
IUCN	International Union for Conservation of Nature
JRC	Joint Research Centre
LCLU	Land Cover and Lands Uses
LU	Land uses
MEA	Millennium Environmental Assessment
Mha	Million Hectors
MODIS	Moderate Resolution Imaging Spectroradiometer

NDVI	Normalized Difference Vegetation Indcx
NGCC	National Geomatics Center of China
SPSS	Statistical package for social sciences
SUA	Sokoine University of Agriculture
TCF	Tropical Coastal Forests
TM	Thematic Mapper
ETM	Enhanced Thematic Mapper
UFR	Uzigua Forest Reserve
UNDP	United Nations Development Programme
URT	United Republic of Tanzania
USA	United States of America
USDA	United States Department of Agriculture

摘要

农业和放牧活动是坦桑尼亚沿海生态系统的主要威胁，因此本研究旨在调查两类活动对坦桑尼亚沿海地区生态系统结构的干扰和对服务价值的影响，即通过调查人类活动干扰地点的土壤空间特征和植被状态，建立沿海生态系统服务价值的时间动态，生成坦桑尼亚热带生态系统可持续的管理信息。

具体而言，研究目标包括：1.明确坦桑尼亚沿海生态系统干扰地中土壤物理性质（电导率，土壤质地和容重）和化学性质（氮（N），碳（C），磷（P）），钙（Ca），镁（Mg），钾（K）和钠（Na）的空间变异.2.调查人类活动干扰对坦桑尼亚天然热带沿海森林中植被和再生潜力的影响；3.分析和评估坦桑尼亚人为干扰和完整的沿海森林生态系统的植被和土壤特性之间存在的相互关系；4.探讨社会经济活动如何促成土地覆盖和土地利用变化及其对坦桑尼亚沿海生态系统服务价值的影响.5.确定和评估坦桑尼亚受干扰的热带沿海生态的当前恢复干预措施并确定恢复阻碍因素。

研究区域位于坦桑尼亚和肯尼亚北部边境以及坦桑尼亚和莫桑比克南部的 850 公里范围内。该区域是坦桑尼亚沿海生态系统最多的区域，在 2000 年至 2016 年之间具显著的生态系统扰动。

为明确受农业和放牧业干扰的沿海地区的结构和服务之间的相互作用，本研究收集了生物物理和社会经济数据。先从覆盖 S37 00，S37 50 的区域计算出土地覆盖和土地利用的分类，并从受两大典型干扰的 Uzigua 森林保护区采集样本。在 ADS，DGS 和 CFS 中从 47 个采样样地

中收集土壤样品。电导率由电导率仪测定，土壤质地用移液管法测定，体积密度用土壤干重除以体积计算。通过凯氏酸消化程序测定总氮，同时通过 Walkley-Black 程序分析总碳。通过 Bray-II 方法测定有效磷。使用醋酸铵（1M NH 4 OAc）（pH 7.0）提取可交换的 Ca，K，Mg 和 Na。然后通过火焰光度计测定 K 含量，同时进行乙二胺四乙酸（EDTA）滴定以测量 Ca 和 Mg。

为分析土地覆盖和土地利用，使用 Arc View 1.3 和 ERDASimag 软件 8.3.1 版进行地理信息系统（GIS）分析，结合地面验证以收集在图像解释的初步阶段中描述的生物物理数据和修改的土地覆盖面积。从对照、农业和放牧地点收集活植被信息，用于确定人类受干扰地点的森林结构。根据事先准备的树种清单，结合每个土地用途计算：（i）每单位面积的活树数量（N/ha），（ii）活树基面积（m^2）/ha），和（iii）每单位面积的活树体积（m^3/ha）。通过基面积测量成年树干、幼树和幼苗所占的面积。采用（i）香农多样性指数，（ii）辛普森多样性指数，iii）物种均匀度和（iv）重要性价值指数（IVI）分析了生物多样性指数。所有计算通过 Microsoft Excel 10 和社会科学软件统计软件包（SPSS）进一步分析，为获得树参数与土壤变量之间的最佳线性组合进行了典型的多元相关分析，去趋势规范对应分析（DCCA）。

结果表明：

人为活动后，沿海生态系统中空间土壤性质以及森林结构参数和时间生态系统服务价值存在显著差异。1. CFS 和 ADS 之间的电导率（μS/cm）的平均变化为 26.197±8.42；CFS 和 DGS 为 5.55±7.45；ADS 和 DGS 为 20.65±3.97。沙质土壤颗粒介于 86.06%至 86.79%之间，粘土介于 11.40%至 14.98%之间，土地用途淤泥介于 1.81%至 2.57%之间。CFS 和 ADS 中的体积密度为 0.05±0.23，在 CFS 中，DGS 为 0.13±0.02，在 ADS 和 DGS 中为 0.08±0.02。氮的平均值（百分比）分别为 16.07±0.34,1.75±0.25,6.5±0.20;碳分别为 14.48±0.23,11.81±0.13,12.24±0.30；CFS，ADS 和 DGS 的磷分别为 14.12±6.57,17.74±3.96 和 13.31±2.86。ADS 的总碳含量略低于 DGS。CFS 中的碳氮比率高于受干扰的地点。 Ca^{2-}的可溶性碱基平均值分别为 3.75， 3.11 和 0.63；Mg^{2+}分别为 0.80， 5.87 和 6.67；K^{+}分别为 0.03，0.55 和 0.52；Na^{+}分别为 0.01,0.31 和 0.31;阳离子交换容量（cmol（+）/ kg）分别为 2.61， 13.74 和 16.36，三个区域的基础饱和度（%体积）分别为 10.29， 5.86 和 4.42。

2.同样，植被林参数和多样性指数存在显著差异。与 ADS 和 DGS 相比，CFS 中的地块具有更高的成年树木，基础面积和体积平均值。ADS 中的地块具有最高的 Shannon-Wiener 幼苗和幼树指数，其次是 CFS 和 DGS。与 DGS 相比，CFS 和 DGS 的幼苗和幼树的 Simpson 指数更高。 ADS 和 DGS 中的地块中 Simpson 指数大于封闭地点更高的成年树。幼苗和幼树的适宜性在 CFS 中最高，其次是 ADS 和 DGS。与 DGS 和

CFS 相比，ADS 中的地块具有更高的成年树木的均匀性。此外，ADS 和 DGS 中的地块对幼苗的重要性价值指数高于 CFS。

3. 多变量典型相关显示 CFS，ADS 和 DGS 之间存在显着差异。对于林分参数（TSP）和土壤物理参数（SPP），典型相关性为 F = 2.400，p <0.012。在 ADS 中，F-检验为 0.529，p = 0.938。在 DGS 中，所有规范轴的显着性为 F = 1.207，p =0.242。可溶性碱基和 TSP 的相关性为 F = 2.448，CFS 中 p =0.018，F = 0.687，ADS 中 p =0.790，DGS 中 F = 0.743，p =0.808。在 CGS 中，不溶性碱基和 TSP 的值为 F = 0.816，p =0.572，FFS 为 0.687，p =0.790，F =0.070，p =0.020。SPP 和 Shannon 指数是 F = 1.103，CFS 中 p <0.388，F = 0.520，ADS 中 p = 0.714，DGS 中 F = 0.932，p =0.444。 SPP 和 IVI 为 F = 0.042，CFS 中 p =0.996，F = 0.819，ADS 中 p =0.620，DGS 中 F = 0.633，p = 0.724。可溶性碱和均衡性为 F = 0.119，CFS 中 p =0.968，F = 0.001，ADS 中 p =0.001，F =0.011，DGS 中 p =0.001。在 CFS，ADS 和 DGS 之间，CNP 与公平性之间几乎没有确定的相关性。

4.2000 年至 2016 年间，沿海生态系统面积（单位 ha）发生了显着变化。森林面积下降为-36 441（-10％），放牧面积为-6 347（-2％），湿地为-112（-2％）。 LCLU 地区的扩张在灌木林中为 11 751（37％），农田为 30 506（43％），水体为 279（14％），人工表面为 365（14％）。森林覆盖率下降 12％，放牧面积下降 2％，湿地面积下降 2％，耕地面积下降 30％，灌木面积下降 27％，水体下降 12％。ESV 和总人口比率在 2000 年，2010 年和 2016 年逐年下降，分别为

80.4 美元，63.8 美元和 46.0 万美元。在农作物种植，牲畜饲养和生物能源利用方面，LCLU 变化和 ESV 值、人口和家庭之间存在完美的正相关关系。

5. 本研究还发现，植树造林，重新造林和保留自然生长的树木是坦桑尼亚沿海地区使用的主要恢复干预措施。在某些情况下，可实行对森林入侵者的驱逐以允许受干扰的森林自然再生。同时，社会经济和气候因素对人工种植和自然再生树产生了显著的负面影响（$p<0.050$）。但目前对恢复干预的生态价值的理解仍有限，一方面缺乏改良的种植材料；另一方面，当地社区对森林资源的过度依赖也严重阻碍了恢复干预措施。

调查结果表明人类活动的干扰影响了坦桑尼亚整个沿海地区的生态系统结构和服务价值。其中，人口和社会经济活动是主要驱动因素，特别是需求增加导致沿海生态系统的价值大幅流失。如果没有外力介入，生态系统有进一步受损的危险。因此，本研究建议 1. 规范人口和社会经济活动，以避免沿海 LCLU 变化产生进一步负面影响。 2. 通过确保充分解决沿海地区的社会经济和生态相互作用，改进目前的恢复干预措施。 3. 为了使社会经济活动成为一个对沿海生态系统无害的解决方案，需要调查和确定沿海地区允许的最低农作物种植和牲畜放牧一体化平衡点。

关键词：耕作干扰；畜牧干扰；沿海生态系统；坦桑尼亚

Abstract

This study aimed to investigate and document human activities disturbances on the ecosystems structure and impacts on ecosystem services values (ESV) along the coastal areas of Tanzania. The study provides initial understanding about spatial characteristics of soils and vegetation on human disturbed sites in comparison with the intact sites, and establishment of temporal coastal ESV to provide information for sustainable management of tropical ecosystems in Tanzania. Specifically, this study aimed 1. To establish spatial variation on soil physical properties (electrical conductivity, soil texture and bulk density), and soil chemical properties (nitrogen (N), carbon (C), phosphorus (P), calcium (Ca), magnesium (Mg), potassium (K) and sodium (Na) across disturbed and intact sites of the coastal ecosystems; 2. To investigate the impacts of human activities disturbances on the vegetation and regeneration potential of naturally occurring tropical coastal forests; 3. To establish the correlations between vegetation and soil properties of the human disturbed and intact coastal forest ecosystems; 4. To explore how socioeconomic activities contribute to temporal change of coastal ecosystems and their impacts of changes on the coastal ESV; 5. To identify and gauge the current restoration interventions and factors influencing restoration of the disturbed coastal ecosystems in Tanzania.

This study was conducted by collecting data from the Coastal Zone sites in Tanzania. The zone stretches within 850km from the boarder of Tanzania and Kenya in the north, and Tanzania and Mozambique in the south. The zone represents the coastal ecosystems, which are affected by human disturbances especially between 2000 and 2016. The classification of land

cover and land uses was firstly computed from the areas that cover S37 00, S37 50 and S37 50, S37 10 geographical coordinates. Samples were collected from Uzigua Forest Reserve (UFR). Certainly, this forest is within 100 km from the coastline of Indian Ocean, thus the reserve is considered among the true representative of the Tropical Coastal Ecosystems in Tanzania.

The study collected bio-physical and socioeconomic data to understand the interplays between human activities, structure and services of the coastal sites. Soil samples were collected from ADS, DGS and CFS in forty-seven (25m × 25m) sampling plots. Each site provided 47 samplings plots, which resulted into 141 plots in total. These plots were laid randomly because agriculture and livestock grazing are not uniformly distributed within the ecosystems. In each of the plots, 10 soil samples were collected from 1-30cm depth because agriculture and livestock grazing activities affect the surface and near surface layers of the soils. Representative soil samples were put into tightened double plastic bags, labeled and stored at 4°C to reduce further microbial degradation during transportation and storage in the laboratory. Fresh air-dried and oven-dried weights were determined before subjecting soil samples into further laboratory analysis.

Soil physical and chemical properties were analyzed as follows: Electric conductivity was determined by electrical conductivity meter, soil texture by pipette method and bulk density calculated by dividing dry weight of soils by volume. Determination of total nitrogen followed the Kjeldahl acid-digestion procedures while total carbon were analysed by the Walkley-Black procedures. Potassium Dichromate ($K_2Cr_2O_2$) and concentrated Sulphuric

Acid (H_2SO_4) were used to produce the reaction and products. Available phosphorus was determined by the Bray-II method. The Ammonium Acetate was used to extract exchangeable Ca, K, Mg and Na. Then K content was determined by using flame photometer while ethylenediaminetetraacetic acid (EDTA) titration was done to measure Ca, Mg and Na. A combined glass–calomel electrode used to determine the pH of aqueous suspensions.
Tree inventories were prepared on each sites to determine forest structure in the human disturbed areas compared to intact sites. From live trees (i) number of live trees per unit area (N/ha), (ii) basal area of live trees (m^2 /ha), and (iii) volume of live tree per unit area (m^3 /ha). Basal area and volume were compared between and across CFS, ADS and DGS. The mean values and t-test were used to compare and judge the variations, which exist across the sites. The results were considered significant at $p < .050$.

To analyze the differences in vegetation the following parameters were computed (i) Shannon diversity index, (ii) Simpson diversity index, iii) species evenness and (iv) the importance value index (IVI). Each of the computed diversity indices were subjected into the Microsoft Excel 10 and Statistical Package for Social Sciences Software (SPSS) for production of means, standard deviation and t-tests.The Detrended Canonical Correspondence Analysis (DCCA)) was used to obtain multiple linear regressions and optimal linear combination between tree parameters and soil variables for comparisons across CFS, ADS and DGS. To analyze the impacts of human activities on changes of ecosystems, land cover and land use (LCLU) and estimation of ESV, area changes were detected based on differences between imagery identification of the changed areas. The LCLU information were used to compare land changes in relationships with the socioeconomic activities, and the dynamics of ESV in the coastal zone. The LCLU data for 2000 and 2010 were from the Globe Land 30 mapping products at 30-meter spatial resolution developed by National Geomatics Center of China, while 2016 images were produced from Landsat 8. Classification of images was done from Landsat TM/ETM+ for 2000, 2010 and 2016 years complemented with MODIS and Normalized Difference Vegetation Index time series, and Chinese HJ imagery. The LCLU categories and ecosystem service coefficients used to compute ESV on each LCLU categories.

There were significant variations in spatial soil properties as well as forest structural parameters and temporal ecosystems service values. 1. The mean variation in electrical conductivity (μS/cm) between CFS and ADS was 26.197 ± 8.42; CFS and DGS was 5.55 ± 7.45; ADS and DGS was 20.65 ± 3.97. The soil particles ranged between 86.06% to 86.79% for sandy, 11.40% to 14.98% for clay and 1.81% to 2.57% for silt across land uses. The bulk density in CFS and ADS was 0.05 ± 0.23, in CFS and DGS was 0.13 ± 0.02 and, in ADS and DGS was 0.08 ± 0.02. The mean values (percentage) for nitrogen = 16.07 ± 0.34, 1.75 ± 0.25, 6.5 ± 0.20; carbon =14.48 ± 0.23, 11.81 ± 0.13, 12.24 ± 0.30; phosphorus =14.12 ± 6.57, 17.74 ± 3.96, and 13.31± 2.86 for CFS, ADS and DGS respectively. There were slightly lower amount of total carbon on ADS than DGS. Carbon-nitrogen ratio was higher in CFS than in the disturbed sites. The mean values for soluble bases were 3.75, 3.11 and 0.63 for Ca^{2+}; 0.80, 5.87 and 6.67 for Mg^{2+}; 0.03, 0.55, and 0.52 for K^{+}; 0.01, 0.31 and 0.31 for Na^{+}; 2.61, 13.74 and 16.36 (cmol(+)/kg) for cation exchange capacity and 10.29, 5.86 and 4.42 (% volume) for base saturation in three areas: CFS, ADS and DGS.

2. There were significant variation across the vegetation stand parameters and as well as for diversity indices. Plots in the CFS had higher mean values of adult trees, basal area and volume than the ADS and DGS. Plots in ADS had the highest Shannon-Wiener index of seedlings and saplings, followed by CFS and DGS. The CFS and DGS had higher Simpson's index for seedlings and saplings than DGS. Plots in ADS and DGS had higher adult tree Simpson than closed sites. The equitability of seedlings and saplings was highest in CFS, followed by ADS and DGS. Plots in ADS had higher equitability of adult trees than DGS and CFS. Moreover, plots in ADS and DGS had higher importance value index for seedlings than CFS.

3. The multivariate canonical correlation showed significant variation across CFS, ADS and DGS. The canonical correlation was $F = 2.400$, $p < .012$ for tree stand parameters (TSP) and soil physical parameters (SPP). In ADS, the F- test was 0.529, $p = .938$. In DGS, the significance of all canonical axes was $F = 1.207$, $p = .242$. Correlation of soluble bases and TSP was $F = 2.448$, $p = .018$ in CFS, $F = 0.687$, $p = .790$ in ADS and $F = 0.743$, $p = .808$ in DGS. The values of non-soluble bases and TSP were $F = 0.816$, $p = .572$ in CFS, $F = 0.687$, $p = .790$ and $F = .070$, $p = .020$ in DGS. The SPP and Shannon index was that $F = 1.103$, $p < .388$ in CFS, $F = 0.520$, $p = .714$ in ADS and $F = 0.932$, $p = .444$ in DGS. The SPP and IVI was $F = 0.042$, $p = .996$ in CFS, $F = 0.819$, $p =.620$ in ADS and $F= 0.633$, $p = .724$ in DGS. Soluble bases and equitability was $F = 0.119$, $p = .968$ in CFS, $F = 0.001$, $p = .001$ in ADS and $F = 0.011$, $p = .001$ in DGS. There were almost no

established correlation between CNP and equitability across CFS, ADS and DGS.

4. Between 2000 and 2016, ecosystems along coast changed significantly. The decline (ha) was in forest by -36 441 (-10%), grazing land by -6 347 (-2%) and wetland by -112 (-2%). The expansion on LCLU areas was in shrub land by 11 751 (37%), farm land by 30 506 (43%), waterbody by 279 (14%) and artificial surface by 365 (14%). The ESV declined in forest cover by 12%, grazing land by 2% and wetlands by 2% while farmland increased by 30%, shrub land by 27% and water body by 12%. The ESV and the total population ratios declined from \$80.4, 63.8 and \$46.0 million in 2000, 2010 and 2016 respectively. Perfect positive correlation was on LCLU change and ESV, population and households in crop farming, livestock keeping and bioenergy use.

5. The study found that afforestation, reforestation and retaining natural growing trees are the major restoration interventions used along the coastal zone of Tanzania. In some cases, eviction of forest invaders are executed to allow disturbed forests to regenerate naturally. Socioeconomic and climatic factors significantly have negatively affected artificially planted and natural regenerating trees ($p<.050$). Indeed, there were limited understanding on the ecological values of restoration interventions. Lack of improved planting materials and local community overdependence on forests resources significantly impede restoration interventions.

The implications of the findings is that human activities disturbances affect the structure of ecosystems and services at large. Population and socioeconomic activities are the main drivers and have increased demand, as

a result the coastal ecosystems is losing ESV largely. If not abetted, there is a danger of further impairments on these ecosystems. Therefore, this study advise 1. To regulate population and socioeconomic activities to avoid further negative impacts of coastal LCLU change. 2. To improve the current restoration interventions by ensuring that the socioeconomic and ecological interplays along the coastal zones are fully addressed. 3. To investigate and establish minimum allowed crop-agriculture and livestock grazing integration equilibrium point in the coastal sites is needed to make socioeconomic activities a solution with no harm on coastal ecosystems.

Keywords: Agriculture Disturbances; Livestock Disturbances; Coastal Ecosystems; Tanzania

CHAPTER ONE: INTRODUCTION

1.1. Definitions of Key Terminologies

Disturbance is defined as "any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment" (Averill et al. 1994). Some disturbances are naturally occurring such as the invasion of insects while others are human induced, for example crop agriculture and livestock grazing. The process of disturbance, response, and recovery changes the current state of the ecosystem because of changes in some species, which get lost and or introduced in the ecosystem (Averill et al. 1994).

Land-use change is the major human influence on habitats and can include the conversion of land cover (e.g. deforestation), changes in the management of the ecosystem or agro-ecosystem (e.g. through the intensification of agricultural management or forest harvesting) or changes in the spatial configuration of the landscape (e.g. fragmentation of habitats) (FRA 2005; 2018).

Ecosystem services are the many benefits, large and small, direct and indirect that ecosystems provide to people (Costanza et al 1997; Millenium Ecosystem Assessment (MEA) 2005). These consist of all the natural products and processes that contribute to human well-being, as well as the personal and social enjoyment derived from nature.

Forest is defined as the land with tree crown cover (or equivalent stocking level) of more than 10 percent and area of more than 0.5 hectares (Forest Resources Assessment (FRA) 2018).

Forest ecosystem is defined as a set of biotic and abiotic components of the ecosystems that provide the critical resources, which support the livelihoods of people worldwide (Smith et al. 2014). Human life dependence on forest ecosystems is authentically evident because forest plays important roles, which range from ecological functions (regulating the climate and water resources and by serving as habitats for plants and animals);

provision of goods (wood, food, fodder and medicines); and used for recreation, spiritual values and other services (Wenhua 2004; FRA 2005; Kideghesho 2015; URT 2015).

Forest degradation is defined by the Food and Agriculture Organization of the United Nations (FAO) as changes within a forest that affect the structure and function of the stand or site and thereby lowering its capacity to supply products or services (FAO 2011; 2015).

Ecological restoration refer to the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (Mishra and Tripathi 2015)

1.2. Human Activities Disturbances on Ecosystems

Human disturbances probably have the most drastic effects on succession in both forested and non-forested terrestrial ecosystems globally (FAO 2015). In recent decades, the socioeconomic activities are rapidly degrading ecosystems all over the world and have resulted in people competing for increasingly scarce natural resources (FRA 2005). Human activities have caused land cover change, and in particular deforestation, which have a significant impact on biodiversity and ecosystems services loss mostly in the tropical countries including Tanzania (URT 2015). Activities such as crop-agriculture and livestock-over grazing, land clearing and logging for investments and developmental activities have dramatically changed the composition, structure and function of ecosystems particularly of forest around the world (FAO 2015; Keenan et al. 2015). Human activities are frequently leading to forest conversion or degradation forms locally and globally (FRA 2005; Kideghesho 2015). The world is experiencing massive changes on croplands, forests, grasslands and wetlands because of changes on land use due to conversion of ecosystems mainly forest and grasslands into cropland and pasture or grazing lands (FAO 2015).

Activities associated with expanded settlements, crop-agriculture, overgrazing, firewood and charcoal production, uncontrolled fires, timber extraction and development of infrastructure/industry affect forest ecosystems globally (Kissinger, Herold and De 2002). In Tanzania, clearing land for agriculture expansion and livestock grazing, pole cutting, charcoal burning, timber harvesting and urbanization are the major factors contributing to forest loss like in many tropical countries (Ahrends 2005; Hartter et al. 2011). These activities disturb the natural settings of forest ecosystems (Smith et al. 2014). As a result many forests existing nowadays are classified as disturbed and secondary following natural and human activities on forests ecosystems (Lindsell et al. 2015; Yirdaw, Tigabu and Monge 2017).

Crop-agriculture and animal husbandry disturbances in particular are the predominant activities causing land use change and impacts on ecosystems services (Smith et al. 2014). Forest ecosystems disturbances continue to be a challenge to forests management in many countries (Anna 2009; Agherkakli et al. 2010; Amato et al. 2011). These two activities are the central challenge for forest scientists, policymakers, and resource managers locally and globally (Thompson et al. 2016). The major reason for this challenge is that agricultural and animal husbandry land use changes are complex and are the results of the interactive effects on socioeconomic and ecological structures (Thompson et al. 2016). Therefore, understanding the impacts of human activity disturbances on environmental settings requires integrative analyses of multiple processes and outcomes (Thompson et al. 2016).

1.3. Impacts of Human Activities on Forest

Forest ecosystems are the main drivers and providers of a wide range of functions and services globally (FRA 2005; FAO 2015), it is imperative to widely use forests in addressing ecosystem structure and ecosystems services dynamics because forest ecosystem are under pressure from the increasing human activities and demand of forest-based products and services. Human activities on forest ecosystems have contributed significantly to changes on the structure, functions and services of these ecosystems in many parts of the world (Pfeifer et al. 2013). The impacts of various activities on forests are destruction of tree species, biomass, biodiversity loss and soil fertility loss (Godoy et al. 2011; Chu and Guo 2013). Human activities cause conversion of land uses, which influence forest ecosystem structure, function and biota that in turn affect ecosystem supply and other ecosystems too (Ziter, Graves and Turner 2017).

The kind of disturbance and the associated cumulative severity affect the distribution and structure of vegetation in an ecosystem (Amato et al. 2011; Yirdaw, Tigabu and Monge, 2017). Disturbances can affect part of trees, whole trees, population, community and the entire components of forest ecosystems hence the floristic structure, composition, functions, diversity, succession development and patterns of resource availability(e.g. soil nutrients) (Agherkakli, Najafi and Sadeghi 2010; Amato et al. 2011; Alexander 2012; FAO 2011).

Moreover, excessive disturbances cause forest degradation through affecting forests biodiversity, productive capacity or protective functions (Bahamondez and Thompson 2016). Human pressure and poor management continue to threaten tropical forests particularly in Sub-Saharan countries, Tanzania inclusive (FAO 2015; Keenan et al. 2015; Halter 2016). For example, agriculture (crop production and livestock grazing) is continuously documented the major driver for land use changes in Sub-Saharan Africa that contributed to massive loss of the remaining forests (Anon 2007). Indeed, human activities contribute to significant loss of forest cover in Tanzania. For example, forest loss reached up to 403,350 hectares in 1990s and 372,871 hectares in 2015 (FAO 2015; Keenan et al. 2015; Kideghesho 2015; URT 2015). Of this recorded loss, 38,800 ha (equivalent to 6%) represents deforestation of coastal forests between 1990 and 2000 (Ndang'ang'a et al. 2008). Moreover, documentation show that Tanzania lost 2.00 million hectares (Mha) of tree cover between 2001 and 2016 (Global Forest Watch (GFW) 2016). This loss is equal to 7.6 % of the country area's tree cover extent in 2000, and equivalent to 149Mt of CO_2 emissions (FAO 2015; GFW 2016). This loss is alarming and calls for field empirical evidence to understand their impacts on forest structure and service values in comparison to other land cover and land use.

1.4. Crop-agriculture and Animal Husbandry Disturbances

1.4.1. Crop-agriculture

Coastal ecosystems especially forests are overexploited because of unsustainable use of coastal resources as well as pressures from the growing agricultural activities depending coastal populations (Kebede, Brown and Nicholls 2010). Clear tree felling from intensive agriculture is associated with timber removal, and with major disturbances by using powered machinery contributes to the opening of larger sites for crop production (Attiwill 1994) making ecosystem vulnerable to disturbances. The detrimental effects of agricultural practices is deforestation, which in turn affects soils in ways such as erosion, desertification, salinization, compaction, lowering soil structure quality and loss of soil fertility (Anyanwu et al. 2015). Deforestation usually led to land degradation and ecological imbalance especially when clearing and burning are accompanied in deforestation methods in preparation of land used for crop production (Anyanwu et al. 2015). Agricultural activities (Tomppo et al. 2014), disturb forests soils and cause high scale severity in soil and vegetation properties (Elliott, Harper and Collins 2011). It is obvious that the ongoing agricultural practices of clearing land for crop production and improved pasture management by using uncontrolled fire accelerate the problem of forest disturbances.

1.4.2. Effects of Grazing on the Ecosystems

Livestock grazing disturbances in forests is a concern in forests management because the life of every kind of human beings and civilization all over the world show well connections between these activities and ecosystems mainly forests (Runsten et al. 2013; Kideghesho 2015; FAO 2015; World Bank 2016). Livestock grazing affects species composition, ecosystem function, and socioeconomic value of forests (Ayers and Lombardero 2000; Nasi and Van Vliet 2009). Literature show that livestock induced disturbances might be among the major factors constraining regeneration and recruitment of species in terrestrial ecosystems.

The physical structure of plant communities is often changed by grazing. Defoliation by grazing herbivores alter plant height and canopy cover, and change species composition to include structurally different types of plants. Trampling may also change the structure of plant communities by breaking and beating down vegetation (Fleischner 1994). Indeed, defoliation can promote shoot growth and enhance light levels, soil moisture, and nutrient availability (Frank et al. 1998). However, grazing animals can decrease flower and seed production directly by consuming reproductive structures, or indirectly by stressing the plant and reducing energy available to develop seeds. Grazing animals can also disperse seeds by transporting seed in their coats (fur, fleece, or hair), feet, or digestive tracts (Lacey et al. 1992; Wallander et al. 1995). For some plant species, grazing may facilitate seed germination by trampling seed into the soil.

Trampling and pawing disturb the soil and in some cases completely destroy soil crusts (Fleischner 1994; Belsky and Gelbard 2000). Reduced vegetative cover and disturbed soil surfaces results into increased wind and water erosion (Belnap and Gillette 1998). In addition, the effect of trampling is compaction of soils, which damages plant roots and causes roots to become concentrated near the soil surface (Dormaar and Willms 1998). These changes may prevent plants from acquiring sufficient resources for vigorous growth (Belsky and Gelbard 2000). The hoof-action of large grazing animals can incorporate plant material into soils and increase organic matter. Grazers enhance mineral availability by increasing nutrient cycling within patches (Holland et al. 1992). Also, the organic components of feces and urine from grazing animals can build soil organic matter reserves (Runsten et al. 2013). These organic components results into soils having increased water-holding capacity, increased water-infiltration rates, and improved structural stability, which can decrease soil loss by wind and water erosion (Hubbard et al. 2004).

1.5. Coastal zone Ecosystems Disturbances

This study presents the impacts of human activities on ecosystems located within 100 kilometers (in this study referred as coastal ecosystems) from the Indian Ocean. In this study, the coastal zone of Tanzania, was selected purposely because the top 10 regions, which were responsible for more than half (82%) of all land cover loss between 2000 and 2016 are found within this zone. Pwani Region (where this study was conducted) had the largest tree cover loss at 274 thousand hectares (kha) compared to an average of 66.6kha in the country (GFW 2016). This loss if not addressed seriously, will continue to worsen land cover mainly forests and woodlands by 2050 (IUCN 2012; URT 2015; Kideghesho 2015). Indeed, the ongoing loss of land cover in Tanzania will likewise affect about

850km^2 of the coastal line ecosystems that are rich in tree species' diversity but classified as threatened ones (IUCN 2012; Mligo 2015a; URT 2015).

In order to compare the impacts of disturbances across land uses, Uzigua Forest Reserve was purposely selected as a study area (Banda, Schwartz and Caro 2006). The reason for using this forest ecosystem is that, this forest is within the region characterized by a high level of species endemism, a severe degree of threat (because of human invasions for crop-agriculture and livestock grazing) and exceptional diversity of its plants (Ndang'ang'a et al. 2008) as also supported in Mori (2011). This study considered that the impacts of human activities in Uzigua reseserve are different across sites, which were not disturbed by human activities (closed forests sites), used as a control, and those disturbed by crop-agriculture and grazing. This kind of study is important in management of coastal ecosystems because identifying human activities, which influence degradation of forests, impact ecosystems services value and restoration practices offers input/solutions for sustainable management of coastal ecosystems (Giliba et al. 2011).

This study considered that an understanding of ecosystems structure under human influences is important, as management of any ecological systems requires detailed information on dynamic processes occurring at spatial and temporal patterns (Webb et al. 2007). Understanding disturbances and recovery can provide clues for treatments that are useful to halt or reverse degradation and assist in the recovery (Polster 2016).

However, empirical evidence about human influence on ecosystems structure and services of tropical coastal forests is lacking. Moreover, an understanding on how disturbances have affected the coastal area abiotic factors (soils) and biotic factors (vegetation stand and diversity structures, and services) is limited. The major reasons for limited studies on disturbance lie on the fact that it is difficult to quantify and monitor effects directly (Hitimana, Kiyiapi and Njunge 2004; Backéus 2006). Therefore, this research was conducted to systematically assess and compare the ecological impacts of disturbances using vegetation structure and soil status using by a combination of extracted satellite images to get classes of land cover and land use changes, and ground verification on vegetation and soil indices (Peres, Barlow and Laurance 2006).

In this presented work, human activity disturbances on the structure of ecosystems and socioeconomic activities that have linkages to degradation of land cover such as deforestation were identified and presented. The study was conducted under the laid principle that *"If we ignore the lessons of ecosystem disturbances and recovery, the concepts of sustainable development and that of conservation of biodiversity are meaningless"* (Averill et al. 1994:13).

1.6. Problem Statement and Justification

Sustainable ecosystem production and management require an understanding of disturbance effects on structures and the consequences in ecosystems services values (Ares et al 2007). This understanding is crucial because ecosystem disturbances such as those, which emanate from deforestation are human-caused and affect biodiversity, carbon storage, soil erosion, habitat connectivity, and soil nutrient dynamics (Corbin and Holl 2012; Widianingsih, Theilade and Pouliot 2016). The existing research and development works on disturbed ecosystems' structure and properties state that disturbances form an integral part of ecosystems. Human disturbances influence ecosystem structural composition and functioning and thus can be important for maintaining biological diversity and facilitating regeneration or biodiversity losses (FAO 2011).

The documentation in FAO (2006); Cathcart et al. (2008) show that there is a considerable work done to address ecosystems disturbances and degradation in different parts of the world. However, FAO (2006), Cathcart et al. (2008) and FAO (2009) indicate that there is incomplete, inconsistent, unreliable data and limited information on the impacts of human ecosystems disturbances to establish the status and structures of many ecosystems particularly forests and soil properties of different habitats. Essentially, Cathcart et al. (2008) suggest that there remains much work to assess the impacts of disturbances on ecosystems structures especially in the tropics. Studies are required to investigate and analyze ecosystems structural changes to understand human disturbances and their implications on services values for sustainable coastal ecosystem management (URT 2013; Keenan et al. 2015).

Therefore, understanding the impacts of human disturbances brought by crop-agriculture and livestock grazing on the coastal ecosystems of Tanzania is crucial. This understanding is important because agriculture and livestock grazing are the major land use categories easily identified in disturbances and patch size distributions relative to those occurring naturally (North and Keeton 2008; Sturtevant et al. 2014). Indeed, this work, attempted to understand the human disturbances on soil physical and chemical properties, forest floristic composition, species diversity and regeneration status in the coastal forest ecosystems (Hitimana et al. 2004, Sapkota, Tigabu and Odén 2009). In addition, the study represents the interplays between socioeconomic activities, land use change and impact on ecosystem services values as well as restoration interventions in the coastal forests of Tanzania.

1.7. Objectives of the Study

1.7.1. The Main Objective of the Study

This study aimed at understanding human disturbances on the coastal ecosystems structure by assessing factors, which affect forest structural parameters, soil physical and chemical properties, and ecosystem services values of the tropical coastal forest ecosystems in Tanzania.

1.7.2. Specific Objectives

Specifically, this study aimed: -

1. To investigate human disturbances on coastal forest ecosystem soils
2. To investigate the effects of human disturbances on coastal forest ecosystem vegetation structure
3. To establish the relationship between forest vegetation structure and soils in the disturbed and intact sites of the coastal ecosystems
4. To evaluate the drivers and effects of socioeconomic activities on changes of coastal ecosystems service values

5. To investigate the current restoration interventions and factors affecting restoration interventions of the disturbed coastal forests in Tanzania

1.7.3. Study Questions

1. How soil physical and chemical properties are characterized under crop-agriculture and grazed land uses compared to intact (closed-disturbed) forest sites?
2. How forests disturbances contribute to affect the floristic structure, composition and diversity patterns of coastal forests?
3. How soil properties and forest vegetation structure are canonically related under crop-agriculture and livestock disturbed sites compared to intact sites of forests?
4. How socioeconomic activities have caused land cover and land use changes and effects on ecosystem services values across 2000, 2010 and 2016, of the coastal ecosystems in Tanzania?
5. How restoration interventions are executed to address recovery of the disturbed coastal forests in Tanzania?

1.7.4. Activities Performed

1. Reviewing literature on natural forests, human disturbances mainly crop-agricultural disturbances and animal husbandry disturbances, land use and land cover changes, as well as restoration of disturbed forests
2. Collecting and analyzing soil samples from crop-agriculture, livestock grazed and intact forests patches to know the status of physical and chemical properties of soils under these different land uses
3. Carrying tree species inventory on crop-agriculture, livestock grazed and intact forest sites to gauge and establish the variation of species across disturbed and non-disturbed sites
4. Extracting and preparing land cover and land use change maps from satellite images, and collecting socioeconomic data to gauge the impacts of land use changes on ecosystem services of coastal ecosystems

5. Field surveys to understand activities for restoration and factors which affect the implementation of restoration interventions of disturbed sites along the coastal zone of Tanzania

1.7.5. Expected Outputs

i. Soil physical and chemical properties under crop-agriculture, grazed land and intact forest sites investigated and documented
ii. The current structure of forest vegetation status under crop-agriculture, grazed land and intact forest sites analyzed and documented
iii. The changes in status of land cover and land use, socioeconomic activities and ecosystem services values investigated, analyzed and documented
iv. The mechanisms and factors affecting restoration of the disturbed coastal forests identified, gauged and documented
v. Suggestions for improving the current frameworks for management of coastal forest under different land uses recommended and documented

1.8. Significance of the Study

This work contributes to provide the basic initial research findings to improve an understanding about human interference and sustainable management of the tropical forest ecosystems of Tanzania. This is the first kind of study, which established the inner relationships between soil properties and above ground forest structure in Tanzania. The study is also unique in the sense that it involved exploration of how socioecological systems interplays to address sustainable coastal forest in Tanzania. The study on forest disturbance and implication on biophysical and ecosystem services dynamics provides fundamental information, crucial for determining the sustainability of tropical coastal forest resource utilization and management (Kissinger, Herold, and De 2002; Preston 2006; Hjältén et al. 2017). The research findings are crucial in providing important initial basic data on the structural status and services of tropical coastal forests using Uzigua Forest Reserve following the past fifty years of human activities encroachment and degradation; a condition needed for re-mapping and exclusion of human activities from this forest as a step towards re-establishing proper management options. The expectation is that public, non-governmental and community-based organizations will use the results after understanding about forest disturbances and ecosystems interplays and so incorporating human induced disturbance on forest management plans and programs.

Indeed, the results from this study contribute to the current efforts of Tanzania and many developing countries partners, and international commitments to improve the current forest management programs under recognition that tropical coastal forests are among the biodiversity rich hotspots but threatened because of human based factors. The suggestions in this work are expected to reflect processes operating over multiple scales crop-agriculture and animal husbandry in the tropics. In addition, this work produced a PhD thesis, which forms a significant contribution to the partial fulfillment for the completion of a PhD in Ecology as offered by the College of Life Sciences, Fujian Agriculture and Forest University, Fuzhou, Fujian, China.

1.9. Dissemination of Research Findings

The suggestive research findings dissemination includes thesis production, scientific papers/articles production, public meetings, local and international conferences, lectures, group discussions, case study presentations and demonstrations.

CHAPTER TWO: LITERATURE REVIEW

2.1. Human Activity Disturbances on Ecosystems

Human activities have pronounced impacts on tropical forests (Islam and Weil 2000) as they affect the forests which are characterized as forests with low ability to buffer from the negative impacts of disturbances and degradation (Islam and Weil 2000). Human activities have been disturbing forests ecosystems in Tanzania before and after independence (in 1960s) (Kideghesho 2015). Soils developed under natural forests in the coastal zone of Tanzania have been degraded by land-use changes for the past 50 years (Silayo et al. 2006; Kideghesho 2015).

Crop-agriculture and livestock grazing have been stressing coastal forests by contributing to soil disturbances and then degradation. These activities lead to the decline in soil quality with continuous reduction of coastal forest productivity (Islam and Weil 2000; Bahrami et al. 2010). Cultivation of land and livestock grazing cause forest disturbances and degradation through affecting the productive capacity, loss of biodiversity, carbon, or protective functions (Bahamondez and Thompson 2016).

2.2. Deforestation in the Coastal Zone of Tanzania

Deforestation and impacts on forest ecosystems along the coastal zone of Tanzania have been exacerbated by the rapid population growth of about 2.7% between the 2000s and 2010s (URT 2016). This population increase is associated with the increased demand for crop and livestock production in the coastal zone of Tanzania. The current operations in crop-agriculture and livestock grazing cause forest ecosystem disturbance particularly in the coastal zone of the country (UNDP 2004; Mligo 2015). The problem is disturbing tropical coastal forests because human activities are accelerated by climate change impacts forcing crop growers to expand their farms by encroaching forest ecosystems while the number of livestock is increasing too (Pan et al. 2013). This pressure puts at risk the coastal forests that are the most affected forest ecosystems in Tanzania (Mligo 2015).

Livestock grazing is putting more pressure on coastal forest ecosystems in 2010s than before because there is a massive shift of livestock keepers from central and northern parts of the country to the coastal zone searching pasture and water mainly for cattle (URT 2016). For example, cattle population in the Pwani Region alone increased from 122 300 in 2002/2003 to 470 000 in 2011/2012 equivalent to an increase of 280 % (URT 2016). This increment puts more pressure on coastal forests located within 100 kilometers of the Indian Ocean, the zone where about 850 km^2 of forest ecosystems are located (Francis and Bryceson 2001).

The information about forests disturbances, soil chemical and physical properties is important in the management of soil resources (Paillet et al. 2010; Alexandre et al. 2013). This assessment is used as a step to identify and set a baseline for investigating local soil properties dynamics hence focusing on conservation priorities, and enabling effective design of forest management efforts (Defries et al. 2010). However, documentation by FAO (2006); Nasi and Van Vliet (2009) show that more work is needed to investigate and generate reliable information about soil heath to provide a good stance for sustainable forest ecosystem management particularly in developing countries, Tanzania inclusive. Therefore, this study contributes to generate information crucial in monitoring of soil attributes and variability (Paillet et al. 2010; Rezapour 2014). An understanding about forest soils health is required as it is lacking in many existing studies such as Ndang'ang'a et al. (2008); Mligo et al. (2011); Mligo (2015b).

2.3. Human activity disturbances on soil properties

Human disturbances and impacts on soil properties is a concern of many tropical soils and a challenge in the management of coastal forests in many developing countries including Tanzania (De Caires et al. 2014). This challenge is persistent because human activities affect land use choices, which in turn affect the status of soil organic matter, soil structure, water and nutrient holding capacity (Gimenez et al. 2010; Tanah and Hutan 2015). It is apparently known that the electrical conductivity, organic matter, soil structure, soil texture, bulk density, water infiltration and biological community controls soil health or quality (Gimenez et al. 2010). However, these properties of soils are compromised by crop-agriculture and livestock grazing through converting forest vegetation into other land uses (Demir et al. 2007; Makineci et al. 2015).

2.3.1. Disturbances and Electrical Conductivity

Although electrical conductivity is a measure and the indicator of soil nutrients availability and loss, information on the current spatial variation of electric conductivity (μScm^{-1}), soil texture (percentage) and bulk density (gcm^{-3}) across the remaining natural coastal forests and disturbed sites in the tropical coastal ecosystems is lacking (Sudduth et al. 2003; Doolittle and Brevik 2014; Rabenberg and Kniffen 2014). Therefore, this study was conducted to address this deficit based on the basic principles that changes in land uses and management (agriculture, livestock grazing) affect the physical, chemical and biological properties of the soil (Certini 2005; Grossmann and Mladenoff 2008; Bahrami et al. 2010; Rezapour 2014). Agriculture and livestock grazing expose soils directly to solar radiation and evaporation, hence lowering the electric conductivity compared to moist soils that are protected by vegetation (Ryšan and Šařec 2008). Exposure of land to evaporation and directly solar radiations affects some properties such as salinity, which increases concentrations of electrolytes in soil water, dramatically increase the electrical conductivity depending on soil drainage and organic matter levels (Ryšan and Šařec 2008).

2.3.2. The Interplays between Soils and Vegetation Structures

In forest ecosystems, farming activities and livestock grazing affect vegetation which in turn disturb the ecological relationship between the above and below ground systems (Islam and Weil 2000; Demir et al. 2007; Bahrami et al. 2010; Charan et al. 2013). As these activities affect the vegetation biodiversity and results into affecting the overall soil health or quality (Charan et al. 2013; Bahrami et al. 2010; Gimenez et al. 2010; Tanah and Hutan 2015).

2.3.3. Disturbances on Soil Texture and Bulk Density

Crop-agriculture and livestock grazing affect soil texture and soil bulk density (USDA 2008). The interplay between electric conductivity, soil texture and bulk density is important in forest management because any activity, which affects any one of these three factors, affects the other two components. For example, cultivation in forest ecosystems contributes to increase the bulk density of soils, the process related to the decrease of soil pore volume, a loss of pore continuity, and induced changes in soil aeration, poor soil water retention and changes in hydraulic conductivity and hence affecting the electric conductivity (Rezapour 2014). Spatial variation of electric conductivity, soil texture and bulk density (gcm^{-3}) across intact forest sites, crop-agriculture and livestock grazing sites is based on the background that (i) human activities contribute to the dynamics of soil quality (Gimenez et al. 2010; Paillet et al. 2010); (ii) human induced activities influence the soil attributes related to soil processes such as oxidation, mineralization, and leaching (Rezapour 2014). Consequently, these processes modify the breakdown of salts into positively and negatively charged ions; and (iii) the accumulation of salt ions in the soil affect the soil electric conductivity in different land uses (Makineci et al. 2015).

2.3.4. Disturbances and Soluble Bases

Although limiting forest ecosystems disturbances is often viewed as essential for maintaining biodiversity, documentation show that there is incomplete, inconsistent, unreliable and limited information about the interplay of disturbance impacts on carbon, nitrogen and phosphorus in the coastal ecosystems under various forms of land uses (Bell et al. 2016; FAO & JRC 2012). Research to support positive or negative impacts of disturbances on coastal forest soil nutrients content is lacking, especially in African countries including Tanzania (FAO 2010). More specifically, comparative studies on the impacts of crop-agriculture and livestock grazing disturbances on the status and variation of carbon, nitrogen and phosphorus along the coastal forest ecosystems are poor. It is important to assess disturbances effects on soil non soluble bases and the interplays between inputs to soil organic matter. This interplay is important because disturbances affect by increasing or lowering decomposition of organic matter/plant litter and animal excreta (Golluscio et al. 2009). Disturbances also affects outputs from soil organic matter pools by accelerating the mineralization and leaching of nutrients such as nitrogen (Golluscio et al. 2009).

2.3.5. Disturbances and Soluble Bases

An understanding of different levels of soil calcium, magnesium, potassium, and sodium, is important in the management of forest ecosystems (Laiho, Penttilä and Lainest 2004; Pulla et al. 2016; Sarmadian, Keshavarzi and Malekian 2010), because cation exchange capacity highly influence vegetation growth in forest ecosystems (Pal, Panwar and Bhardwaj 2013). Despite the importance of these elements, little is understood about their patterns and variability in tropical coastal forest ecosystems particularly on crop-agriculture and livestock grazed land uses (Yavitt et al. 2009). There is limited documentation on the status and variations of cation exchange capacity and percentage base saturation in the closed forest, forestland sites subjected to crop-agriculture and livestock disturbances especially in the tropical coastal forests like in many other tropical forest ecosystems (Yavitt et al. 2009).

Disturbances on the tropical coastal forests affect soluble bases (Johnson et al. 2009). As a result, many of the tropical forests are characterized by limited soluble bases (Heineman et al. 2016). The variation of nutrients exists between different ecosystems because of processes such as pedogenesis variability of parent rock materials and land uses (Pulla et al. 2016; Sarmadian et al. 2010; Kaspari and Yanoviak 2008). While cutting down of native vegetation to convert forestland into farms counted as one of the processes that add soil nutrients, yet this addition is considered a temporal return of mineral nutrients in soil stock (Moreira and Fageria 2009). Thus, any conversion of natural vegetation into crop or grazing lands contributes to alter some soil nutrients (Moreira and Fageria 2009). The depletion of nutrients is severe especially when fertilizers are not used as one of the corrective measures (Moreira and Fageria 2009).

Unfortunately, crop-agriculture in the coastal forest reserves is practiced without additional of fertilizers, while literature on the differences in nutrients status between and across intact forests, agriculture and livestock disturbed sites is limited. It is known that forest disturbances brought by human activities or natural processes affect vegetation, which in turn influence nutrients biogeochemistry through variation in the quantity and chemistry of plant litter (Hobbie et al. 2005). Forest disturbances affect litter accumulation, thus lowering the capacity of forest ecosystems to slow soil erosion and mineral nutrients leaching (the most factors for soluble nutrients loss in the tropics) (Kaspari and Yanoviak 2008; Johnson et al. 2009).

Activities that cause land cover change for example those associated with deforestation cause soluble bases depletion and extinction of some plant species in the tropics hence limiting the development of forest ecosystems (Kirby and Potvin 2007; Vourlitis and Lobo 2015). Because of the roles played by soluble bases in controlling soil acidity and plant community welfare, an understanding about soluble elements quantities and variation is crucial in forest management (Kabrick and Goyne 2011).

The existing studies have documented on the impacts of land cover change and carbon storage (Kirby and Potvin 2007; Nave et al. 2010; Parras-Alcántara, Lozano-García and Galán-Espejo 2015). Studies on soil organic carbon conducted by Shelukindo et al. (2014), Nitrous Oxide and Methane by Ishizuka et al. (2005), and plant diversity in Mligo (2015) and (Howell et al. 2012). Although a study by Pal et al. (2013) investigated soil fertility on different land uses, documentation on the comparative differences of soluble bases across forest sites subjected to different land uses along the tropical coastal forests including those found in Tanzania is lacking. This lack of information is a challenge on the management of coastal forests in the tropics.

Inadequate information about soil soluble bases puts forests management in risk because the knowledge about the existence of forest resources is not enough to address the entire reciprocal function of soil properties and the interplays between vegetation and soils soluble bases ecosystems (Kaspari and Yanoviak 2008; Vourlitis and Lobo 2015). A study about soluble bases status and variation is important in the tropical coastal forests because these forests face pressure from human activities mainly crop-agriculture and livestock grazing (Ligate, Wu and Chen 2017). Information generated in this section is crucial in contributing on the effective management and protection of tropical coastal forest ecosystems (Kabrick and Goyne 2011).

2.4. Human Disturbances on Forest Structure

Protecting the remnant of coastal forests and recovering disturbed sites is an important worldwide concern (Potter 2014; Mligo 2015a; World Bank 2016). Common strategies used locally and globally include excluding human settlements, crop-agriculture, and livestock grazing (Redford and Fearn 2007; Navroodi 2015; Tadesse and Kotler 2013; Schieltz and Rubenstein 2016). These efforts aim to allow the regeneration of trees and other vegetation since tropical forests have a pronounced power of self-maintenance through regeneration (Sundarapandian and Swamy 2013). However, under the current exclusion management options, there is insufficient documentation about the distribution and concentration of trees diversity in response to disturbances, particularly in the tropical coastal forests.

Deforestation due to human pressures and poor forest management systems affects forest structure and ecosystems (Guerrero and Bustamante 2007; Halter 2016; Bonari, Acosta and Angiolini 2017). Forest disturbances and degradation affect the structure of forest ecosystems at large (Bargali et al. 2013). Human activities contribute to forest biodiversity decline or loss (DeFries et al. 2010). The main activities contributing to forest loss, especially in the tropics, include clearing land for crop-agriculture, pole cutting, charcoal burning, timber harvesting, and settlements (Majumdar and Datta 2014; Keenan et al. 2015; Bonari, Acosta and Angiolini 2017). Human disturbances reduce the capacity of forest to regenerate, function, and offer various ecological services (Thompson et al. 2009; Kimaro and Lulandala 2013; Joyi et al. 2015).

However, documentation shows that some degree of disturbances are actually beneficial, as they contribute to the increase of biodiversity and nutrient circulation. These disturbances are thus considered important for long term sustainability and productivity of most ecosystems on earth (Kalaba et al. 2013; Kijazi et al. 2014). Indeed, disturbances are important in the modification of forest structures (i.e., stand parameters and species diversity), thus helping forests to undergo successional stages and maintain values. Unfortunately, in many cases these structures affected by natural and human activities under varied environmental conditions (Bargali et al. 2013).

This study was conducted on the forests located along the coastal zone of Tanzania. This ecological area is rich in biodiversity as it has about 190 forest species, of which 92 are endemic (Howell et al. 2012). However, as in many other tropical forests, farming (crop-agriculture), livestock grazing, timber harvesting, and charcoal making threaten forests. As a result forests are disappearing at an alarming pace (Devi and Yadava 2006). Because of these human activities, tropical coastal forests located in the coastal zone of Tanzania have lost about 69% of their primary vegetation (Howell et al. 2012). If not abetted, further degradation will continue to threaten about 1500/300,000 (i.e., 0.5%) of global vascular plants found in this zone (Mligo et al. 2009). Crop-agriculture and livestock grazing are the main human activities accelerating the rate of coastal forests degradation in Tanzania (Kimaro and Lulandala 2013).

Encroachment through these activities threatens the coastal forests, which cover an area of about 800 km2 along the coastal zone of the country (Kimaro and Lulandala 2013). These activities alter the distribution and structure of the forests (Ares et al. 2007; Amato et al. 2011). Changes in spatial and temporal patterns, and the subsequent regeneration capacity put forest management efforts in jeopardy (Huang et al. 2003; Merganic et al. 2012). Yet studies on how coastal forests, such as those comprising the study area, respond to crop-agriculture and livestock grazing disturbances are not available.

This first study compares the regeneration of trees across land uses after crop-agriculture and livestock grazing exclusion in tropical coastal forests of Tanzania. This paper presents tree composition, structure and regeneration potential across land use sites (Bharathi and Prasad 2015). The study used data from the Uzigua Forest Reserve located along the coastal zone of Tanzania to serve an important stage and contributing measure in the planning for sustainable coastal forest management (Guerrero and Bustamante 2007; Sundarapandian and Swamy 2013). Therefore, this section in this work explored and presents findings to gain an understanding of trees' responses after disturbances and the exclusion of human activities because knowledge of forest structure, including tree regeneration, is a critical determinant of forest direction in order to attain sustainable management (Bargali et al. 2013).

2.5. Canonical Correlation of Vegetation and Soil Properties

Knowledge about the influence of human activities on structures and the correlation of vegetation structure (tree stand parameters and diversity indices), soil health parameters (soil physical and chemical properties such levels of calcium, magnesium, potassium, phosphorus and sodium) is important in ecosystems management (Poorter et al. 2015). This information is crucial because plants in forest ecosystems have influence on soil conditions (Gairola et al. 2012; Wagg et al. 2014). Nevertheless, information about the reciprocal relationships between soil properties and vegetation structure of the remaining and regenerating tropical coastal forest (TCF) sites is lacking (Ichikogu 2014; FAO 2015).

Little knowledge exists about the interplays between stand parameters, composition and soil physical properties in the tropical coastal forests. This deficit is contributing to put management of tropical coastal forests in jeopardy. Therefore, this piece of work address the missing relationship between above and below ground coastal forest ecosystems after exclusion of human activities (mainly farming and livestock grazing) (Poorter et al. 2015).

Human induced disturbances bring soil degradation, which are defined in this study as any physical or chemical alteration of the soil caused by different operations in forest ecosystems (Curran et al. 2003). Different process and activities occurring in forest ecosystems affect forest structural parameters by providing favorable or unfavorable conditions in forest ecosystems (Kalaba et al. 2013). Existing studies show that disturbances in forest ecosystems affect the ecological relationship between forest vegetation and soils (Cavelier et al. 1999; Brosofske, Chen and Crow 2001; Eni, Iwara and Offiong 2012).

Human activities especially those involving clearance of forests expose soils to erosion, loss of organic matter and other necessary elements useful for vegetation growth (Chen and Li 2003). Soil properties differ in different soil horizon as a result of biological and geochemical processes at different depths after human disturbances (Chen and Li 2003). Disturbances in the tropics affect forest structures (i.e. the spatial arrangements of various components of forest ecosystems) (McElhinny et al. 2005; Nizam, Jeffri and Latiff 2013). These disturbances affect the number of trees, heights of different canopy levels, diameter, spatial distribution, basal area, volume and species composition (Huang et al. 2003; Mbwambo et al. 2008; Delang and Li, 2013; Kijazi et al. 2014).

Although disturbances are reported to disrupt ecological settings, ecologically they are sometimes essential processes, necessary at some levels of intensity and periodicity for the long-term sustainability and productivity of forest ecosystems (Averill et al. 1994). There is relationship across forest structures and soil physical and chemical properties.

This relationship is based on the fact that forest vegetation determine soil properties forest systems through process which accelerate soil erosion, oxidation and destruction of biomass (Chu and Guo 2013). Thus, there is a strong relationship between disturbances on plant species composition and impacts on soil parameters (Chen and Li 2003; FAO 2009).

Human activities in the coastal forests ecosystems of Tanzania affect the structure of trees biodiversity and soils properties. These activities are forms of land uses, which have caused variation in habitat conditions characterized by biogeography and disturbance levels (Huang et al. 2003; Wagg et al. 2014; Joyi et al. 2015; Howell et al. 2012; Tomppo et al. 2014; Mligo 2015a). The vegetation disturbance is related with soil status because there is a close relationship between forest and land use management on species diversity, and soils conditions (Martin et al. 2016). For example, low species diversity in disturbed areas is associated with low values of soil elements such as carbon, nitrogen and phosphorus (Joyi et al. 2015).

Nevertheless other studies show that the diversity of tree species between undisturbed and disturbed forests sometimes is not significant (Huang et al. 2003). A study by Merganic et al. (2012) show that natural forests are not influenced by human activities rather than conditions of abiotic environment. In this case, it is important to find the correlation between forests structure and soils properties. This understanding is important in gauging the dynamics of forests structure and environmental variables (Rayfield et al. 2005).

Agriculture and livestock grazing disturb forests and cause high scale severity in soil and vegetation properties (Elliott, Harper and Collins 2011; Tomppo et al. 2014). Clear tree felling because of intensive agriculture makes ecosystem vulnerable to disturbances (Attiwill 1994). More specifically, livestock grazing affects species composition and ecosystem function by feeding and trampling on vegetation (Ayers and Lombardero 2000).

The impacts of agriculture and livestock grazing are large especially when there is agriculture intensification and reduced grazing areas (Nasi and Van Vliet 2009; Runsten et al. 2013). Within low carrying capacity of the forests, farming activities and livestock grazing destroy plant species and destruct soils. In addition, these activities expose the land into erosion and nutrients losses. Therefore it is imperative to establish information about forest structure and soil relationship in forest management because vegetation and soils are interconnected and exert symbiotic effects on each other (Ichikogu 2014; Wagg et al. 2014).

In order that the direction of the relationships of forest structure and soil status is established in the disturbed coastal forests, this work presents the first kind of findings done in the disturbed coastal forest ecosystems after human activities exclusion. This presented section was conducted under the hypothesis that there is positive relationship between above ground forest structures and soil properties of the disturbed sites in the tropical coastal forest ecosystems. This work guided by one major study question: How forest parameters (density, height, basal area and volume, and species composition and diversity) canonically correlated with bulk density, soil texture and electric conductivity across land uses.

2.6. Socioeconomic Activities and Ecosystems Services Values

2.6.1. Changes on Land Coastal Ecosystems

Change on land cover and land use (LCLU) is among the major drivers of biodiversity loss and ecosystem's degradation at both global and local levels (Nkonya et al. 2012; Kindu et al. 2016; Quintas-Soriano et al. 2016; Xu et al. 2017). The change has significantly affected almost all terrestrial biomes (Baumgartner and Cherlet 2016; Borrelli et al. 2017; Zhou et al. 2017). To underpin this, Baumgartner and Cherlet (2016), Scull et al. (2017), Temesgen, and Wei (2018) confirmed that these impacts pronounced in Sub-Saharan Africa. Besides, numerous human activity disturbances have amplified the problem by affecting the ecosystem functions and services (Cork and Shelton 2000; FAO 2011; Temesgen et al. 2018). Concurrently, changes on natural settings of land also affect the function of ecosystem and service values (Dale and Polasky 2007; Keenan et al. 2015; Yirsaw et al. 2016).

2.6.2. Drivers for Land Cover and Land Uses Change

Spatially, the coastal ecosystems across the world are among the systems that are at risk of degradation due to human interventions posed by development pressures among the coastal communities (Schmidt, Moore and Alber 2014; Santha 2015; Ligate et al. 2017). This pressure is exerted by rapid population growth and expansion of socioeconomic activities (Madriñán, Rickman and Ye 2012). Anxious socioeconomic activities are associated with unprecedented agriculture, expanding settlements, industrialization and other investments (Zhao et al. 2004; Xu et al. 2017). Weak management institutions and climate change that threaten the ecological potentials (Maitima et al. 2009; Keenan et al. 2015; Xu et al. 2017) amplify the resulting impacts.

Apparently, human based pressure and other stressors have contributed to the transformation of the naturally covered land systems (especially coastal forest) and open grassland into farmland and bush/shrubs encroachment (Fetene et al. 2015; United Republic of Tanzania (URT) 2015; Temesgen et al. 2018). The transformation in LCLU categories affects ecosystem processes and services (Fujita et al. 2013; Hu et al. 2008; Smith et al. 2014; Zhang et al. 2015a). Obviously, human decisions about land use affect the status and potentials of ecosystem service provision (Quintas-Soriano et al. 2016; Temesgen et al. 2018). Therefore, the processes and human activities, which bring changes on LCLU are continuously and severely affecting ecosystems in various biomes (Szuster, Chen and Borger 2011; Song et al. 2014; UNEP 2015). In addition, projections from various models indicate that Sub-Saharan Africa will continue to experience the fast LCLU changes because of the rapid human population pressure and regional expansion of agricultural land (Van der Esch et al. 2017).

2.6.3. The LCLU Changes in the Coastal Zone of Tanzania

Tanzania, which is located in Sub-Sahara Africa, is among the worst affected countries in terms of forest degradation. In 2015, it had a net loss of 372 thousand hectare per year, while a generalized global trend shows that during the same year, there was net gain in land cover (FAO 2015; Keenan et al. 2015; UNEP 2015). This loss placed Tanzania at number five among the top ten countries with significant annual loss of land cover concerning forest (FAO 2015; Keenan et al. 2015; Patra et al. 2015). The major reasons for this loss being: land is lost for urban development, is claimed for agricultural purposes (crops and livestock grazing), forests are harvested for small and large-scale commercial investments of timber and for subsistence and commercial fuel wood reasons (Sloan and Sayer 2015; UNEP 2015).

2.6.4. Threats to Coastal Ecosystems of Tanzania

Tanzanian coastal ecosystems provide critical ecological functions such as protection of the coastal zone and habitats of many living organisms including human beings (Luc van Hoof and Kraan 2017; Kubiszewski et al. 2017). However, human activities threaten the viability of coastal ecosystems located within the coastal zone like many other global systems (Zhao 2004; Kindu et al. 2016; Luc van Hoof and Kraan 2017). Despite of the posed threats, the documentation of the Tanzania coastal LCLU change and its implications on the dynamics of ecosystem services suffer numerous challenges. Therefore, a study that addresses LCLU change and its implications on the value of ecosystem services and human-ecosystem services interplay along the coastal zones is quite imperative (Otsuka and Place 2014).

2.6.5. Ecosystems Services Values

Previously, there were some studies conducted on LCLU and ecosystem services' value; however, they had different focuses. Costanza et al. (1997) evaluated the global ecosystem service values and laid a foundation for evaluation of global ecosystems, while Zhao et al. (2004) reported on the decline of ecosystem services. Furthermore, Li et al. (2007) showed a significant conversion of forests and grassland into shrub land and cultivated land. Literally, Maitima et al. (2009) and Temesgen et al. (2018) pointed out that land use changes largely contribute to modification of Africa's land cover. On top of that Nkonya et al. (2013) and Otsuka and Place (2014) specified farmland as a leading form of land use change in the Sub-Saharan region. Moreover, Warinwa, Mwaura and Kiringe (2016) and Temesgen et al. (2018) studied LCLU and ESV in Kenya and Ethiopia respectively.

These studies show a continuous loss of many tropical ecosystems because of land use change (UNEP 2015; Temesgen et al. 2018). However, the direction of any land cover or land use depends on the nature, location, activities and temporal variations of human activities (Rautiainen, Virtanen and Kauppi 2016). Human activities consequently affect ecosystems to the direction of gaining or losing service values (Bhagabati et al. 2014; Warinwa, Mwaura and Kiringe 2016; Yirsaw et al. 2016).

There has been a great deficit on the documentation of the interplays between LCLU change and impacts on ecosystems' service values in the tropical coastal zones (particularly in Tanzania). This deficit is challenging ecologists, economists, policymakers and the public (Costanza et al. 1997; Cork and Shelton 2000; Zhao 2004; Van der Esch et al. 2017).

2.6.6. Addressing LCLU and Ecosystems Services Values

To address this deficit, this assessment was conducted by using field surveys and satellite imagery to attempt (1) investigation of the socioeconomic activities which contribute to LCLU change in the coastal zone of Tanzania; (2) mapping changes in the area of each LCLU category in 2000, 2010, and 2016 reference years; (3) valuation of the dynamics of ecosystem service values along the coastal zone of Tanzania across sixteen years. Practically, the section presents the investigated the area and ecosystem service values of seven major ecosystems, namely: artificial surfaces, farmland, forest, grazing land, shrub land, and waterbody and wetland ecosystems. These categories were selected because they are the major suppliers of many recognized services, locally and globally (Marc, Babu and Hamilton 2005; Pan et al. 2011; Kubiszewski et al. 2017).

The information generated in this evaluation is crucial because it can validate the ecosystem service values (Cork and Shelton 2000; Zhao et al. 2004; Temesgen et al. 2018). Furthermore, this work expected to improve the current national and coastal development planning (Guerry et al. 2015). Indeed, it can promote the understanding of the mutual interplay across the dynamics of human population, socioeconomic activities, LCLU and ecosystem service values (Willcock et al. 2016). Eventually, knowledge about this interplay used as a tool for decision-making on management of coastal ecosystems (Thompson et al. 2016; Luc van Hoof and Kraan 2017; Temesgen and Wei 2018).

2.7. Restoration Interventions of Disturbed Forests

Forest restoration is recognized as a global priority (Kärvemo et al. 2017) because restoration interventions attempt to respond to deforestation and the pace of ecosystem destruction from human and natural disturbances (Corbin and Holl 2012; Jones et al. 2018). Understanding about forest restoration is crucial for developing ecologically sustainable forest management strategies (Kuuluvainen 2002; Stanturf, Goodrick and Outcalt 2007; Soulé, White and Van De Gevel 2012). This understanding is important because forests suffer from natural and human disturbances globally (Lamb and Gilmour 2005; Ribeiro et al. 2009; Mori 2011; Mishra and Tripathi 2015; Kärvemo et al. 2017).

2.7.1. The Role of Restoration Interventions

Restoration interventions attempts to reestablish the structure and services of disturbed forests after poor forest management practices (Goldmark 2014). In most cases to augment the amount and the quality of remaining natural habitats (Pinto et al. 2014) It is well known that changes in forest ecosystems especially those associated with human disturbance and degradation require human intervention to facilitate recovery (Cortines and Valcarcel 2009; Zhao et al. 2016). Human-facilitated regeneration mainly aim to promote the return of trees and other vegetation in the disturbed sites (Lamb and Gilmour 2005; Zhao et al. 2016; Kärvemo et al. 2017). The underlying principle in restoration is that, disturbed forest ecosystems especially those in the tropics are capable of returning spontaneously to their former conditions (Cortines and Valcarcel 2009; Lindsell et al. 2015).

Essentially, the restoration process of disturbed and degraded ecosystems consists of human interventions to re-establish forest ecological functions and services (Pinto et al. 2014; Jacob, Lechowicz and Chapman 2017). These interventions base on the idea that the most effective strategies are those that assist the natural recovery processes (Polster 2016). Ecological restoration in most cases refer to the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (Mishra and Tripathi

2015; Polster 2016; Lee, Cha and Moon 2017). Indeed, human interventions facilitate ecological restoration locally and worldwide (Haugo et al. 2015; Hjältén et al. 2017; Lee et al. 2017; Jones et al. 2018).

2.7.2. Interventions for Restoration

In attempts to restore disturbed forests, there are multiple existing establishments, which broadly categorized according to the level of management intervention (Jones et al. 2018). These establishments aim to facilitate regeneration of native species that could not otherwise establish by removal of abiotic constraints alone (Rayfield, Anand, and Laurence 2005; Jones et al. 2018). In the restoration projects, more frequently, reforestation and afforestation used at large or small-scale plantation (Lamb and Gilmour 2005; Thomas et al. 2014). Indeed, reforestation and afforestation are considered as one-dimensional restoration efforts mainly involving single abiotic variables, single species, or single applications (Rayfield et al. 2005).

In some cases, restorations process include exclusion of human induced disturbances to allow natural regeneration to occur because natural processes re-vegetate disturbed sites (Polster 2016). Exclusion works under the consideration that leaving disturbed sites to regenerate naturally is vital because establishment of forest species does necessary is in support by planting trees, as is often practiced in many disturbed sites (Prach et al. 2014). In this view, interventions after disturbances are beneficial in promoting forest biodiversity (Prach et al. 2014).

Most restoration studies have addressed forest disturbances, restoration and recovery. The comparatively few tropical coastal forests studies tend to focus on remnant trees in disturbed sites (Sturtevant et al. 2014); forest recovery patterns in response to divergent disturbance regimes, the natural disturbance patterns and manipulation of forest structure and development (Lamb and Gilmour, 2005; Goebel, Wyse and Gregory Corace 2005; Lin et al. 2015). Certainly, some studies present conservation and restoration actions of forests (Ribeiro et al. 2009; Dickinson 2014; Hjältén et al. 2017).

Ecological restoration of disturbed and abandoned forests have been studied by Haugo et al. (2015) and Jacob et al. (2017). Forest disturbances, recovery and succession reported in Soulé et al. (2012); Hartter et al (2011) and Mishra and Tripathi (2015). A study by Lindsell et al. (2015) documents seed dispersal and restoration; Hartter et al (2011) shows the temporal and spatial forest change while effects of restoration methods are reported in Kärvemo et al. (2017). Indeed, assessments on ecosystem management based on natural disturbances are reported well in Mori (2011). Moreover, forest restoration strategies are covered in Shorohova et al. (2011); Franklin and Johnson (2012); Corbin and Holl (2012); Jones et al. (2018) and factors affecting forests restorations in Kauano et al. (2013).

2.7.3. Goals of Restorations Interventions

Documentations show that the most used restoration interventions are (1) reforestation and afforestation as biological measures for ecosystem restoration and (2) permitting natural regeneration to occur through protection or exclusion of human activities. These strategies show that the goal of forest ecosystem restoration is to develop some ways that help disturbed sites and landscapes to ecologically recover (Goebel et al. 2005; Haugo et al. 2015).The expected outcome is land improvements on structure and land cover of forest. Unfortunately, forest cover along the coastal zone of Tanzania is declining annually (URT 2015; FAO 2015). In spite this decline, a systematic evaluation of restoration interventions/treatments on these tropical coastal forest is lacking. Furthermore, assessments on the current implementation of different restoration treatments to guide future restoration interventions are not documented. Therefore this sections attempted to fill this knowledge gaps.

Thus, this presented section examined forest restoration interventions to understand how disturbed forests are restored in Tanzania. This kind of study is important because understanding about restoration interventions is critical for forest ecosystem based management and the development of strategies to help mitigate or adopt to novel disturbance regimes (Sturtevant et al. 2014; Corbin and Holl 2012). Indeed, this understanding is critically important as it contributes to gauge restoration options of the ecologically highest plant species richness and endemism (IUCN 2012), but highly disturbed by human activities than any other forest ecosystems in Tanzania (URT 2015).

CHAPTER THREE: MATERIALS AND METHODS

3.1. Description of the Study Area

3.1.1. Location

This study was conducted in the coastal ecosystems located along the Coastal Zone of Tanzania. This zone stretches within 850km from the boarder of Tanzania and Kenya in in the north, and Tanzania and Mozambique in the south (Luc van Hoof and Kraan 2017). The coastal zone was purposely chosen because is among the areas with the leading forest cover loss in Tanzania particularly between 2000 and 2016 (Figure 1-1a). Classification of land cover and land uses firstly computed from the areas that cover S37 00, S37 50 and S37 50, S37 10 geographical coordinates. Secondly, forestland cover and land use classifications were carried out for Uzigua Forest Reserve (UFR) found in Bagamoyo and Chalinze Districts, Pwani Region in the Coastal Zone of Tanzania Mainland.

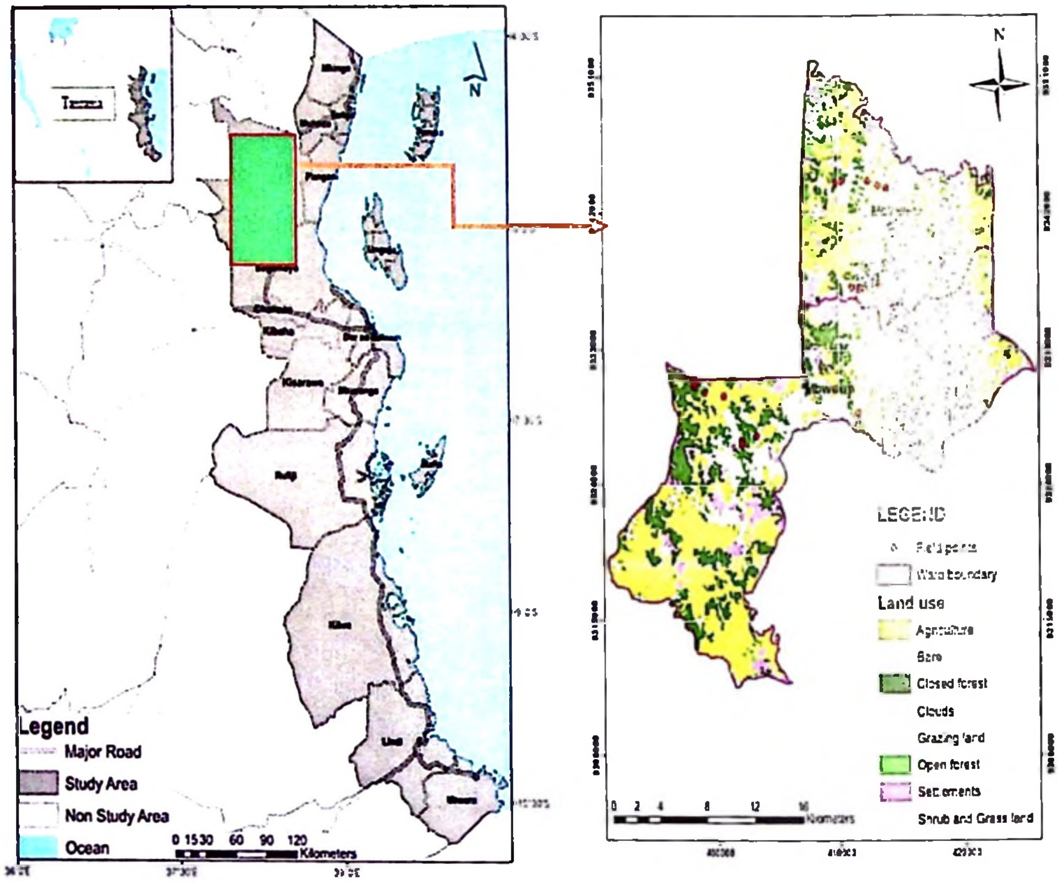
The UFR is located between 50 58 '00" S and 38 04 '00" E (Figure 1-1b) with a coverage area of 24,730 ha (URT 2015). This forest was purposely selected to represent other forest ecosystems along the coastal, which have been encroached mainly for crop-agriculture and livestock grazing. Certainly, this forest is within 100 km from the coast of Indian Ocean and thus considered among the tropical coastal forests in Tanzania (Godoy et al. 2011). The Central Government under the Forest and Bee-keeping division of the United Republic of Tanzania, Ministry of Natural Resources and Tourism manages the UFR (URT 2015). This forest reserve is supposed to be completely restricted from human use, serving for catchment and biodiversity conservation (URT 2015). Unfortunately, due to poor protection and surrounding settlements, the entire forest is affected by human based activities such as harvesting trees for fuel-wood, fodder, grazing pressure and encroachments for agriculture. These activities have significantly affected this forest; however, it is the reserve among a few remaining tropical coastal forests in Tanzania. These activities are threatening this forest like many other coastal forests, which are documented to harbor diverse plant species that make them, and hence included as one of the 34-world biodiversity hotspots that need special conservation measures (IUCN 2012; Mligo 2015a).

3.1.2. Population and Occupations

The current population of the coastal zone of mainland Tanzania is 11 549 190 representing the population of Tanga, Dar es Salaam, Pwani, Lindi and Mtwara regions (URT 2016). The Bagamoyo and Chalinze Districts have the population of 347 336 (URT 2016). The main occupations along the coastal zone are agriculture, livestock keeping, fishing, timber and charcoal production. The UFR in Bagamoyo and Chalinze purposely selected because is affected by different human activities following the fact that this forest lacked proper management since its establishment in the early 1950s. It was the expectation of the researcher that from this forest reserve and the adjacent selected villages could gauge the impacts of agriculture and grazing on tree species and soil properties as well as ecosystems services dynamics.

3.1.3. Climate

The coastal zone of Tanzania mainland receives annual average rainfall of 917.23 mm where by the peak periods of rainfall are in January to April and November to December. Uzigua forest reserve is located in the tropical and sub-humid area with 700 mm to 1000 mm rainfall. October to May is a wet season while June to September is dry. The annual minimum temperature is 22.4°C while the annual maximum temperature is 31.7°C (URT 2016). The soils are well-drained, red sand clay, loamy with brown friable top soils covered by more or less decomposed litter. The area is undulating with continuous hills with altitude ranging from 400 to 600 meters above sea level (masl) (Silayo et al. 2006). However, the current climate change and variability along the coast greatly influence temperature, rainfall, and the distribution pattern of plant species in these tropical coastal forests, and therefore the composition of the forest fragments at large (Mligo, Lyaruu and Ndangalasi 2009).



a. b.

Figure 3- 1: A map of the study area (a = The Coastal Zone, b = The Uzigua Forest Reserve)

3.1.4. Vegetation

The vegetation in coastal zone specifically the UFR is diverse, characterized with open coastal woodland dominated with *Acacia, Brachystegia*, *Combretum*, *Terminalia, Diospyrus* and *Albizia* species (Silayo et al. 2006). Also, herbs and grasses are found and grow up to 1.5m high; dominating the ground cover. Some of the common indigenous species still existing in the reserve and some remnant sites of the degraded lands are *Combretum molle*, *Tamarindus indica* and *Dombeya* sp. (Silayo et al. 2006).

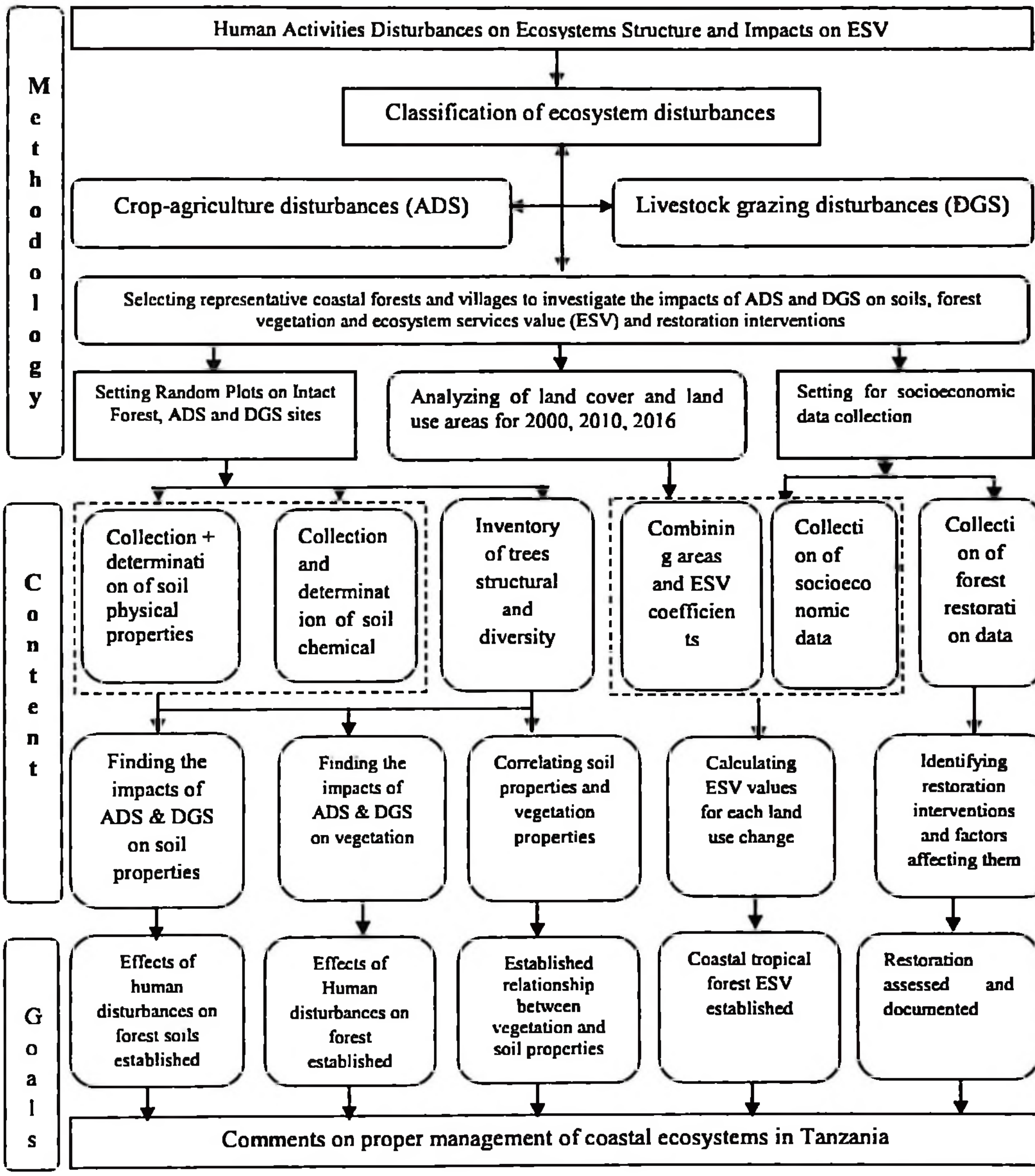


Figure 3- 2: The technical route of the study

3.2. The Technical Route of the Study

This study were conducted by collecting and analyzing soil properties, vegetation structures and ecosystems services values. Soil chemical and physical properties and tree stand and diversity parameters were used as indicators for disturbances across the land uses because these properties often change by farming and livestock grazing (Fleischner 1994). In order to understand the impacts of disturbances on forest soils and forest structure, this study followed a systematic classification of land use categories where crop-agriculture and livestock grazing were identified as the major sites for sample collection. The study used land use classes from land satellite imageries in combination with ground data verification collected in May 2016 to August 2017. This methodological combination used to address the impacts of disturbances on the ground. Based on this methodology, the effects of canopy factor was eliminated in understanding the impacts of disturbances on soil properties and forest structures and services unlike in many existing studies (Peres, Barlow and Laurance 2006). Additionally, the study investigated the current restoration practices and their challenges. Therefore, this study was conducted through the technical route as illustrated in Figure 1-2.

3.3. Land Use Classification

Prior to detailed field survey, reconnaissance surveys were conducted to get geographical coordinates, which were then used to produce stratified different land uses (LU) classes. The satellite image interpretation were used to identify areas for ground study (Backéus et al. 2006). Both satellite imagery, ground validation and calibration were used because remote sensing technology alone doesn't provide detailed analysis of land uses (Axelsson et al. 2012; Banda et al. 2006). The LU classes were identified and developed by checking on the satellite images and corresponding mean layer values, and normalized difference vegetation index (NDVI). The NDVI with support vector machine classifier were used in land use classification to generate land uses. Closed forest, open forest, shrub and grassland, agriculture, grazing land, settlements, bare and clouds were classified. These thresholds were obtained by checking the known land use type in the image and the corresponding NDVI values as well as adapting from literature and field experiences.. Three LU (closed forest (CFS), Agriculture (ADS), and grazed land (DGS) were purposively selected and used to gauge the variation and correlation of electrical conductivity (EC), soil texture (ST) and bulk density (Bd) across the three LU.

3.4. Sampling Approaches

Systematic and stratified sampling techniques were used in this study. These approaches were adopted to get CFS, ADS and DGS for their contrasting site conditions (the characteristics of disturbances), and used as representative sites to map the EC, ST and Bd (Nasi et al. 2009; IUCN 2012). Attention was paid to make sure that soil samples were collected from disturbed and undisturbed sites (FAO 2009). A comparison between variables and correlation were studied across each site subjected to different LU. To compare the variations on EC, ST and Bd, forty-seven random plots were established in the three major LU (CFS, ADS and DGS). Soil samples were collected from 1-30 cm depth because agriculture and livestock grazing activities like many human activities affect the surface and near surface layers of the soils (Gimenez et al. 2010). At each field, 10 to 15 soil samples were collected following a *zig zaga* sampling technique. Soil samples were collected with an Edelman auger as adopted from (Curran et al. 2003; Aref et al. 2011; Berber et al. 2015).

3.5. Determination of Soil Physical Properties

3.5.1. Determination of Electrical Conductivity (EC)

The EC meter with an automatic temperature compensation were used to get EC accurate results following the preparation of 1:5 (soil: water) solution put in rotary shaker for one hour. Then, the solution was put into a centrifuge at 8000-10000 rotation per minute for about 10 minutes then a clear solution was decanted and the EC measured in the decanted solution after calibrating the instrument by means of 0.01M KCl. The EC meter were used to get a rapid and inexpensive data (Sudduth et al. 2003; De Caires et al. 2014; Doolittle and Brevik 2014). All the data were analyzed for mean, standard deviation and standard mean errors. The paired samples t-test and Pearson correlation analysis were carried out by using the Statistical Package for Social Science (SPSS version 20.0) and MS. Excel computer programs. The confidence intervals of significant determination were obtained at $p < .050$ across all the LU.

3.5.2. Determination of Bulk Density and Soil Texture

Core sampling units (volume = 100 cm^3) were used to collect samples for Bd determination, where the calculations were computed by dry weight of soils divided by their volumes (gcm^3) (Burt 2004; USDA 2008). Soil samples were air dried and sieved through a 2 mm sieve and then ST (silt = 2-20 µm, clay < 2 µm) were determined by using the pipette method as described in Burt (2004). The resulting findings were presented as percent sand, silt and clay by plotting the percentage ratio of each textural class using the ST triangle (Tanah and Hutan 2015).

3.6. Determination of Non Soluble Bases (C, N and P)

Soil parameters (Total Nitrogen, (TN), Total Carbon (TC) and Phosphorus (P) were analyzed by following the standard protocols for soil analysis as follows: (i) Determination of TN was done following the Kjeldahl acid-digestion procedures (Kjeldahl 1883); (ii) Soil TC was analyzed by the Walkley-Black Procedures where by Potassium Dichromate ($K_2 Cr_2 O_2$) and concentrated Sulphuric acid (H_2SO_4) were used to produce the reaction and products as shown in this chemical reaction $2Cr_2O7^{2-} + 3C^0 + 16H^+ \rightarrow 4Cr^{3+} + 3CO_2 + 8H_2O$ (Walkley and Black 1934). In computing the results, a correction factor of 1.33 applied to adjust the organic carbon recovery because of incomplete oxidation in Walkley-Black combustion procedures. (iii) Available phosphorus was determined by the Bray-II method (Bray and Kurtz 1945). Statistical Package for Social Sciences (SPSS) version 20.0 used together with MS-Excel computer program to run statistical analysis for getting mean and t-values at 5% significance level for TN, TC and P differences between and across CFS, ADS and DGS.

3.7. Determination of Soluble Bases (Ca, Mg, K and Na)

A combined glass–calomel electrode used to determine the pH of aqueous suspensions (1:2.5 soils: solution ratio to aid on deciding on the proper techniques for further analysis of samples. Ammonium Acetate (1M NH4OAc) (pH 7.0) was used to extract exchangeable Ca, K and Mg. Potassium content was determined by flame photometer (Moreira and Fageria 2009). Ethylenediaminetetraacetic acid (EDTA) titration done to measure Ca and Mg from the soil solution. Cation exchange capacity (CEC) in (cmol/kg) and percentage base saturation BS (V%) were calculated following the methods by Yeshaneh (2015) and Blanchet et al. (2017) as shown in equation (i) and (ii)

CEC= $\sum$(Ca, Mg, K and Na) (cmol/kg) .... (Eq. 1)

Where, Ca, Mg, K and Na exchangeable are in cmol/kg

Base saturation (BS) (V %) = $\frac{\sum(Ca, Mg, K, Na)}{CEC} \times 100\%$ ....(Eq.2)

Laboratory results were further subjected into Statistic Package for Social Science (SPSS version 20.0) software and Microsoft Excel computer programs for computation of means, standard deviation and t-values at 5% significance interval. These outputs were compared across the land uses and elevations to gauge the differences and similarities of soluble bases.

3.8. Tree Data Collection

Inventory field data collection carried out from May to August 2016 and repeated on June to August 2017 using the stratification field inventory approaches (Jayakumar et al. 2011; Tomppo et al. 2014; URT 2015). Land use (LU) classes identified and developed by checking on the images and corresponding mean layer values and the normalized difference vegetation index (NDVI). The NDVI used together with the support vector machine (i.e., a machine used for classification and regression analysis) for image processing and production of LU classes (Ustuner, Sanli and Dixon, 2015). The closed forest, open forest, shrub and grassland, agriculture, grazing land, settlements and bare lands were classified. From these classes, closed forest (CFS) agriculture (ADS) and grazing lands (DGS) purposely selected.

In addition, local people supported the identification of LU sites based on the history of crop and livestock production activities in the study area. The selected LU sites surveyed to gauge the response of trees three years after the exclusion of human activities. The researcher chose to compare regeneration across CFS, ADS and DGS because farming and livestock grazing are the major factors for disturbing tropical forests (Kimaro and Lulandala 2013; Keenan et al. 2015).

3.8.1. Sampling Procedures

3.8.2. Sampling Design

A systematic sampling design was used in this study. For the sake of covering representative sample of forested blocks and disturbed land in UFR, stratification approach was adopted Tomppo et al. 2014, URT, 2015). A comparison between floristic compositions of forests under different management systems was carried out. For the purpose of comparing impacts of disturbance on vegetation, random plots were established in the three major land uses i.e. CFS, ADS and DGS.

3.8.4. Sampling Intensity, Size and Shape of Plots

In order to determine the status of disturbance signs, forest disturbance, soil and vegetation attributes were measured and assessed. On each of the three strata (CFS, ADS and DGS), the 25m x 25m field plots were established and from which samples were drawn (Tomppo et al. 2014, URT, 2015). Ground forest inventories were carried out (Axelsson et al. 2012) by measuring and identifying tree species from CFS, ADS and DGS. Because human activities are not uniformly distributed, random selections of sites for plot establishment were adopted in this study. Trees and composition parameters were recorded from each land use. About 47 quadrats of 25m x 25m size were laid down for collection of adult tree data, while nested plots of 2m x 2m (Shankar 2001; Bharathi and Prasad 2015) (i.e. within the established 25m x 25m plots) were laid for collection of seedlings and saplings. Stems with a diameter of ≥ 20cm at breast height (DBH) (approximately 1.34m height above the ground) were counted as trees. All tree species with <20 cm girth were considered as regenerates in the following subdivisions: (i) seedlings included only trees with <0. 40m height and (ii) saplings included all trees from ≥0.40m to <1m heights as adapted from (Bharathi and Prasad 2015).

Trees were identified in the field by using field guide books with the help of a local and qualified botanists, while some voucher samples were taken to Sokoine University of Agriculture botanic laboratory for further identification using Flora of Tanzania and field guide book by Lovett, et al. (2006). All dbh were recorded at 1.3 m above the ground and as well as height (h) for each poles and trees at 1% sampling intensity.

3.8.5. Vegetation Data Analysis

Geographical information system (GIS) analysis was carried out by using Arc View 1.3 and ERDAS imagine software, version 8.3.1 programs to understand the current status of forest vegetation types under different land uses. Ground verification was carried out to collect biophysical data and modified land covers described in the preliminary stages of image interpretation. The Microsoft Excel windows 10 and Statistical Package for Social Sciences Software (SPSS) were used to analyze the inventory data. From each land use, live vegetation results were used to determine forest structure in relationships to human disturbances. Before the computation of forest stand parameters, a checklist of tree and shrub species were prepared. From the checklist, the following forest stand structural parameters/indicators were calculated per each land use, which were converted into unit hectares (ha). :- (i) number of live trees per unit area (N/ha), (ii) basal area of live trees (m^2 /ha), and (iii) volume of live tree per unit area (m^3 /ha) (Hitimana et al. 2004).

Basal area was used to measure the area occupied by the living tree stems. Computation of basal area was carried out as follows: - (i) Basal (BA) = $((dbh)^{2}*\pi)/4$; where dbh = diameter at breast height. The volume was calculated by the formula (V) = ghf; where V = volume estimation (m^3/ha), g = basal area of the tree (m^2/ha), h=height of the tree (m) and f = form factor (0.5). The form factor of 0.5 was used as an average for natural forest form factor that range between 0.4 and 0.6 (Phillip 1983). Forest structure indicators were compared between and across different lands uses. The t-test was applied to compare the existing differences between and across the non-disturbed and disturbed forest ecosystems (Joyi et al. 2015). The results were considered significant at $p <. 050$.

3.8.6. Analysis of Biodiversity Indices

Biodiversity indices were analyzed and used to measure disturbances. In this case, species diversity indices were used because they are the most popular methods for measurement and quantification of plant species (Kikvidze and Ohsawa 2002; FAO 2009, Axelsson et al. 2012; Merganic et al. 2012; Kijazi et al. 2014). Therefore, the following four major species diversity indices were computed in this study: - (i) Shannon diversity index, (ii) Simpson diversity index, (iii) species evenness and (iv) the importance value index (IVI).

3.8.6.1. The Shannon Diversity Index

Shannon-wiener index of diversity (H') were used to determine vegetation species diversity. This index is the most widely used for diversity as it combines species richness and evenness and it is not affected by sample size. The Shannon diversity index was computed as ($H' = \Sigma Pi*\ln Pi$); where H' is the index of diversity and Pi is a decimal fraction of a relative basal area. The interpretation of results from this index were judged on the criterion that Shannon index increases with the number of species in the community but in practice, for biological communities H' does not exceed 5.0 units.

3.8.6.2. Index of Dominance/ Simpson's Index of Diversity

The index of dominance sometimes is called the Simpson's index of diversity is a measure of the distribution of individuals among the species in a community. Simpson's diversity index was computed as $D = \sum (ni/N)^2$; where D = the index of dominance, ni = the number of individuals of species 'i' in the sample, N = the total number of individuals (all species) in the sample and $\sum$= the summation symbol. Determination of values for comparison between land uses was based on the criteria that the greater the value of dominance index, the lower is the species diversity in the community and the vice versa.

3.8.6.3. Species Evenness

Species evenness refers to how close in numbers each species in an environment are. Species evenness is defined as a measure of biodiversity which quantifies how equal the community is numerically. Species evenness was computed by using Shannon's equitability index (H'E), which was calculated mathematically as $H'E = H'/ H_{max}$; where Hmax is defined as ln S (specie richness), H'E value ranges from 0 to 1, in which 1 indicates complete evenness. Its interpretation decision criterion was that the less variation in communities between the species, the higher the H'E is.

3.8.6.4. Importance Value Index (IVI)

Importance value for species (IVI) j was computed from the sum of the relative frequency relative density and relative dominance for the species i.e. IVj=Frj + Drj + Brj. In order to compute all the parameters in this equation, the following series of calculations were carried out.

i. Calculation of the total distance (dt)

$$dt = \sum_{i=1}^{n} di \ldots\ldots\ldots\ldots\ldots Meters$$

Where dt is the total distance, di is the distance to tree number i, and n is the total number of trees

ii. Calculation of average distance between trees,

$$d = dt \div n \ldots\ldots\ldots\ldots\ldots meters$$

Where d= distance between tress, dt = dt is the total distance across all trees

iii. Calculation of average area occupied per tree, A:

$$A = d^2 \ldots\ldots\ldots\ldots meters^2$$

Where A = Area in meters, d = average distance

iv. Calculation of absolute density for all trees, Da, in trees per hectare (ha):

$$Da = 10^4 m^2 \div A \ldots\ldots\ldots\ldots\ tree/ha$$

Where Da is the absolute density for trees

v. Calculation of absolute frequency for each tree species

$$Faj = Mj \div m$$

Where Mj is the number of points where species j occurred, and m is the total number of points.

vi. Calculation of relative frequency of species j, Frj, is the absolute frequency of species j divided by the sum of the absolute frequencies for all species

$$Frj = Faj \div \sum_{k=1}^{p} Fak \times 100\%$$

Where the denominator is the sum of the absolute frequencies for all species, k is the species number, and p is the total number of species

vii. Calculation of relative density of species j, Drj, is the number of occurrences of species j divided by the total number of trees:

$$Drj = Nj \div n \times 100\%$$

Where Nj is the number of occurrences of species j and n is the total number of trees.

viii. Calculation of absolute density of species j

$$Daj = Drj \times Da$$

Where Daj is the absolute density for all trees, Drj is the number of occurrences of species j divided by the total number of trees, Da is the absolute density for trees

ix. Calculation of absolute dominance for species j, Baj, is the mean basal area for species j times the absolute density of species j:

$$Baj = Baj \times Daj$$

Where k is an individual of species j, and t is the number of occurrences of species j.

x. Calculation of relative dominance of species j, Brj, is the absolute dominance of species j divided by the sum of dominance for all species:

$$Brj = \frac{Baj}{\sum_{l=1}^{p} Bai} \times 100\%$$

Where the denominator is the sum of the absolute dominance for all species, and p is the total number of species.

3.9. Canonical Multivariate Data Analysis

The tree and soil data were subjected into Canoco software following the procedures in (Leps and Smilauer 2003). In this work, detrended canonical correspondence analysis

(DCCA) were used to obtain multiple linear regressions and optimal linear combination between tree parameters and soil variables. The computation of these variables in the DCCA facilitated the possibility to test the null models by Monte-Carlo permutation on each set of data. This method was chosen because it permitted the whole community composition data to be carried out and produced the results that are much more informative about species and environmental variables reaction (Legendre 2008; Cankaya, Balkaya and Karaagac 2010). The F-ratio were used to test the significance of correlation at 5% confidence interval.

3.10. Computing LCLU and ESV

The process of quantifying and analyzing ecosystem services value followed a methodological flow chat as in (Figure 3-3). This work adopted ecosystem service evaluation approaches as laid down in Costanza et al. (1997); Zhao et al. (2004); Li et al. (2007); Hu et al. (2008) and Temesgen et al. (2018). Unlike in the previously applied methodologies, in this investigation the data of each LCLU category for 2000 and 2010 were from the Globe Land 30 mapping products at 30-meter spatial resolution developed by National Geomatics Center of China (NGCC 2014), while 2016 images produced from Landsat 8. The Globe Land 30 mapping products provided LCLU data with higher resolution compared to earlier sources of data that taken at 1km and 300m resolution (Han et al. 2015; Zhang et al. 2015a).

Therefore, these sources of data helped to overcome the empirical inherent problem in producing ecosystem services evaluation in Costanza et al. (1997). The classification of images done from Landsat TM/ETM+ data, covering the reference years 2000, 2010 and 2016. These data complemented with MODIS NDVI time series data and Chinese HJ imagery (Han et al. 2015). The NDVI used in LCLU classification, together with Support Vector Machine (SVM) classifier, to indicate characteristics of vegetation (Zhang et al. 2015a). The 2000, 2010 and 2016 LCLU data mapped in ArcGIS, overlays and changes established based on LCLU's total area.

3.10.1. Land Use Classification

The classification of LCLU along the coast produced seven land categories, as representing the estimated areas from S37 00, S37 50 and S37 50, S37 10 geographical coordinates. The seven LCLU categories included (i) farm land (crop land used for agriculture, horticulture and gardens including paddy fields, irrigated and dry farmland, vegetation and fruit gardens); (ii) forest (land covered with trees and vegetation cover over 30%); (iii) grazing land (grassland/rangeland/ land covered by natural grass with cover over 10%); (iv) wetland, (land covered with wetland plants and waterbody including inland marsh, lake marsh, river flood plain wetland, forest/shrub wetland, peat bogs, mangroves, fish and salt marshes; (v) shrub land (land covered with woody perennial plants ranging between >0.5m and <5m); (vi) waterbody (waterbodies in the area including rivers, lakes, reservoirs and fish ponds; and (vii) artificial surfaces (land modified by human activities for settlements, industrial and mining areas, transportation and urban zones) (IPCC 2003; Maitima et al. 2009). Area changes detected based on differences between imagery identification of the changed areas. Each changed category of LCLU was obtained from remotely sensed imagery acquired in years 2000, 2010 and 2016 (Chen et al. 2013a; Kindu et al. 2016). The LCLU information used to compare land changes, socioeconomic activities, and the dynamics of ecosystem services in the coastal zone (Chen et al. 2013b).

3.10.2. Computing the Ecosystem Service Values

The ecosystem service values (ESV) for each of the seven LCLU categories computed. The most representative biomes for each category used as the proxy for a particular land category (Costanza et al. 1997; Temesgen et al. 2018) (Table 7-1). Nevertheless, this assessment adopted ecosystem service coefficients as modified and used in Temesgen et al. (2018). These coefficients used under the assumption that they represented standardized coefficients' values from tropical areas, mainly Sub-Sahara Africa. Indeed, these values used because suitably developed for computing ESV from low-income countries, Tanzania inclusive (Van der Ploeg et al. 2010; Kindu et al. 2016; Temesgen et al. 2018).

However, in this work, the researcher did not identify or gauge factors, which affect local community ecosystems services utilization. The researcher constantly assigned the coefficients values across the community for all ecosystem settings under assumption that, other factors (e.g. access, priorities and availability of ecosystems services) kept constant. This meant that the randomly sampled communities have equal access etc. to ecosystems services. Therefore, land area (ha), confidents and monetary (US$)) methods were used to evaluate the trend of ecosystem service values (Kubiszewski et al. 2017; Costanza et al. 2014).

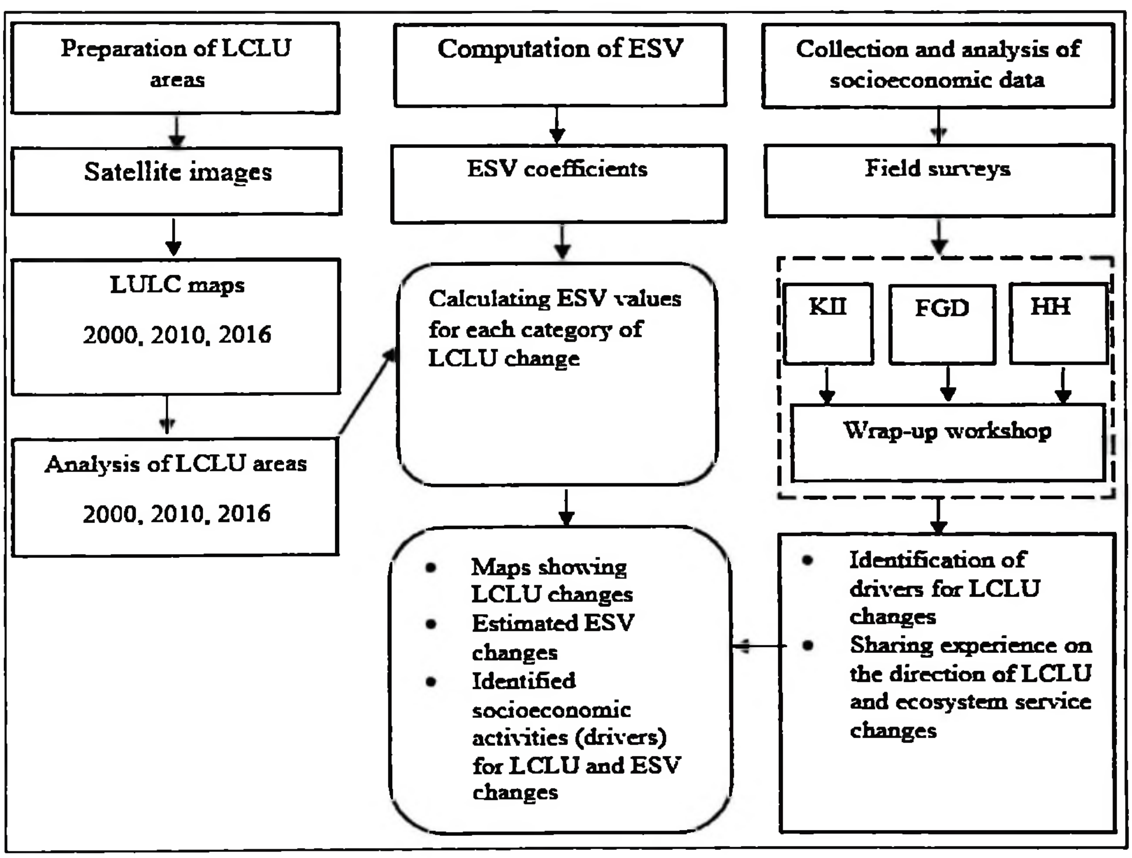


Figure 3- 3: Methodological chat flow for computing ESV

3.10.3. Quantifying LCLU Changes

Land cover and land use dynamics for each land use category computed to map the quantity change across 2000, 2010 to 2016. The change for each LCLU categories were compared quantitatively to map the difference across sixteen years of changes. The rate of change were evaluated as in equation 1 (Tang, Shi and Bi 2014; Yirsaw et al. 2016; Temesgen et al. 2018).

$$K = \frac{Ub - Ua}{Ua} \times \frac{1}{T} \times 100 \qquad \text{(Eq.1)}$$

Where K is the single land cover dynamic index; Ua and Ub are the areas of a certain LCLU class at time "a" and time "b" respectively; T is the time span from time a to time b. When T is in a unit of year, then K is the annual rate of change in area for this land cover type. The values of K range from negative one to one. When $K<0$ it means that the land cover type is in a state of depletion (Tang, Shi and Bi 2014; Yirsaw et al. 2016). The interpretation criterion is that the larger the absolute values of K, the more intensively land depleted. If the values of $K \geqq 0$, it means that, such land category not intensively depleted.

3.10.4. Assessment of Ecosystem Service Values

The total value of the ecosystem service represented by each of LCLU categories obtained by multiplying the estimated size of each land category by the value coefficient of the biome used as the proxy for that category as in equation 2.

$$ESV = \sum(Ak \times Vck) \qquad \text{(Eq.2)}$$

Where ESV is the evaluated ecosystem service values, Ak is the area and Vck is the value coefficient ($/ha/yr) for land use category k (Li et al. 2007; Kindu et al. 2016). The change in ecosystem service values evaluated by calculating the difference between the values for each land cover category in 2000, 2010 and 2016. The percentage ESV changes calculated as in equation 3.

$$\text{Percentage ESV change} = \left(\frac{ESVfy - ESViy}{ESViy}\right) \times 100 \qquad \text{(Eq.3)}$$

Where by ESV = total estimated ecosystem service value, $ESVfy$ is the ESV in the final year, $ESViy$ is the ESV in the initial year. Positive values suggest an increase while negative values show a decrease in ESV (Li et al. 2007; Kindu et al. 2016).

Table 3- 1: Description of the representative biomes with their respective ecosystem service valuation coefficients globally and locally from Costanza et al. (2014) and Temesgen at al. (2018) respectively

LCLU type	Composition	Equivalent biome	Coefficient value (\$/ha/y)	
			Global value	Local value
Forest	Forest land, open forest land	Forest	969.00	1093.20
Grazing land	Moderate coverage grassland and high coverage grassland	Grasslands	232.00	355.50
Shrub land	Grass/rangelands	woody perennial plants, >0.5 m & <5 m	232.00	897.00
Farmland	Paddy field, maize and sesame field	Cropland	92.00	169.20
Wetland	Wetland plants and water bodies	Wetland	14785.00	2856.10
Waterbody	Rivers, land reservoirs fishery, and lakes	Lakes/rivers	8498.00	3226.80
Artificial surfaces	Residential, commercial, Settlement and roads	Urban	0.00	0.00

3.10.5. Human to Ecosystem Services Values

Human beings cause LCLU changes. From LCLU changes, the wellbeing of the human community is affected. The interplay across LCLU change and ecosystem service dynamics used to understand the direction of a human-to-ecosystem service (H-ESV) values. The H-ESV indices for each LCLU category helped to assess the relationship between the human population and the ESV. This index obtained by dividing the ESV to the total population of a given reference year (2000, 2010 and 2016) as in equation 4 (Yirsaw et al. 2016).

$$H - ESV = \frac{TESV}{TP} \qquad \text{(Eq.4)}$$

Where *H-ESV* is the human to ecosystem service values, *TESV* is the total ecosystem service value of each land use, and *TP* is the population of the coastal zone in 2000, 2010 and 2016 years.

3.10.6. Methods for Collecting Socioeconomic Data

Collection of socioeconomic data began with a desk review of the published literature/documents from different sources. The sources included books, articles and reports from the Ministry of Natural Resources and Tourism of the United Republic of Tanzania. Thereafter, a cross section research design employed to collect field data from May 2017 through August 2017. The data collected deliberately from Changalikwa, Kwaluhombo, Kwang'andu, Mbwewe and Mpaji villages, because they are located within the coastal zone of Tanzania.

3.10.6.1. Focus Group Discussions and Field Observations

Prior to intensive households and key informants survey and interviews, focus group discussions (FGD) conducted from each village to learn about local conditions in relationship with land use and economic activities. Ten members formed one heterogeneous group, whereby the composition of each group considered gender, sex, occupation and age differences. Direct field observation and note taking supported FGD during data collection. The purpose of FGD and observations were to collect information for opening up discussions with respondents on LCLU and socioeconomic activities changes.

3.10.6.2. Household Surveys

The households randomly picked from the village register books in which all households' heads listed. In villages where register books were absent, the names of people recorded with the assistance of village leaders from each hamlet and random selection employed to avoid/reduce bias. Simple random sampling technique used to get households listed as farmers (crop producers), livestock keepers, charcoal producers and sellers, house constructors, carpenters, village security and environmental committee members. It took between 30 to 60 minutes to complete one surveys then followed immediate memos production.

3.10.6.3. Key Informants Interviews

The purposive sampling techniques used to interview village and ward executive officers, and district forest, environmental, livestock and agricultural officers. These officers formed a key informant group category. The purpose of key informant interviews and group discussions was to deepen and clarify the understanding of the factors contributing to coastal zone disturbances and socioeconomic activities trends across sixteen years. The in-depth interviews took between 60 to 120 minutes including time for memos production.

3.10.6.4. Wrap-up Workshop

To fine-tune the research findings, a one-day triangulation workshop held just after partial data analysis. The aim of the workshop was to present preliminary results to the households, key informants and group discussion representatives before final presentation of the results. In addition, the workshop functioned to minimize the possibilities of generating biased results because the composition of the workshop members considered all the categories, which interviewed in their respective categories.

Therefore, the process of computing ecosystems service values involved biophysical, social and economic assessments (Figure 3.4).

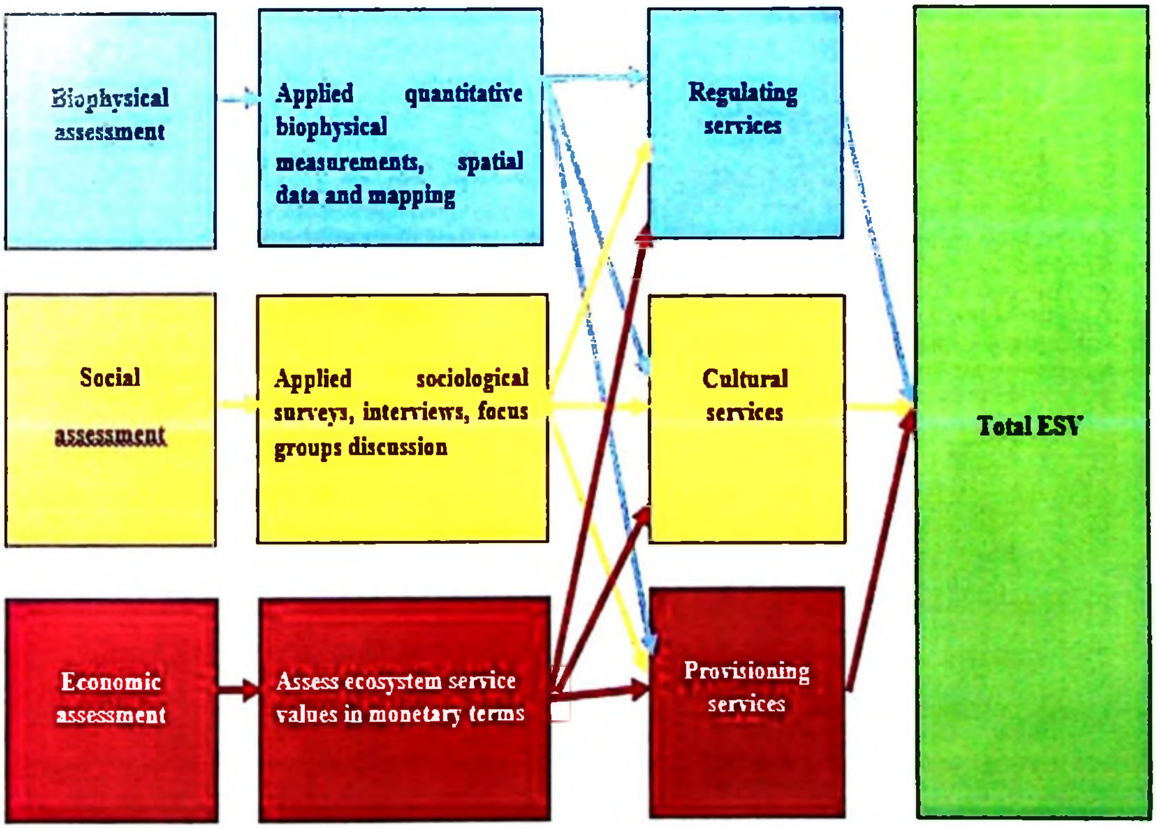


Figure 3- 4: Summarized ecosystems services assessment methods

3.10.6.5. Tools for Data Collection

Socioeconomic data collected by means of household questionnaires and checklist for key informants (Annex 1 and 2). Discussions from spoken interviews recorded using tape recorders and then transcribed to produce memos on the themes of local community understanding about LCLU changes and socioeconomic activities changes over the past sixteen years. The following questions guided the surveys, interviews and discussions. (1) How local community, leaders and experts gauge the trend of LCLU change along the coastal zone of Tanzania. (2) How sets of socioeconomic activities have taken place and what sets have changed over the past sixteen years to affect LCLU along the coastal zone. (3) Why the degradation of coastal ecosystems increased in the past sixteen years. These questions generated both qualitative and qualitative information, which used to complement each other in the discussion of the results. The data subjected into further analysis by using the Statistical Package for Social Sciences (SPSS version 20.0) and Excel (Microsoft, Redmond, WA, USA) computer software.

3.10.7. Collection of Restoration Data

In order to identify and gauge forest restoration interventions, both primary and secondary data gathered in this work. Secondary data obtained through literature survey of the published and government's reports. Primary data were collected from the five villages (Changalikwa, Kwaluhombo, Kwang'andu, Mbwewe and Mpaji) purposely selected and surveyed because are located within the perimeter of this forest. A cross-sectional research design adopted where both qualitative and quantitative approaches used to get data, which collected at a single point in time from selected sample of respondents to represent some large populations. This design adopted for the study because it is economical and served time.

The samples drawn from the households randomly picked from the villages' register books in which all households' heads listed. In villages where register books were absent, the lists of households recorded with the assistance of village leaders from each hamlet and random selection employed to avoid/reduce bias.

Participatory rural appraisal (PRA) employed to assess restoration interventions at local community level. The PRA methods used included focus group discussions and direct field observation. As a research tool, PRA methods adopted for opening up discussions with villagers on restoration of disturbed forests sites in the study area. Focus group discussions purposely chosen to explore information from people of different ages, sex, occupations and residential duration.

Household questionnaires used to collect primary data from respondents (Annex 1). Both open and close-ended questions were included in the data collection protocols. A checklist was prepared to solicit information from key informants (Annex 2). In this study, the key informants comprised individuals who are knowledgeable, accessible and willing to discuss about forest restoration issues. Therefore, a checklist was a tool used to collect data from village leaders, ward executive officers, district and regional forest officers.

3.11. Restoration Data Analysis

Data analyzed by using Statistical Package for Social Sciences (SPSS version 20.0) and Excel (Microsoft, Redmond, WA, USA) computer software. Descriptive statistical analysis then used to explore data for distribution of responses, central tendencies and dispersion. Cross-tabulation and multiple response analyses also performed to ascertain responses and percentages. Two multiple regression models were developed to show the relationship between socioeconomic and environmental factors and artificial /natural trees restoration interventions as dependent variables. The multiple regression equations developed were-

$$Y1 = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + \beta_3 X_3 + \beta_4 X_4 + \beta_5 X_5 + \ldots \beta_n X_n + e$$
$$Y2 = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + \beta_3 X_3 + \beta_4 X_4 + \beta_5 X_5 + \ldots \beta_n X_n + e$$

Where:

Y1 = artificial trees planting intervention

Y2 = natural trees regeneration intervention

X_1 to X_n = independent variables (socioeconomic and environmental factors)

β_0 = a constant showing intercepts for regression equation

β_1 to β_n = independent variables coefficients

e = error term

i = 1, 2, 3...n

n = Sample size (total number of respondents i.e. 255 for the purpose of this study)

The hypotheses tested were:

H_o: $\beta_i = 0$ that is regression coefficients are equal to zero to imply that socioeconomic and environmental factors (independent variables) have no significant contribution to affect artificial trees planting and natural regeneration interventions ($p < .050$).

H_a: $\beta i \neq 0$ that is regression coefficients are not equal to zero to mean that socioeconomic and environmental factors have significant contribution to affect artificial trees planting and natural regeneration interventions ($p < .050$).

The regression models applied to explain the relationship between factors affecting restoration interventions through the amount of artificially planted and naturally managed trees at household level across the study villages. Therefore, the variables included in the regression models were:-

X_1 = Extension services

It hypothesized that respondents who are aware about the importance and restoration interventions would invest in planting and retaining trees to restore the disturbed sites. This could have positive coefficient (+) in the sense that extension services would influence and enhance villagers' willingness and understanding about restoration practices Therefore, extension services were coded with respect to the number (frequency) the respondents were visited by extension or forest officers.

X_2 = Availability of incentives for restoration compensation

The assumption here is that incentives motivate respondents to be engaged in restoration programs. In addition, incentives empower the respondents to have access to alternatives of livelihood activities other than depending on forest resources. Therefore, the study investigated respondent's access to restoration incentives. This variable expected to have a positive regression coefficient (+).

X_3 = Household income

It was hypothesized that respondents with relatively higher income could plant and retain more trees than those with low income. The assumption was that, higher income enable respondents to afford to purchase planting materials, as well as land and pay for tree nursery attendants and would afford to use alternative sources of fuels domestically to safeguard trees. It implies that more trees be planted and well managed by households with higher income. Data on households income were collected and the expected sign of the regression coefficient was positive (+).

X_4 = Availability of tree planting materials (seeds/seedlings)

It hypothesized that if the planting materials were conveniently available within the local settings, respondents would plant them to restore the disturbed sites (given other factors constant). To gauge the availability of planting materials, ranking of materials availability done at household and during group discussions. This variable assigned a positive coefficient (+).

X_5 = Climatic factors

It was hypothesized that under good climatic conditions (mainly precipitation), restoration practices would be successful. Respondents ranked the climatic factors, which affect restoration interventions in the study area. The expected sign for this variable was positive (+).

X_6 = Encroachment by human activities

It hypothesized that human activities such as crop-agriculture and livestock grazing affect the planted trees and naturally regenerating trees. Respondents asked to identify and rank human encroachment activities, which hinder restoration of the disturbed sites. This variable assigned a positive regression impacts coefficient (+).

X_7 = Household size

The hypothesis in this case was that the bigger the household, the more the labor would be available for various activities such as tree planting and caring. It was assumed that high number of trees be planted by households with higher members if other factors were held constant. In defining this variable, household members with the age above 18 years considered as adults, potential participants in tree planting and caring. The expected sign of the regression coefficient was positive (+).

X_8 = Land ownerships (ha)

The underlying hypothesis is that land ownerships determine restoration practices. The bigger the size of the land, the more the space would be available for tree planting, resulting into high number of trees planted by the households. In this case, the expected sign of the regression coefficient was positive (+) implying that there is positive proportionality between land size and number of trees planted or left to regenerate naturally.

X_9 = Residence duration (in years)

The assumption in this variable is that the higher the number of year's respondents lived in a given area, the higher the chances of planting or retaining trees. This assumption is because trees are perennial crops; their tangible benefits obtained after some years of production. Temporally, residential duration in this case affect tree planting. Therefore, the number of years an individual stayed in a given village recorded. The expected sign of the regression coefficient was positive (+).

X_{10} = Donor dependence

Many programs for restoration of forests supported by international or local funding organizations. The assumption put forward here is that some restoration project cease after donor support. Therefore, this work attempted to understand whether tree planting or retaining is donor or locally community influenced. The expected sign of the regression coefficient was positive (+).

X_{11} = Enforcement of laws and regulation

It hypothesized that national and by-laws enforcement and compliance contribute to implementation of restorations practices beyond individual willingness. Therefore, this study tried to understand how laws instituted at local levels to promote tree planting and permitting natural regeneration. The expected sign of the regression coefficient was positive (+) if laws are functionally in place.

The coefficient of determination (R^2) used to show the level of variations in the dependent variables as were explained by independent variables. The higher the value of R^2 the stronger was the model and vice versa.

CHAPTER FOUR: RESULTS

4.1. Electrical Conductivity, Bulk Density and Soil Texture

4.1.1. Electrical Conductivity across Land Uses

The results showed a significant variation across in Electrical conductivity (µS/cm) across the LU. The mean differences (mean ± standard mean error (SEM) of electric conductivity between CFS and ADS was 26.197 ± 8.42, (t = 3.11, $p < .003$); EC in CFS and DGS was 5.55 ± 7.45, (t = 0.75, $p < .460$); EC in ADS and DGS was 20.65 ± 3.97, (t = 5.20, $p < .001$). These results were in the following order EC in CFS> DGS> ADS (Figure 4-1).

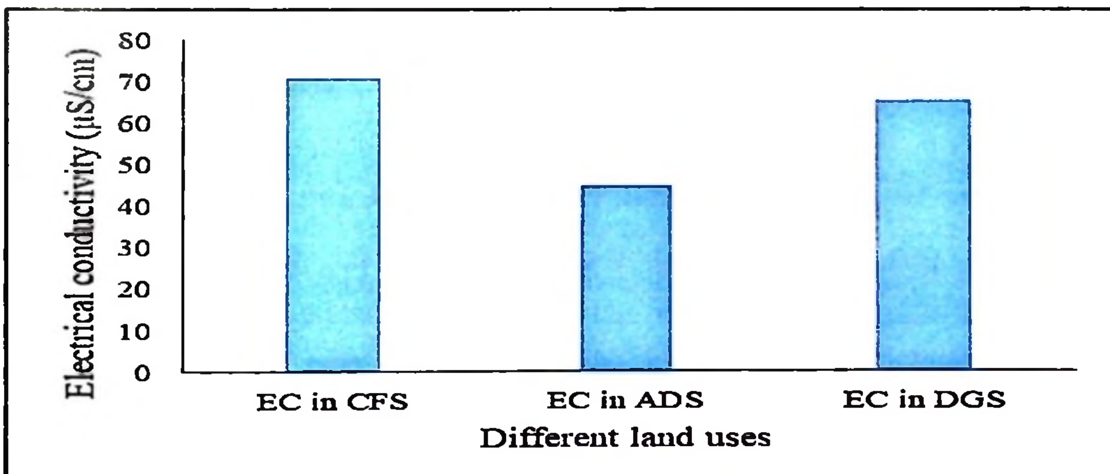


Figure 4- 1: Comparison of electrical conductivity means values across LU

4.1.2. Mean Values of Soil Texture across the Land Uses

There were significant differences between textural classes across the LU (Table 4-1). The percentage mean range were 86.06 to 86.79, 11.40 to 14.98 and 1.81 to 2.57 for sandy, clay and silt respectively. Closed forest sites were dominated by sand, while DGS had large percentage of clay and silt particles. The means (mean ± SEM) differences between EC and each textural classes were significant as follows: EC and clay % in CFS = 59.57 ± 8.92, (t = 6.68, $p < .001$); EC and silt in CFS = 69.17 ± 8.88, (t =7.79, $p < .001$); EC and sand in CFS = 15.81 ± 9.11 (t = 1.74, $p < .089$); EC in CFS and clay in ADS =

59.19 ± 9.09, (t = 6.51, $p < .001$); EC in CFS and silt in ADS = 68.82 ± 8.98, (t = 7.66, $p < .001$); EC in CFS and sand in ADS = 15.09 ± 8.84, (t = 1.71, $p < .095$); EC in CFS and clay in DGS = 56.00 ± 9.05, (t = 6.19, $p < .001$); EC in CFS and silt in DGS = 68.40 ± 8.95,(t = 7.64, $p < .001$); EC in CFS and sand in DGS = 11+8.91, (t = 6.46, $p < .204$); EC in ADS and clay in DGS = 29.80 + 1.02, (t = 29.12, $p < .204$); EC in ADS and silt in DGS = 42.21 + 0.92, (t = 45.68, $p < .001$); EC in ADS and sand in DGS = 37.67 + 0.81, (t = 46.13, $p < .001$).

Table 4- 1: Mean values of soil texture across land uses (N = 47)

	Soil Classes		
LU	Clay	Silt	Sand
CFS	11.40 ± 0.14	1.81 ± 0.15	86.79 ± 0.21
ADS	11.79 ± 0.33	2.15 ± 0.15	86.06 ± 0.39
DGS	14.98 ± 0.30	2.57 ± 0.12	82.45 ± 0.36

4.1.3. The Mean Values of Bulk Density across Land Uses

The means difference of Bd across LU shown in Figure (4-2). The Bd in CFS and ADS was 0.05 ± 0.23, (t = 2.16, $p < .360$); Bd in CFS and DGS was 0.13 ± 0.02, (t = 5.88, $p < .001$); Bd in ADS and DGS was 0.08 ± 0.02, (t = 3.63, $p < .001$). The Bd was in the order of DGS > ADS > CFS.

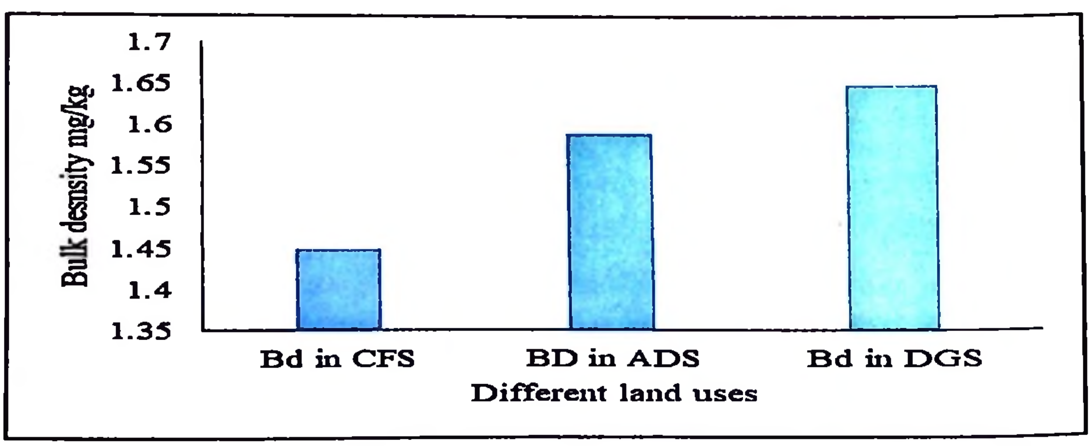


Figure 4- 2: Comparison of bulk density means values across land uses

4.1. 4. Electrical Conductivity versus Soil Texture

The correlation between EC and each soil textural classes was not significant across all the LU. Both positive and negative correlations recorded as follows: - electrical conductivity correlated positively with clay in CFS, ADS but negatively in DGS clay. There was a positive correlation between EC and silt in CFS, ADS and DGS. Electrical conductivity was positively correlated with sand in DGS but negatively correlated in CFS and ADS (Table 4-2).

4.1. 5. Electrical Conductivity and Bd across Land Uses

Although not significant, the results showed a positive correlation between EC across all the LU. The Bd negatively correlated with EC in CFS and ADS, CFS and DGS except in ADS and DGS where it showed a weak positive correlation (Table 2-3).

4.1. 6. Mean Difference of EC and CEC within Land Use

The mean value of EC in CFS was higher than in all other LU (70.98 ± 8.97) followed by EC in DGS (65.43±4.00). The EC in ADS was the lowest compared to that in CFS and DGS (44.78±.88). The mean differences (mean ± SEM) between EC and CEC in CFS = 44.06 ± 8.79, (t = 5.01, $p < .001$); EC and CEC in DGS = 24.77 ± 3.76, (t = 6.57, $p < .001$). EC and CEC in ADS = 20.47 ± 0.80, (t = 25.50, $p < .001$). The CEC mean values were in the order of DGS > CFS > ADS, indicating more depletion of nutrients in ADS than in other two sites. There were positive correlations between EC and CEC within LU and in all the sites (Figure 4-3). The correlation between EC and CEC within land uses was - r = 0.598, $p < .001$, r = 0.581, $p < .001$ and r = 0.345, $p < .018$ in CFS, ADS and DGS respectively.

Table 4- 2: Correlation (r) between electrical conductivity and soil texture (N = 47)

LU	EC and clay		EC and silt		EC and sand	
	r	Sig.	r	Sig.	r	Sig.
CFS	0.349	0.016	0.617	0.001	0.641	0.001
ADS	0.358	0.013	0.090	0.546	0.333	0.022
DGS	0.278	0.059	0.125	0.404	0.187	0.209

Where: sig. = significance level

Table 4- 3: Correlation (r) between EC and Bd across LU (N = 47)

LU	r	Sig.	LU	r	Sig.
EC in CFS and EC in ADS	0.652	0.001	Bd in CFS and Bd in ADS	0.247	0.095
EC in CFS and EC in DGS	0.571	0.001	Bd in CFS and Bd in DGS	0.143	0.339
EC in ADS and EC in DGS	0.154	0.303	Bd in ADS and Bd in DGS	0.353	0.015

Where: sig. = significance level

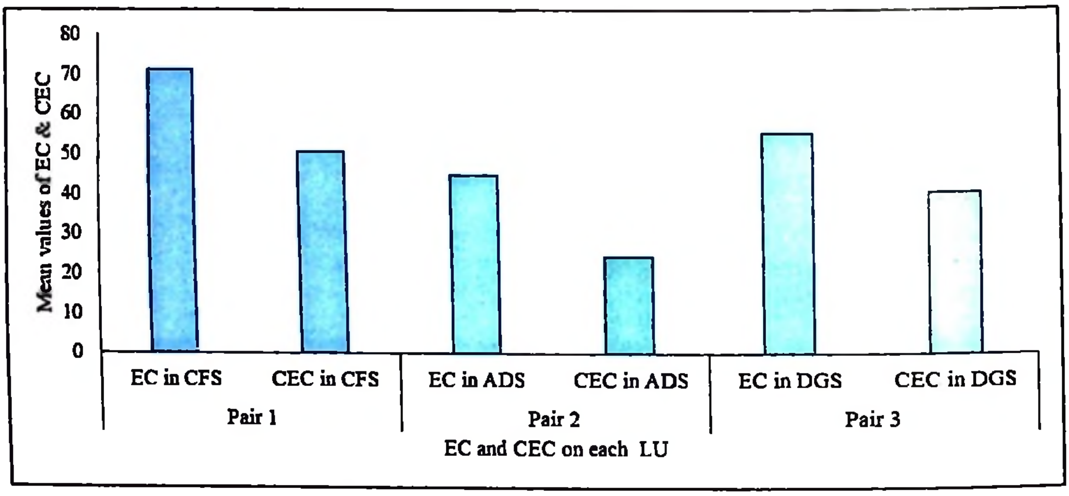


Figure 4- 3: Mean differences between EC and CEC within LU

4.1. 7. Correlation of EC and CEC across Land Uses

The mean differences (mean ± SEM) between EC and CEC across different LU were EC in CFS and CEC in ADS = 46.67 ± 8.89, (t = 5.25, $p < .001$), EC in CFS and CEC in DGS = 30.31 ± 9.24, (t = 3.28, $p < .002$); EC in ADS and CEC in DGS = 4.12 ± 0.10, (t = 1.97, $p < .056$). There was a positive correlation between EC and CEC across the LU. The correlation between EC and CEC in ADS and DGS was weak compared to those recorded in CFS and ADS as in CFS and ADS too (Figure 4-4).

4.1.8. Electrical Conductivity versus Elevation

The mean elevation was 417 across all the sites. The EC was on the average of 70.98 in all sites and at different elevations in CFS, 65.43 in DGS and 44.78 in ADS plots. Regardless of different elevations, CFS had the highest CE than the other LU. The mean difference (mean ± SEM) of EC and elevation was EC and elevation in CFS = 346.38 ± 11.15, (t = 31.08, p < .001); EC and elevation in ADS = 372.58 ± 6.16, (t = 60.50, p < .001); EC and elevation in DGS = 351.93 ± 7.65, (t = 46.02, p < .001). Indeed, the correlation of EC and elevation was negative across all the sites in different elevations.

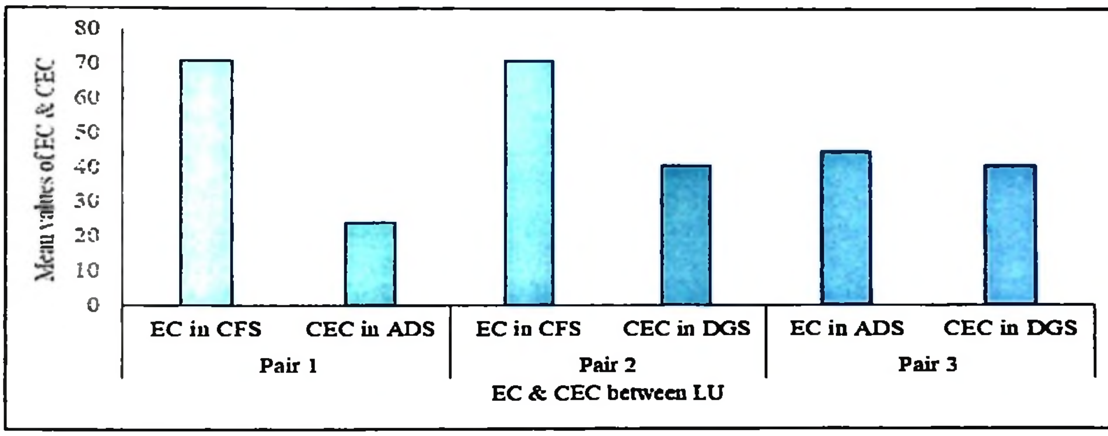


Figure 4- 4 : Mean difference between EC and CEC

4.1.9. Elevation versus Soil Texture across Land Uses

The results showed significant positive and negative correlation between elevation and texture across the three LU. Elevation negatively correlated with clay and silt across all the LU and different elevations. There was a positive correlation between elevation and sand across all the LU (Table 4-4).

Table 4- 4: Correlation (r) between elevation and soil texture across LU (N = 47)

Elevation and ST	r	Sig.	Elevation and ST	r	Sig.
E and clay in CFS	0.087	0.562	E and sand in ADS	0.093	0.535
E and silt in CFS	0.041	0.785	E and in DGS	0.006	0.968
E and sand in CFS	0.083	0.580	El and silt in DGS	0.089	0.553
E and clay in ADS	0.048	0.750	E and sand in DGS	0.035	0.816
E and silt in ADS	0-0.143	0.339			

Where: sig. = significance level

4.1.10. Elevation and Bulk Density

The study area ranged between 300 to 600 masl, and the mean elevation was 417 masl across the study sites. The mean Bd was 1.56 mg/kg in CFS to 1.69 mg/kg in DGS indicating that sites disturbed by livestock grazing across the elevation levels had high Bd followed by ADS. At different elevations, CFS had the lowest Bd. The mean differences (mean ± SEM) between elevation and Bd in CFS was 415.80 ± 6.08, ($t = 86.38$, $p < .001$); elevation and Bd in ADS = 415.75 ± 6.08, ($t = 68.38$, $p < .001$); elevation and Bd in DGS = 415 ± 6.08, ($t = 68.35$, $p < .001$). The elevation negatively correlated with Bd in CFS and DGS and positively correlated in ADS.

4.2. Status of Carbon, Nitrogen and Phosphorus

For comparing the differences between and across closed forests, agriculture and livestock disturbed sites; the following consistence were maintained in the presentation of results. The mean and t-values were kept constant in the order of TN (CFS vs. ADS), TN (CFS vs. DGS), and TN (ADS vs. DGS); TC (CFS vs. ADS), TC (CFS vs. DGS), TC (ADS vs. DGS); P (CFS vs. ADS), P (CFS vs. DGS) and P (ADS vs. DGS) for total nitrogen, carbon and available phosphorus consecutively.

4.2.1. Variation of Total Nitrogen across Land Uses

Total nitrogen variation between CFS vs. ADS was $t = 11.66$, $p < .001$. Due to the means values of TN between CFS and ADS, and the direction of t-value, it can be concluded that there was a significance difference (%) of TN in CFS and ADS from 13.07 ± 0.34 to 11.75 ± 0.25, a difference of 1.32 ± 0.11; TN variation in CFS vs. DGS was: $t = 2.21$, $p < .032$, with a mean difference from 13.07 ± 0.34 to 12.57 ± 0.20, a variation of 0.50 ± 0.23; TN in ADS and DGS showed a variation of $t = 5.34$, $p < .001$, with mean difference from 11.75 ± 0.25 to 12.57 ± 0.20, i.e. 0.82 ± 0.15 TN-variation between ADS and DGS showed that TN in DGS is higher than in ADS.

4.2.2. Variation of Total Carbon across Land Uses

Total carbon differences between CFS vs. ADS was: $t = 11.80$, $p < .001$. Due to the mean values of TC between these two LU and the direction of t-value, a conclusion was drawn that there was significant difference (%) of TC from CFS to ADS (i.e. 14.48 ± 0.23 to 11.81 ± 0.13), a difference of 2.67 ± 0.23; the difference of TC between CFS vs. DGS was: t = 7.66, $p < .001$ with the mean values differing from 14.48 ± 0.23 to 12.24 ± 0.30, a significant difference of 2.24 ± 0.29, and TC in ADS vs. DGS was: t = 2.18, $p < .035$ and the mean difference was 11.81 ± 0.13 to 12.24 ± 0.30, showing a variation of 0.43 ± 0.19, this variation shows that there was less TC in ADS than in DGS.

4.2.3. Variation of Available Phosphorus across Land Uses

The available phosphorus variation between CFS vs. ADS was: t = 24.78, $p <.001$. Due to the means values of phosphorus between CFS vs. ADS, and the direction of the t-value, it was concluded that there was a significant difference of available phosphorus between these two LU from 13.12 ± 6.57 to 11.97 ± 6.96, a variation of 1.15 ± 0.93; variation of available P between CFS vs. DGS was: t = 4.04, $p < .001$, with the mean difference from 13.12 ± 6.57 to 10.12 ± 2.86, a difference of 3.00 ± 1.56 and variation in available phosphorus between ADS vs. DGS was: t = 1.54, $p < .131$, with a mean difference from 11.97 ± 6.96 to 10.12 ± 2.86, a difference of 1.85 ± 01.10.

4.2.4. Carbon-Nitrogen Ratio across Land Uses

Carbon-nitrogen ratio variation between CFS vs. ADS was: t = 3.97, $p < .001$. Due to the mean values of CN ratio between CFS vs. ADS, and the direction of t-value, it was concluded that there is a significant difference of CN ratio between these two LU i.e. from 8.62 ± 2.84 to 9.88 ± 2.91, a difference of 1.26 ± 2.77; variation of CN ratio between CFS vs. DGS was: t = 2.33, $p < 0.02$, with the mean difference from 8.62 ± 2.84 to 6.53 ± 2.06, a difference of 2.09 ± 0.57, and variation in CN ratio between ADS vs. DGS was: t = 2.94, $p < .001$, with a mean difference from 9.88 ± 2.91 to 6.53 ± 2.06, a difference of 3. 35 ± 1.39.

4.2. 5. Variation of TN, TC and P against Elevation

Between 300-390 m elevations, differences in each nutrient (percentage) were TN in ADS > CFS >DGS; TC was in the order of CFS > DGS > ADS while P was in the order of CFS > ADS > DGS. Between 391 to 447 m, TN was in DGS > ADS >CFS; TC was recorded in CFS > DGS > ADS; while P was in CFS > ADS > DGS orderly. At the elevation of 448-500 m, TN was DGS > ADS > CFS; TC order was TFS > ADS > DGS; P was DGS > CFS > ADS. Across the study sites, there were negative correlations between nutrients levels and elevations. This trend indicates that with increase in elevation there is unit loss of nutrients (Figure 3-1: a, b, c, d, e, f, g, h and i).

4.2.6. Correlation (R^2) of TN, TC and P within Land Uses

The correlation (p = .05) between TN, TC and P within LU were:- (i) positive correlation between TN & TC in ADS (R^2 = 0.59); TN & TC in DGS (R^2 = 0.84); (ii) weak positive correlation between TN & TC in CFS (R^2 = 0.18); TN & P in CFS (R^2 = 0.14); TN & P in ADS (R^2 = 0.01); TN & P in DGS (R^2 = 0.12); TC & P in DGS (R^2 = 0.11; (iii) negative correlation between TC & P in CFS (R^2 = 0.07) and TC & P in ADS (R^2 = 0.17).

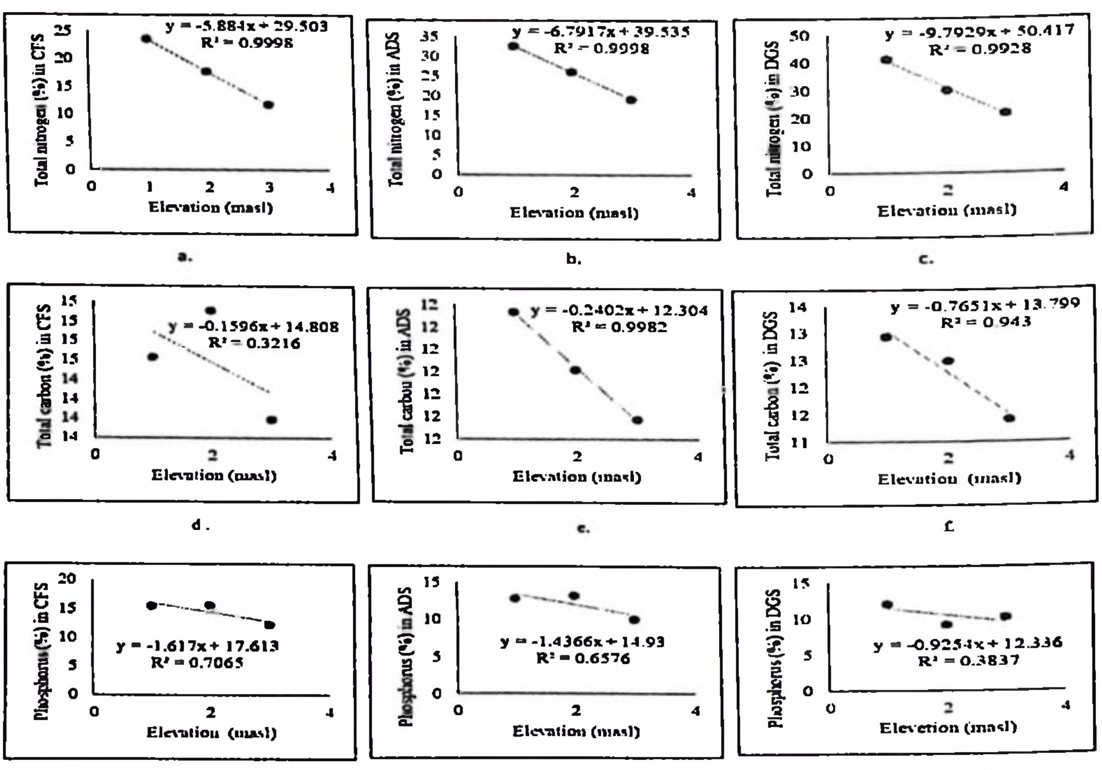


Figure 4- 5: Correlation between nutrients levels (%) and elevation (masl×102) (a = TN in CFS vs. elevation, b = TN in ADS vs. Elevation, c = TN in DGS vs. Elevation, d = TC in CFS vs. Elevation, e = TC in ADS vs. Elevation, f = TC in DGS vs. Elevation, g = P in CFS vs. Elevation, h = P in ADS vs. Elevation and i = P in DGS vs. Elevation.

4.2.7. Correlation of TN, TC and P across Land Use

The correlation (p = .050) between TN, TC and P between LU showed both positive and negative relationships as follows: (i) strong positive correlation between: TN in CFS and ADS ($R^2 = 0.62$), TN in CFS and DGS ($R^2 = 0.96$), TN in ADS and DGS ($R^2 = 0.61$), TC in ADS and DGS ($R^2 = 0.97$), P in CFS and ADS ($R^2 = 0.98$), (ii) weak positive correlation between TC in CFS and DGS ($R^2 = 0.09$), TC in CFS and DGS ($R^2 = 0.15$), (iii) weak negative correlation between P in CFS and ADS ($R^2 = 0.14$) and P in ADS and DGS ($R^2 = 0.18$) (Figure 4-5).

4.3. Soluble Bases Values

In comparing variation across CFS, ADS and DGS, the following consistence maintained in the presentation of results. The means and t-values are kept constant in the order of Ca (CFS vs. ADS), Ca (CFS vs. DGS), and Ca (ADS vs. DGS); Mg (CFS vs. ADS), Mg (CFS vs. DGS) and Mg (ADS vs. DGS); K (CFS vs. ADS), K (CFS vs. DGS), and K (ADS vs. DGS); Na (CFS vs. ADS), Na (CFS vs. DGS), and Na (ADS vs. DGS); CEC (CFS vs. ADS), CEC (CFS vs. DGS), and CEC (ADS vs. DGS) and BS (CFS vs. ADS), BS (CFS vs. DGS), and BS (ADS vs. DGS). In this work, all exchangeable bases are expressed in cmol/Kg/ha, CEC in cmol/Kg/ha while BS values are expressed in percentage volume/ha.

4.3.1. Calcium Variation across CFS, ADS and DGS

Calcium variation between CFS and ADS was t = 3.78, p <.001. Due to the mean (mean ± standard error mean (SE)) values of Ca between CFS and ADS and the direction of the t-value, it was concluded that there was a significant difference of Ca between these two land uses from 14.12 ± 1.07 to 10.37 ± 0.47 with a variation of 3.75 ± 0.99. The variation of Ca on CFS vs. DGS was: t = 2.91, p <.001, with the mean difference from 14.12 ± 1.07 to 11.00 ± 0.22, a difference of 3.11 ± 1.07, the variation in Ca between ADS and DGS was: t =1.10, p = .280, with a mean difference from 10.37 ± 0.47 to 11.00 ± 0.22, and a difference of 0.63 ± 1.10 (Table 4-5).

4.3.2. Magnesium Variation across CFS, ADS and DGS

Magnesium variation between CFS and ADS was t = 4.71, p <.001. Due to the mean values of Mg between CFS and ADS and the direction of the t-value, concluded that there was a significant difference of this element between these two land uses from 2.96 ± 0.06 to 2.16 ± 0.17 with a variation of 0.80 ± 0.17. The variation of Mg in CFS vs. DGS was t = 13.85, p <.001, with the mean difference from 2.96 ± 0.17 to 8.84 ± 0 .44, a difference of 5.87 ± 1.07, and the variation in Mg between ADS and DGS was: t = 17.11, p <.001, with a mean difference from 2.16 ± 0.17 to 8.84 ± 0.44, and a difference of 6.67 ± 0.39 (Table 4-5).

4.3.3. Potassium Variations across CFS, ADS and DGS

Potassium variation between CFS vs. ADS was t = 0.41, p =.860. Due to the mean values of K between CFS and ADS and the direction of the t-value, concluded that there was a significant difference of K between these two land uses from 0.70 ± 0.05 to 0.72 ± 0.05 with a variation of 0.03 ± 0.06. The variation of K between CFS and DGS was: t = 5.88, p <.001, with the mean difference from 0.70 ± 0.05 to 1.24 ± 0.08, a difference of 0.55 ± 0.09, and the variation in K between ADS and DGS was: t = 5.88, p <.001, with a mean difference from 0.72 ± 0.05 to 1.24 ± 0.08, and a difference of 0.52 ± 0.09 (Table 4-5).

4.3.4. Sodium Variation in CFS, ADS and DGS

Sodium variation between CFS and ADS was t = 0.19, p = .240. Due to the mean values of Na between CFS and ADS and the direction of the t-value, concluded that there was a significant difference of Na between these two land uses from 0.13 ± 0.01 to 0.14 ± 0.01 with a variation of 0.01 ± 0.01. The variation of Na between CFS and DGS was: t = 7.96, p < .001, with the mean difference from 0.13 ± 0.01 to 0.44 ± 0.04, a difference of 0.31 ± 0.01, and the variation in Na between ADS and DGS was: t = 7.37, p < .001, with a mean difference from 0.14 ± 0.01 to 0.44 ± 0.04, and a difference of 0.31 ± 0.04 (Table 4-5)

4.3.5. The CEC Variation across CFS, ADS and DGS

Cation exchange capacity variation between CFS and ADS was t = 3.12, p= .030. Due to the mean values of CEC between CFS and ADS and the direction of the t-value, concluded that there was a significant difference of CEC between these two land uses from 26.92 ± 0.30 to 24.31 ± 0.87 with a variation of 2.61 ± 0.84. The variation of CEC between CFS and DGS was: t = 8.62, p <.001, with the mean difference from 26.92 ± 0.30 to 40.66 ± 1.54, a difference of 13.74 ± 1.59, and the variation in CEC between ADS and DGS was: t = 7.48, p <.001, with a mean difference from 24.31 ± 0 .87 to 40.66 ± 1.54, and a difference of 16.36 ± 2.19 (Table 4- 6).

4.3.6. Base Saturation Variation across CFS, ADS and DGS

Base saturation variation between CFS and ADS was t = 2.75, p <.001. Due to the mean values of BS between CFS and ADS and the direction of the t-value, concluded that there was a significant difference of BS between these two land uses from 49.95 ± 1.93 to 60.24 ± 4.41 with a variation of 10.29 ± 3.74. The variation of BS between CFS and DGS was: t = 2.19, p =.030, with the mean difference from 49.95 ± 1.93 to 55.82 ± 2.01, a difference of 5.86 ± 2.67, and variation in BS between ADS and DGS was: t = 0.84, p = .400, with a mean difference from 60.24 ± 4.41 to 55.82 ± 2.01, and a difference of 4.42 ± 5.26 (Table 4- 6)

Table 4- 5: Soluble bases variation across land uses

	Ca		Mg		K		Na	
LU	mean	p	mean	p	mean	p	mean	p
CFS vs. ADS	3.75 ± 0.99	<.001	0.80 ± 0.17	<.001	0.03 ± 0.06	<.680	0.01 ± 0.01	<.240
CFS vs. DGS	3.11 ± 1.07	<.001	5.87 ± 0.42	<.001	0.55 ± 0.09	<.001	0.31 ± 0.04	<.001
ADS vs. DGS	0.63 ± 0.58	<.280	6.67 ± 0.39	<.001	0.52 ± 0.09	<.001	0.31 ± 0.04	<.001

Where: p = p-value

Table 4- 6: The variation of CEC and BS across CFS, ADS and DGS

	CEC		BS	
Land use	mean	p-value	Mean	p-value
CFS vs. ADS	2.61 ± 0.84	< .030	10.29± 3.74	< .010
CFS vs. DGS	13.74 ± 1.59	< .001	5.86 ± 2.67	< .030
ADS vs. DGS	16.36 ± 2.19	< .001	36.03± 5.26	< .400

4.3.7. Correlations of Soluble Bases between Land Uses

There was a statically positive correlation between; Ca in CFS and ADS (r = 0.139); Mg in CFS and DGS (r = 0.153); Mg in ADS and DGS (r = 0.168); K in CFS and ADS (0.062); Na in CFS and DGS (r = 0.004); Na in DGS and CFS (r = 0.179); CEC in CFS and ADS (r = 0.290); BS in CFS and ADS (r = 0.007). There was statistically weak positive correlation between Ca in CFS and DGS (r = 0.005); Mg in CFS and ADS (r = 0.018); K in ADS and DGS (r = 0.004); Na in ADS and DGS (r = 0.078). There was a negative correlation between Ca in ADS and DGS (r = 0.083); K in CFS and DGS (r = 0.006); Na in CFS and ADS (r = 0.005); CEC in ADS and DGS (r = 0.375). There was strong negative correlation between CEC in CFS and DGS (r = 0.006); BS in ADS and DGS (r = 0.054). Table (4-7) shows a correlation summary of soluble bases and Table (4-8) indicates the correlation of CEC and BS between land uses.

Table 4- 7 : Paired soluble bases correlation between across land uses

LU	Ca		Mg		K		Na	
	r	*p*-value	r	*p*	r	*p*	r	*P*
CFS vs. ADS	0.373	<.010	0.135	<.365	0.247	<.094	0.042	<.780
CFS vs. DGS	0.074	<.623	0.320	<.028	0.074	<.622	0.421	<.003
ADS vs. DGS	0.288	<.050	0.463	<.001	0.051	<.734	0.075	<.616

Where: r = Correlation value, *p* = p-value

Table 4- 8: Paired sample correlation of CEC and BS across land uses

LU	CEC		BS	
	r	*p*	r	*P*
CFS vs. ADS	0.279	<.058	0.538	<.000
CFS vs. DGS	0.079	<.596	0.082	<.584
ADS vs. DGS	0.613	<.000	0.263	<.001

Where: r = Correlation value, *p* = p-value

4.3.8. Correlation of Soluble Bases with Elevation Levels

The correlation was positive between base elements and elevation (E) as follows: Ca vs. E in CFS, Mg vs. E in CFS, Na vs. E in CFS, CEC vs. E in DGS and BS vs, E in CFS. The correlation was negative in the following patterns: Ca vs. E in ADS, Ca vs. E in DGS, Mg vs. E in ADS, Mg vs. E in DGS, K vs. E in CFS K vs. E in ADS, Na vs. E in ADS, Na vs. E in DGS, CEC vs. E in CFS, CEC vs. E in ADS, BS vs. E in ADS and BS vs. E in DGS (Table 4-9)

Table 4- 9: Correlation of Ca, Mg, K, Na, CEC and BS with elevation

LU and Elevation	Ca		Mg		K		Na		CEC		BS	
	r	p	r	p	r	p	r	p	R	p	r	p
Elevation and CFS	1	0.250	0.04	0.794	0.09	0.539	0.14	0.365	0.08	0.615	0.05	0.750
Elevation and ADS	0.18	0.222	0.25	0.095	0.02	0.987	0.08	0.589	0.15	0.329	0.01	0.972
Elevation and DGS	0.04	0.775	0.03	0.863	0.02	0.890	0.03	0.851	0.05	0.727	0.12	0.418

Where: r = Correlation value, *p* = p-value

4.4. Analysis of Tree Species

4.4.1. Species Density

Seventy species belonging to 25 families sampled from CFS, ADS and DGS. These families were grouped into five major categories based on their percentage contribution to the main sample: (i) Fabaceae (26%), (ii) Combretaceae (7%), (iii) Malvaceae and Meliaceae (4%), (iv) Anacardiaceae, Annonaceae, Apocynaceae, Bignoniaceae, Capparidaceae, Moraceae, Phyllanthaceae and Sapotaceae (3%) and (v) Asteraceae Araliaceae, Balanitaceae, Bombacaaceae, Clusiaceae, Dichapetalaceae, Ebenaceae, Euphorbiaceae, Loganiaceae, Rubiaceae, Rutaceae, Santalaceae, and Verbenaceae (1% of each family). Results show that the average population density (N/ha) of seedlings was in 10, 45 and 85 species for CFS, DGS and ADS accordingly.

Saplings density ranged from 15, 20 and 65 for CFS, ADS and DGS respectively. Adult trees density was 130, 65 and 30 in that order for CFS, ADS and DGS (Figure 4-6).

Seedlings and saplings had sizeable frequency of *Combretum molle, Cynometra webberi, Dialium holtizii, Ficus sur, Hymenea verrucosa, Khaya anthotheca, Millicia excelsa, Millettia stuhlmannii, Sclerocarya birrea* and *Stercularia appendiculata*. A large component of adult trees density consisted by *Acacia brevispica, C. webberi, C. molle, H. verrucosa, D. holtizii, M. stuhlmannii, Terminalia sambesiaca* and *Vitex zanzibarensis*. In ADS and DGS, the density of adult trees was largely composed of a few remnant trees such as *A. brevispica, Acacia polyacantha, Acacia senegalensis, Afzelia quanzensis, Cassia abbreviata, Combretum collinum, Cussonia spicata, Erythrina abyssinica, Sterculia abbreviate* and *Strychnos henningsii*.

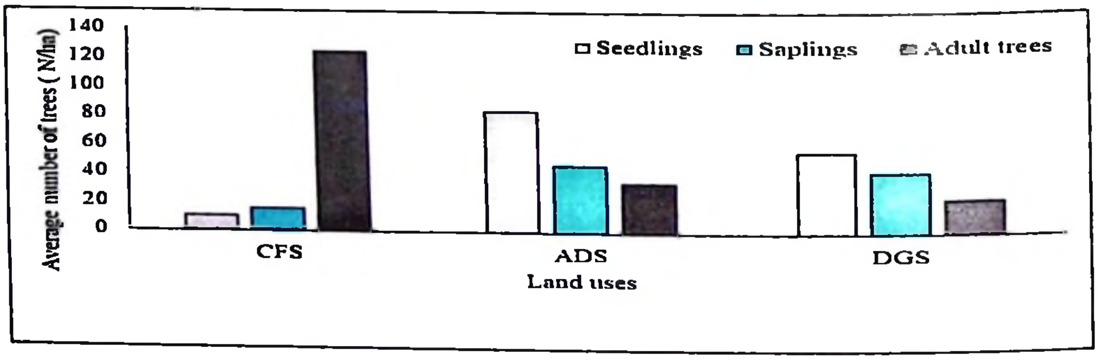


Figure 4- 6: Number of tree sub-categories per hectare in each land use

4.4.2. Diameters of Tree across Land Uses

Tree sub-categories diameters (seedlings, saplings and adults) differed across LU sites (Figure 3-7). In CFS, seedling diameters ranged from 4 to 10cm, saplings had a range from 8 to 12cm, while adult trees had 13 to 98cm. In ADS and DGS seedlings and saplings had similar average diameters average and trends that ranged from 3 to 6cm and 7 to 12cm respectively, while trees had dbh between 25 to 31cm. Tree species, that had large dbh included *Combretum mole, Cynometra webberi, Dialium holtizii, Ficus stuhlmannii, Hymenea verrucosa, Khaya anthotheca, Millicia excelsa, Pteleopsis myrtifolia, Sclerocarya birrea, Tamarindus indica* and *Terminalia sambesiaca*. The following trees had small dbh across all the LU sites: *Ficus sur, K. anthotheca, S. birrea, M. excelsa* and *Millettia stuhlmannii*. Small dbh dominated the seedlings and saplings sub-categories of tree populations across LU sites.

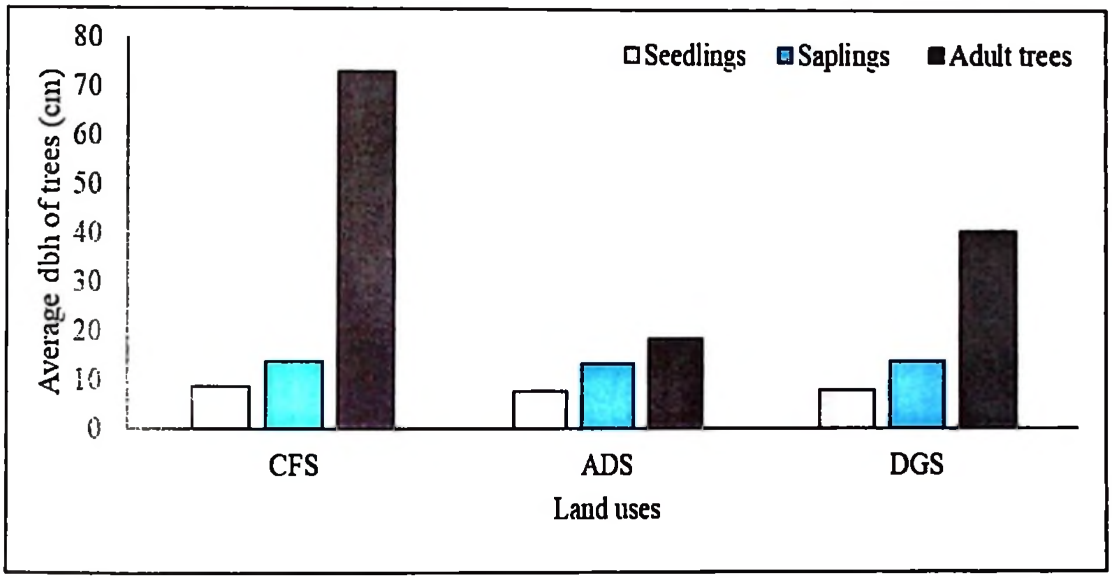


Figure 4- 7: Average dbh of trees sub-categories in each land use

4.4.3. Basal Area of Trees across Land Uses

Basal area (m^2/ha) differed across LU sites for all tree sub-categories. Seedlings had a BA of 0.20 to 1.30, 0.30 to 0.69 and 0.2 to 0.28 in CFS, ADS and DGS respectively; saplings BA ranged from 0.2 to 1.47; 0.5 to 1.13 and 0.20 to 0.64 in ADS, CFS and DGS while adult trees BA was 1.82 to 73.50, 1.13 to 24.62 and 0.50 to 7.96 for CFS, DGS and ADS correspondingly. There was significant mean paired differences in BA between and across LU sites (Table 5-1). At $p \leq .05$, the t-values for seedlings paired mean (in brackets) were 0.83 (CFS and ADS), 13.08 (CFS and DGS), 11.68 (ADS and DGS); for saplings they were 0.83 (CFS and ADS), 13.08 (CFS and DGS), 11.68 (ADS and DGS) and for adult trees they were 10.24 (CFS and ADS), 9.33 (CFS and DGS) and 3.36 (ADS and DGS).

4.4.5. Volume of Trees Sub-Categories across Land Uses

A small volume (m^3/ha) in seedlings and saplings, was recorded, unlike in adult trees across all LU sites. The range of seedling volume was 0.1 to 0.17, 0.2 to 0.8 and 0.10 to 0.80 for CFS, ADS and DGS in CFS. Saplings volume was 0.27 to 1.03, 0.20 to 1.02 and

0.13 to 0.25 in CFS, DGS, and ADS respectively; adult trees volume ranged from 0.60 to 414 in CFS, 0.61 to 135 in DGS and from 0.60 to 43.78 in ADS.

The mean paired differences showed a significant variation in volume between and across LU sites (Table 4-10). At $p \leq .05$, the t-values for seedlings paired mean (in brackets) were 5.46 (CFS and ADS), 7.51 (CFS and DGS), 3.68 (ADS and DGS); for the saplings values were 0.83 (CFS and ADS), 13.08 (CFS and DGS), 11.50 (ADS and DGS) and for adult trees they were 10.24 (CFS and ADS), 9.33 (CFS and DGS) and 3.36 (ADS and DGS). The order of seedlings volume was CFS > ADS > DGS. The order was similar for saplings, while in adult trees the order was CFS > DGS > ADS.

4.4.6. Shannon-Wiener Index

Seedlings and saplings Shannon-Wiener indices were ADS > CFS > DGS and ADS > CFS > DGS in order. There was a significant variation between ADS saplings compared to a non-significant variation between CFS and DGS. The adult trees Shannon-Wiener index was CFS > DGS > ADS accordingly (Table 4-10). At $p \leq .050$, the t-values for seedlings for each paired mean (in brackets) were 1.35 (CFS and ADS), 3.23 (CFS and DGS), 6.14 (ADS and DGS); for saplings the values were 0.98 (CFS and ADS), 625.97 (CFS and DGS), 7.44 (ADS and DGS) and for adult trees they were 1.18 (CFS and ADS), 1.18 (CFS and DGS) and 5.33 (ADS and DGS).

Table 4- 10: Basal Area, Volume, Shannon, Simpson and Equitability Indices

Land uses	Basal Area			Volume			Shannon Index			Simpson's Index			Equitability		
	Se	Sa	At	Se	Sa	At	Se	Sa	At	Se	Sa	At	Se	Sa	At
CFS	0.51 ± 0.03	0.76 ± 0.03	34.60 ± 2.97	0.80 ± 0.01	0.51 ± 0.02	190.33 ± 16.33	2.93 ± 0.60	1.41 ± 0.00	2.08 ± 0.08	1.49 ± 0.00	1.93 ± 0.00	0.18 ± 0.00	0.35 ± 0.08	0.85 ± 0.00	0.01 ± 0.00
ADS	0.32 ± 0.02	0.72 ± 0.03	3.73 ± 0.21	0.05 ± 0.01	0.48 ± 0.02	20.54 ± 1.16	2.94 ± 0.09	1.98 ± 0.91	0.72 ± 0.01	1.49 ± 0.20	1.93 ± 0.00	1.96 ± 0.00	0.15 ± 0.01	0.71 ± 0.00	0.16 ± 0.00
DGS	0.21 ± 0.02	0.31 ± 0.01	5.72 ± 0.57	0.03 ± 0.02	0.21 ± 0.01	31.49 ± 3.15	2.93 ± 0.60	1.41 ± 0.62	0.88 ± 0.00	1.40 ± 0.00	1.92 ± 0.00	1.96 ± 0.00	0.13 ± 0.01	0.94 ± 0.01	0.15 ± 0.00

Where: Se = seedlings, Sa = saplings, At = adult trees

4.4.7. The Simpson's Index across Land Uses

The Simpson's index in CFS and ADS for seedlings, and saplings had similar diversity values, but were higher than those in the DGS. This shows that there was a higher diversity of seedlings as well as for saplings in DGS than in CFS and ADS. However, adult trees diversity was lower in CFS than in any other LU site. The variations in paired mean differences between and across LU sites shown in (Table 4-10). At $p \leq .05$, the t-values for seedlings for each paired mean (in brackets) were 20.37 (CFS and ADS), 74.05 (CFS and DGS), 60.36 (ADS and DGS); in saplings the values were 38.31 (CFS and ADS), 242.92 (CFS and DGS), 12.39 (ADS and DGS) and for adult trees they were 18.16 (CFS and ADS), 160.71 (CFS and DGS) and 0.07 (ADS and DGS).

4.4.8. Shannon Equitability

The Shannon equitability shows that seedlings in ADS and DGS had lower values than CFS, while saplings equitability showed that there was a high value in CFS, followed by DGS and then ADS, in that order. The mean equitability value was lower in CFS adult trees when compared to ADS and DGS. These observations show that there was a lower frequency of seedlings in CFS than in ADS and DGS, while there was a lower saplings frequency in DGS and CFS than in ADS. Therefore, adult trees' frequency was higher in CFS than in ADS and DGS. The paired mean variation across LU sites as shown in (Table 4- 10). At $p \leq .050$, the t-values for seedlings paired mean (in brackets) were 137.75 (CFS and ADS), 104.91 (CFS and DGS) and 208.27 (ADS and DGS); for saplings the values were 0.00 (CFS and ADS), 0.81 (CFS and DGS) and 0.52 (ADS and DGS), and for adult trees they were 1.18 (CFS and ADS), 0.72 (CFS and DGS) and 1.13 (ADS and DGS).

4.4.9. Important Value Index across Land Uses

Seedlings in DGS scored the highest IVI, followed by those in CFS and ADS. The mean value of saplings IVI was highest in CFS followed by ADS and DGS. Adult trees had highest IVI in CFS followed by ADS and DGS. The trend of IVI shows that there was a high dominance of adult trees in CFS, while ADS and DGS had higher regeneration values of seedlings and saplings.

Abutilon mauritianum, Acacia senegalensis, Albizia petersiana, Combretum collinum, Dalbergia nitidula, Holarrhena pubescens, Julbernardia globiflora, Millettia stuhlmanii, Ormocarpum kirkii, Sclerocarya birrea, Tamarindus indica, Uvaria acuminata and *Xeroderris stuhlmannii* dominated seedlings and saplings components while *A. senegalensis, Brachystegia boehmii, H. pubescens, Khaya anthotheca, S. birrea, T. indica* dominated the adult trees IVI component in CFS.

Species that had large IVI in seedlings and saplings were: *A. mauritianum, A. petersiana, Afzelia quanzensis, Brachytergia microphylla, Combretum schumannii, Cynometra webberi, Erythrina abyssinica, H. pubescens, J. globiflora, O. kirkii, S. birrea, T. indica* and *X. stuhlmannii,* while adult trees were dominated by a few species such as *A. petersiana, A. quanzensis, C. schumannii, J. globiflora, Pterocarpus rotundifolius, Sorindeia madagascariensis* and *X. stuhlmannii.*

Seedlings and saplings IVI in DGS largely represented by *A. mauritianum, Acacia polyacantha, A. quanzensis, B. microphylla, P. rotundifolius, T. indica,* while *A. quanzensis, B. microphylla, Hymenea verrucosa,* and *P. rotundifolius* had a greater IVI than other adult trees. The mean values of tree species dominated the IVI across the three land uses were as in (Figure 4-8, 4-9 and 4-10) for each land uses.

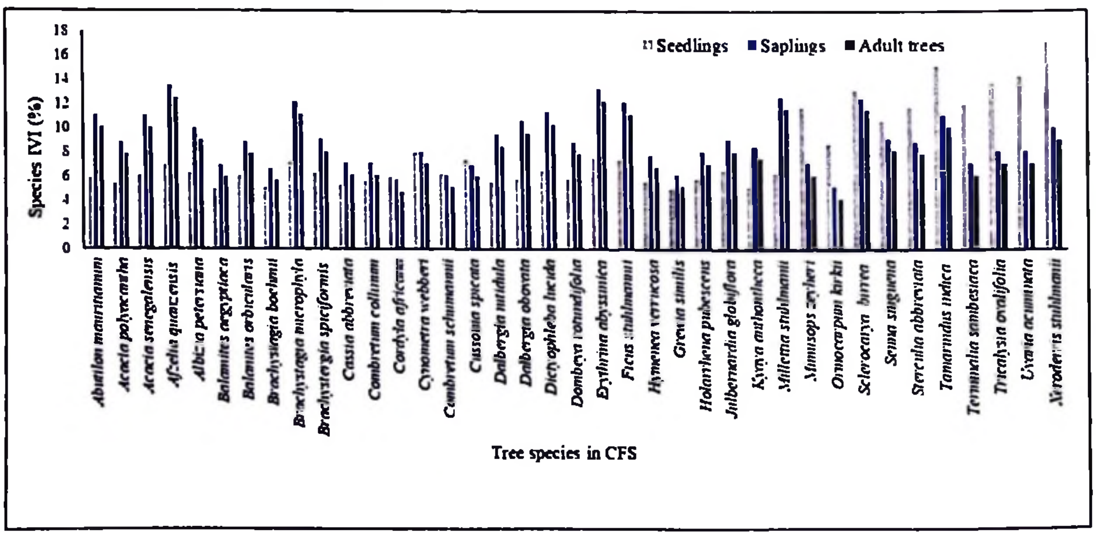


Figure 4- 8: Important Value Index in CFS

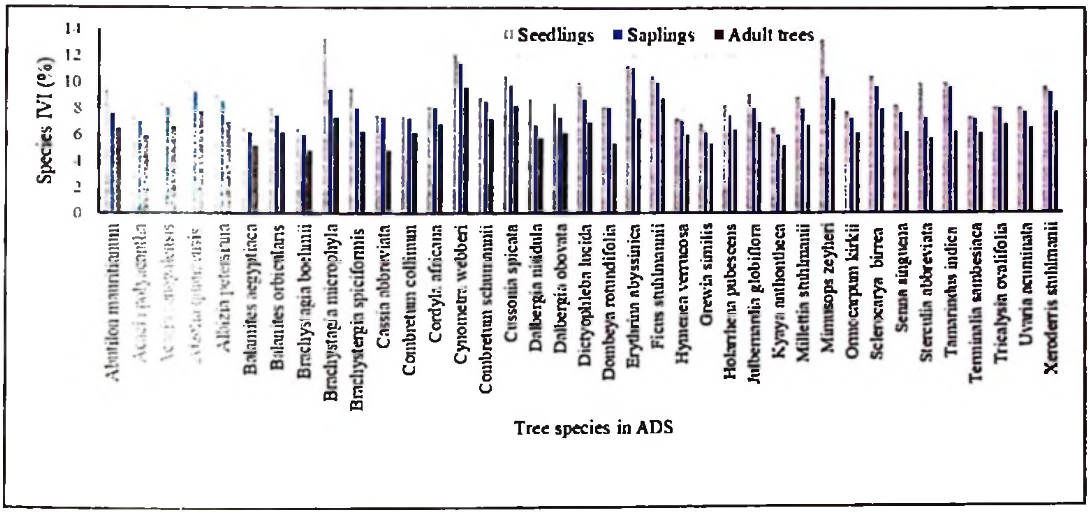


Figure 4- 9: Importance Value Index in ADS

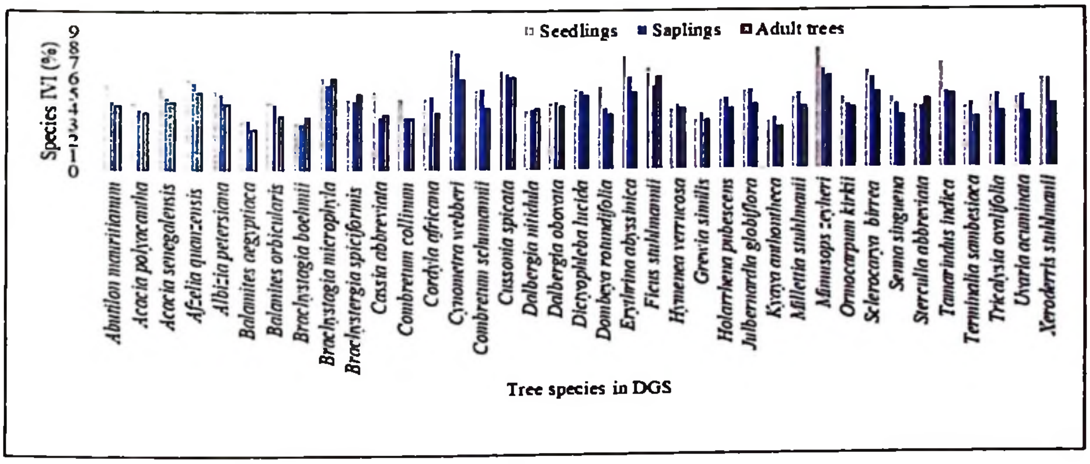


Figure 4- 10: Important Value Index in DGS

4.5. Tree Stand Parameters and Soil Properties

4.5.1. Tree Stand Parameters and Soil Physical Properties

There were strong positive correlation between soil physical properties (SPP) and tree stand parameters (TSP) across the land uses. The Monte Carlo test of significance of all canonical axes in CFS was F = 2.400, $p < .012$ for STP and SPP. In ADS, the F- test was 0.529, $p = .938$. In DGS, the significance of all canonical axes was F = 1.207, $p = .242$.

The species- environment correlation between STP and SPP for individual axis had the average values in the order of 0.435, 0.248 and 0.338 for CFS, ADS and DGS respectively. (Table 4-11).

Table 4- 11: Canonical correlation between Soil Physical Properties and Tree Stand Parameters (TSP) across Land Uses

	SPP vs. TSP in CFS				SPP vs. TSP in ADS				SPP vs. TSP in DGS			
Axes	1	2	3	4	1	2	3	4	1	2	3	4
EV	0.02	0.00	0.00	0.00	0.01	0.01	0	0	0.01	0.00	0.00	0.00
LG	0.36	0.19	0.11	0.19	0.19	0.14	0.08	0.08	0.31	0.21	0.15	0.15
SEC	0.55	0.45	0.42	0.32	0.36	0.25	0.18	0.20	0.45	0.36	0.26	0.28
CPVS	13.60	14.60	14.90	15.00	3.70	4.10	4.20	4.30	4.30	4.60	4.90	5.00
CPVSER	70.90	83.60	0.00	0.00	58.60	74.50	0.00	0.00	61.90	75.20	0.00	0.00

Where: SPP = Soil physical properties, TSP = Tree Stand Parameters, CFS = Coastal Forest Sites, ADS = Agriculture Disturbed sites, EV = Eigen values, LG = Lengths of gradient, SEC = Species-environment correlations, CPVS = Cumulative percentage variance of species data, CPVSER = Cumulative percentage variance of species-environment relation.

4.5.2. Tree Stand Parameters and Soil Chemical Properties

The canonical multivariate data analysis showed a Monte Carlo test of significance of all canonical axes between the correlation of soluble bases (Ca, Mg, K and Na) and tree stand parameters (density, height, basal area and volume (TSP)) as $F = 2.448$, $p = .018$ in CFS, $F= 0.687$, $p = .790$ in ADS and $F = 0.743$, $p = .808$ in DGS. The average species-environmental correction was 0.338 in CFS, 0.305 in ADS and 0.288 in DGS (Table 6-2). The Monte Carlo test of significance of all the canonical axes for the correlation between non-soluble elements (carbon, nitrogen and phosphorus-(CNP)) and TSP were $F = 0.816$, $p = .572$ in CFS, $F = 0.687$, $p = .790$ and $F = .070$, $p = .020$ in DGS. The average of species- environmental correlations was 0.47 in CFS, 0.223 in ADS and 0.392 in DGS (Table 4-12)

Table 4- 12: Canonical Correlation between Soluble Base and Tree Stand Parameters

	Soluble Bases and TSP in CFS				Soluble Bases and TSP in ADS				Soluble Bases and TSP in DGS			
Axes	1	2	3	4	1	2	3	4	1	2	3	4
EV	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00
LG	0.31	0.21	0.15	0.15	0.24	0.07	0.17	0.17	0.24	0.07	0.17	0.17
SEC	0.45	0.36	0.26	0.28	0.42	0.25	0.23	0.25	0.42	0.25	0.23	0.25
CPVS	4.30	4.60	4.90	5.00	4.00	4.40	4.40	4.40	4.00	4.40	4.40	4.40
CPVSER	61.90	75.20	0.00	0.00	71.50	80.40	0.00	0.00	71.50	80.40	0.00	0.00

Where: SPP = Soil physical properties, TSP = Tree Stand Parameters, CFS = Coastal Forest Sites, ADS = Agriculture Disturbed sites, EV = Eigen values, LG = Lengths of gradient, SEC = Species-environment correlations, CPVS = Cumulative percentage variance of species data, CPVSER = Cumulative percentage variance of species-environment relation.

Table 4- 13: Canonical Correlation between CNP and Tree Stand Parameters

	CNP vs. TSP in CFS				CNP vs. TSP in ADS				CNP vs. TSP in DGS			
Axes	1	2	3	4	1	2	3	4	1	2	3	4
EV	0.00	0.00	0.00	0.04	0.01	0.00	0.00	0.04	0.01	0.00	0.00	0.03
LG	0.16	0.10	0.04	0.68	0.27	0.09	0.14	0.78	0.34	0.23	0.24	0.87
SEC	0.48	0.21	0.19	0.01	0.36	0.26	0.28	0.01	0.57	0.49	0.49	0.02
CPVS	2.70	4.20	4.40	42.80	6.20	6.60	6.80	34.20	8.10	8.90	9.10	28.80
CPVSER	49.50	77.50	0.00	0.00	85.50	89.70	0.00	0.00	88.00	94.10	0.00	0.00

Where: SPP = Soil physical properties, TSP = Tree Stand Parameters, CFS = Coastal Forest Sites, ADS = Agriculture Disturbed sites, EV = Eigen values, LG = Lengths of gradient, SEC = Species-environment correlations, CPVS = Cumulative percentage variance of species data, CPVSER = Cumulative percentage variance of species-environment relation.

4.6. Diversity Indices and Soil Properties

4.6.1. Diversity Indices and Soil Physical Properties

The multivariate diversity indices had a positive correlation with soil physical properties (SPP). The canonical Monte Carlo tests of significance of all canonical axes in the correlation between SPP and Shannon index showed that $F = 1.103, p < .388$ in CFS, $F = 0.520, p = .714$ in ADS and $F = 0.932, p = .444$ in DGS. The average species-environmental correlation between SPP and Shannon index was 0.248 in CFS, 0.085 in ADS and 0.170 in DGS (Table 4-14).

Table 4- 14: Canonical Correlation between Soil Physical Properties and Shannon Index

	SPP vs. Shannon in CFS				SPP vs. Shannon in ADS				SPP vs. Shannon in DGS			
Axes	1	2	3	4	1	2	3	4	1	2	3	4
EV	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LG	0.02	0.00	0.09	0.09	0.01	0.00	0.05	0.05	0.05	0.05	0.10	0.10
SEC	0.31	0.34	0.33	0.01	0.22	0.01	0.01	0.00	0.29	0.29	0.01	0.00
CPVS	9.70	9.70	90.70	91.30	4.80	4.80	83.70	94.10	8.30	8.50	95.80	95.30
CPVSER	99.80	0.00	0.00	0.00	100.00	0.00	0.00	0.00	172.20	100.00	0.00	0.00

Where: SPP = Soil physical properties, TSP = Tree Stand Parameters, CFS = Coastal Forest Sites, ADS = Agriculture Disturbed sites, EV = Eigen values, LG = Lengths of gradient, SEC = Species-environment correlations, CPVS = Cumulative percentage variance of species data, CPVSER = Cumulative percentage variance of species-environment relation.

The canonical correlation between SPP and equitability showed that F = 0.093, p = .978. The results showed zero correlation between SPP and equitability in ADS and DGS. Indeed, the species-environment correlation was almost zero in ADS and DGS (Table 4-15). Interestingly, the canonical correlation between SPP and IVI showed that F = 0.042, p = .996 in CFS, F = 0.819, p = .620 in ADS and F = 0.633, p = .724 in DGS. The average of species-environmental correlation between SPP and IVI was 0.015 in CFS, 0.098 in ADS and 0.065 in DGS (Table 4-16).

Table 4- 15: Canonical Correlation between Soil Physical Properties and Equitability

	SPP vs. Equitability in CFS				SPP vs. Equitability in ADS				SPP vs. Equitability in DGS			
Axes	1	2	3	4	1	2	3	4	1	2	3	4
EV	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LG	0.00	0.00	0.03	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
SEC	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPVS	0.90	0.90	94.10	99.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPVSER	99.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Where: EV = Eigen values, LG = Lengths of gradient, SEC = Species-environment correlations, CPVS = Cumulative percentage variance of species data, CPVSER = Cumulative percentage variance of species-environment relation

Table 4- 16: Canonical Correlation between Soil Physical Properties and Independent Value Index

	SPP vs. IVI in CFS				SPP vs. IVI in ADS				SPP vs. IVI in DGS			
Axes	1	2	3	4	1	2	3	4	1	2	3	4
EV	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LG	0.02	0.00	0.16	0.16	0.01	0.00	0.03	0.03	0.04	0.01	0.21	0.16
SEC	0.06	0.00	0.00	0.00	0.39	0.00	0.00	0.00	0.26	0.07	0.00	0.00
CPVS	0.40	0.40	87.90	95.50	7.10	7.10	57.40	79.90	3.50	3.60	50.20	69.00
CPVSER	90.90	0.00	0.00	0.00	96.50	0.00	0.00	0.00	93.50	100.00	0.00	0.00

Where: SPP = Soil physical properties, TSP = Tree Stand Parameters, CFS = Coastal Forest Sites, ADS = Agriculture Disturbed sites, IVI= Importance Value Index, EV = Eigen values, LG = Lengths of gradient, SEC = Species-environment correlations, CPVS = Cumulative percentage variance of species data, CPVSER = Cumulative percentage variance of species-environment relation

4.6.2. Diversity Indices and Soil Chemical Properties

The canonical results showed that there were weak but positive correlation between soil chemical properties and diversity indices. The correlation between soluble bases and Shannon showed a correlation as in (Table 4-17) across CFS, ADS and DGS land uses. The Monte Carlo test of all the canonical axes showed that F = 0.574, p = .680 in CFS, F = 0.410, p = .804 in ADS and F = 0.910, p = .480 in DGS. The results showed a weak correlation between soluble bases and equitability across the land uses (Table 4-18).The canonical test of significance for all canonical axes between soluble bases and equitability showed that F = 0.119, p = .968 in CFS while ADS had F = 0.001, p = .001 in DGS the results showed that F = 0.011, p = .001. There were positive correlation between soluble bases and IVI (Table 4-19). In CFS, F = 0.083, p = .986, in ADS, F = 0.750, p = .664 while in DGS F = 0.374, p = .956.

Table 4- 17: Canonical Correlation between Soluble Bases and Shannon Index

	Soluble Bases vs. Shannon in CFS				Soluble Bases vs. Shannon in ADS				Soluble Bases vs. Shannon in DGS			
Axes	1	2	3	4	1	2	3	4	1	2	3	4
EV	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LG	0.01	0.01	0.09	0.09	0.01	0.00	0.05	0.05	0.01	0.00	0.05	0.05
SEC	0.18	0.18	0.00	0.00	0.28	0.00	0.00	0.00	0.28	0.00	0.00	0.00
CPVS	3.00	3.30	78.90	89.60	7.80	7.80	96.40	95.80	7.80	7.80	96.40	95.80
CPVSER	92.90	92.00	0.00	0.00	94.00	0.00	0.00	0.00	90.00	0.00	0.00	0.00

Where: SPP = Soil physical properties, TSP = Tree Stand Parameters, CFS = Coastal Forest Sites, ADS = Agriculture Disturbed sites, EV = Eigen values, LG = Lengths of gradient, SEC = Species-environment correlations, CPVS = Cumulative percentage variance of species data, CPVSER = Cumulative percentage variance of species-environment relation.

Table 4- 18: Canonical Correlation between Soluble Bases and Equitability

	Soluble Bases vs. Equitability in CFS				Soluble Bases vs. Equitability in ADS				Soluble Bases vs. Equitability in DGS			
Axes	1	2	3	4	1	2	3	4	1	2	3	4
EV	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LG	0.00	0.00	0.03	0.01	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.00
SEC	0.06	0.00	0.00	0.00	0.18	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPVS	0.30	0.30	84.40	99.10	3.20	3.20	97.60	92.20	0.00	0.00	0.00	0.00
CPVSER	84.70	0.00	0.00	0.00	99.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Where: SPP = Soil physical properties, TSP = Tree Stand Parameters, CFS = Coastal Forest Sites, ADS = Agriculture Disturbed sites, EV = Eigen values, LG = Lengths of gradient, SEC = Species-environment correlations, CPVS = Cumulative percentage variance of species data, CPVSER = Cumulative percentage variance of species-environment relation.

Table 4- 19: Canonical Correlation between Soluble Bases and Independent Value Index

	Soluble Bases vs. IVI in CFS				Soluble Bases vs. IVI in ADS				Soluble Bases vs. IVI in DGS			
Axes	1	2	3	4	1	2	3	4	1	2	3	4
EV	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LG	0.03	0.02	0.21	0.15	0.03	0.02	0.21	0.15	0.08	0.08	0.06	0.06
SEC	0.27	0.14	0.00	0.00	0.27	0.14	0.00	0.00	0.99	0.99	0.00	0.00
CPVS	3.20	3.70	59.60	79.60	3.20	3.70	59.60	79.60	97.40	98.60	99.50	99.10
CPVSER	76.90	98.00	0.00	0.00	76.90	98.00	0.00	0.00	97.00	98.00	0.00	0.00

Where: SPP = Soil physical properties, TSP = Tree Stand Parameters, CFS = Coastal Forest Sites, ADS = Agriculture Disturbed sites, IVI= Importance Value Index, EV = Eigen values, LG = Lengths of gradient, SEC = Species-environment correlations, CPVS = Cumulative percentage variance of species data, CPVSER = Cumulative percentage variance of species-environment relation.

The canonical correlation was positive between CNP and Shannon index across CFS, ADS and DGS (Table 4-20). The correlations were shown by $F = 0.127$, $p = .002$ in CFS, $F = 0.254$, $p = .002$ in ADS and $F = 0.097$, $p = .002$ in DGS. There were almost no established correlation between CNP and equitability across CFS, ADS and DGS (Table 4-21). The CNP and IVI had some positive correlation as shown in (Table 4-22). The test of significance of all the canonical axes were $F = 4.246$, $p = .014$ in CFS, $F = 2.729$, $p = .018$ in ADS and $F= 2.007$, $p = .060$ in DGS.

Table 4- 20: Canonical Correlation between CNP and Shannon Index

	CNP vs. Shannon in CFS				CNP vs. Shannon in ADS				CNP vs. Shannon in DGS			
Axes	1	2	3	4	1	2	3	4	1	2	3	4
EV	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LG	0.08	0.08	0.06	0.06	0.05	0.05	0.05	0.05	0.10	0.10	0.10	0.10
SEC	0.99	0.99	0.00	0.00	1.00	1.00	0.00	0.00	1.00	1.00	0.00	0.00
CPVS	97.40	98.60	99.50	91.10	99.30	99.50	99.80	99.10	99.70	99.00	99.10	89.20
CPVSER	73.70	90.00	0.00	0.00	75.70	90.00	0.00	0.00	90.80	90.00	0.00	0.00

Where: SPP = Soil physical properties, TSP = Tree Stand Parameters, CFS = Coastal Forest Sites, ADS = Agriculture Disturbed sites, EV = Eigen values, LG = Lengths of gradient, SEC = Species-environment correlations, CPVS = Cumulative percentage variance of species data, CPVSER = Cumulative percentage variance of species-environment relation.

Table 4- 21: Canonical Correlation between CNP and Equitability

	CNP vs. Equitability in CFS				CNP vs. Equitability in ADS				CNP vs. Equitability in DGS			
Axes	1	2	3	4	1	2	3	4	1	2	3	4
EV	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LG	0.02	0.00	0.03	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
SEC	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPVS	23.50	23.50	90.50	97.90	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CPVSER	90.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Where: SPP = Soil physical properties, TSP = Tree Stand Parameters, CFS = Coastal Forest Sites, ADS = Agriculture Disturbed sites, EV = Eigen values, LG = Lengths of gradient, SEC = Species-environment correlations, CPVS = Cumulative percentage variance of species data, CPVSER = Cumulative percentage variance of species-environment relation.

Table 4- 22: Canonical Correlation between CNP and IVI

	CNP vs. IVI in CFS				CNP vs. IVI in ADS				CNP vs. IVI in ADS			
Axes	1	2	3	4	1	2	3	4	1	2	3	4
EV	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LG	0.10	0.10	0.16	0.16	0.01	0.01	0.03	0.03	0.06	0.03	0.19	0.17
SEC	0.48	0.48	0.00	0.00	0.52	0.24	0.00	0.00	0.46	0.19	0.00	0.00
CPVS	23.30	23.60	90.20	98.00	14.20	16.40	56.10	76.00	11.10	11.60	43.10	60.10
CPVSER	77.00	90.00	0.00	0.00	87.70	90.00	0.00	0.00	89.50	90.00	0.00	0.00

Where: SPP = Soil physical properties, TSP = Tree Stand Parameters, CFS = Coastal Forest Sites, ADS = Agriculture Disturbed sites, IVI= Importance Value Index, EV = Eigen values, LG = Lengths of gradient, SEC = Species-environment correlations, CPVS = Cumulative percentage variance of species data, CPVSER = Cumulative percentage variance of species-environment relation.

4.7. Land Cover and Land Use, and ESV Dynamics

4.7.1. The LCLU Area across 2000, 2010 and 2016

In 2000, the study area (ha) had 46% forest coverage with 39% grazing land, while farmland use was 9%, shrub land covered 4%, wetland 1%, artificial surface by 0.4% and waterbody by 0.3%. In 2010, forest area was 42%, grazing land was 39% while farmland and shrub land covered 13% and 5% respectively. Wetland had 1% while waterbody and artificial surface each had 0.3% coverage. In 2016, forest covered 41%, grazing land 38%, farmland 13% and 6% shrub land. Wetland, artificial surface and waterbody were equal to that in 2010 (Table 4-23).

Table 4- 23: Estimated area and change for each LCLU category

	Total area coverage (ha) across the study periods			Gain/loss in land area		
LCLU	2000	2010	2016	2010-2000	2016-2010	2016-2000
Forest	353041 (46%)	322637 (42%)	316600 (41%)	-30404 (-9%)	-6037 (-2%)	-36441 (-10%)
Grazing land	301707 (39%)	296680 (39%)	295360 (38%)	-5028 (-2%)	-1320 (0.4%)	-6347 (-2%)
Shrub land	31538 (4%)	40342 (5%)	43288 (6%)	8804 (28%)	2946 (7%)	11751 (37%)
Farm land	70495 (9%)	96645 (13%)	101001 (13%)	26150 (37%)	4356 (5%)	30506 (43%)
Wetland	5825 (1%)	5766 (1%)	5713 (1%)	-59 (-1%)	-53 (-1%)	-112 (-2%)
Waterbody	2061 (0.3%)	2303 (0.3%)	2340 (0.3%)	242 (12%)	37 (2%)	279 (14%)
Artificial surface	2691 (0.4%)	2986 (0.4%)	3056 (0.4%)	295 (11%)	70 (2%)	365 (14%)
Total	767358	767358	767358			

Where: Values in percentage indicated in brackets

4.7.2. Gain or Loss of LCLU across 2000, 2010 and 2016

There were some changes on the area for each LCLU category (Figure 4-11). Between 2000 and 2016, farmland expanded more than other land categories at 43%, followed by shrub land, which increased by 37%, and waterbody and artificial surface increased by 14%. Between 2000 and 2016, forest decreased by 10%; grazing land and wetland decreased by 2% each. Although forest coverage dominated the study area across all years, this land shrank from 353 041 in 2000 to 322 637 in 2010 and then to 316 600 in 2016. Wetland size decreased from 5 825 in 2000 to 5 766 in 2010 and 5 713 in 2016. Grazing land decreased from 301 707 in 2000 to 296 680 in 2010 and 295 360 in 2016.

The highest relative proportions of decrease in the study area largely recorded in forest, grazing land and wetland (Table 4-23).

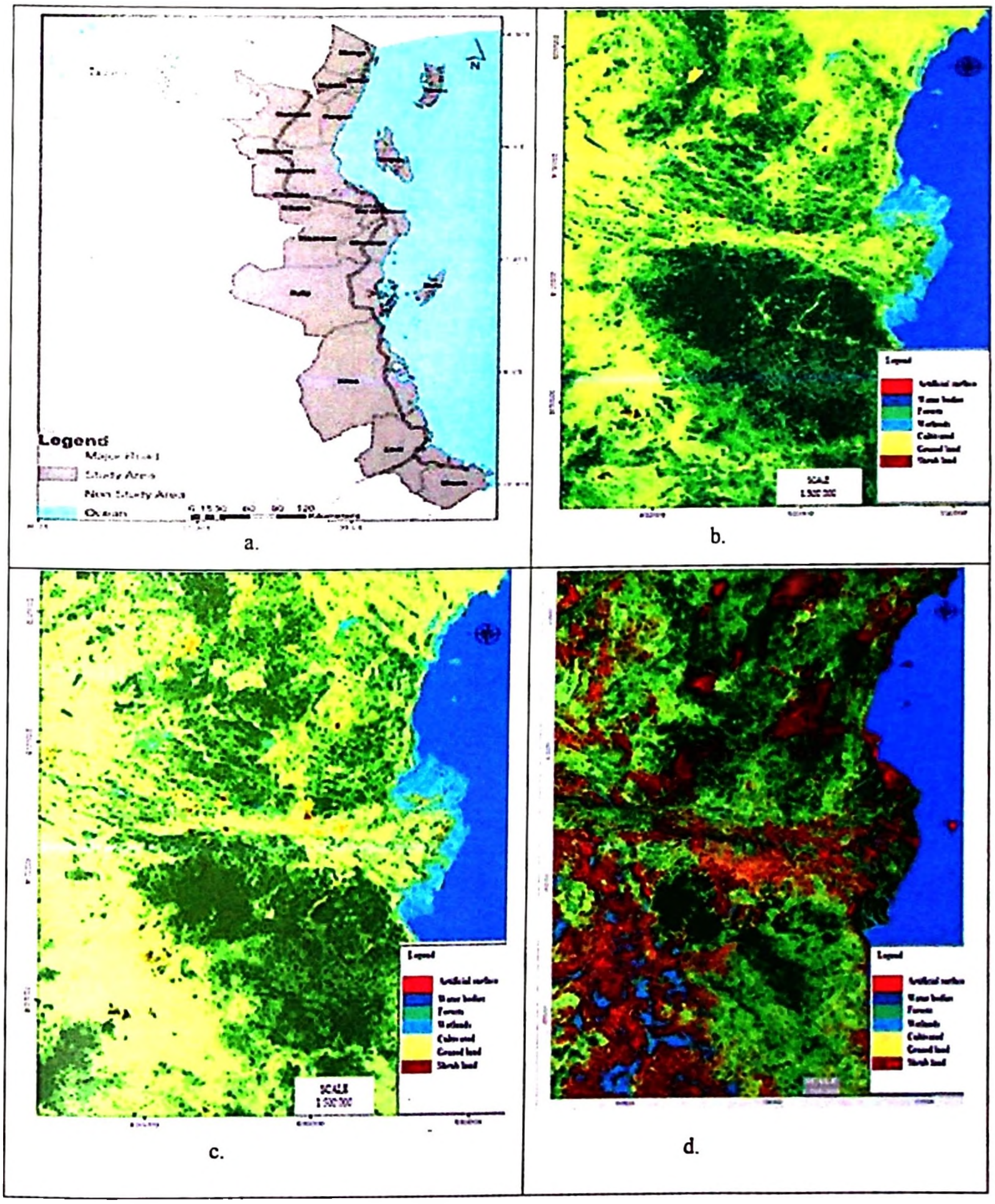


Figure 4- 11: a = Study area: Land cover and land use changes for 2000 (b), 2010 (c) and 2016 (d)

4.7.3. Changes of Ecosystem Service Values

Ecosystem service values (US$) varied across LCLU categories. Forest had 69% of the total ESV in 2000, which declined to 66% in 2010 and 65% in 2016. Grazing land had the second higher ESV at 19% in 2000 and 20% in both 2010 and 2016. Shrub land ESV was 5% in 2000 and 7% in both 2010 and 2016. Farmland ESV was 2% in 2000 and 3% in both 2010 and 2016. Wetland had 3% while waterbody had 1% across sixteen years. Between 2000 and 2016, the net ESV gains were in farmland, shrub land, and water body, while the net losses recorded for forest, grazing land and wetlands.

The annual ESV change was highly affected by loss in forest and gains on shrub land and farmland. The aggregated loss of forest, wetland and grazing land surpassed the gains accrued in shrub land, farmland and waterbody. Hence, the overall ESV was negative across 2000, 2010 and 2016 (Table 4-24). There was perfect positive linear correlation between area changes and ESV in forest, shrubs, farmland, wetlands and waterbody (p = 1.00). Additionally, only the grazing land had strong positive correlation (p = .768), and the correlation was significant at the 0.01 level (Figure 4-12).

Table 4- 24: Ecosystem service value (USS) and their changes across 2000, 2010 and 2016

	Total ESV across the study periods			Gain/loss in ESV across the study periods		
LCLU type	2000	2010	2016	2000-2010	2010-2016	2000-2016
Forest	3.9×10^{8} (69%)	3.5×10^{8} (66%)	3.5×10^{7} (65%)	-3.3×10^{7} (9%)	-6.6×10^{6} (2%)	-4.0×10^{7} (12%)
Grazing land	1.1×10^{8} (19%)	1.1×10^{8} (20%)	1.1×10^{8} (20%)	-1.8×10^{6} (2%)	-4.7×10^{5} (1%)	-2.3×10^{6} (2%)
Shrub land	2.8×10^{7} (5%)	3.6×10^{7} (7%)	3.9×10^{7} (7%)	7.8×10^{6} (28)	2.6×10^{6} (7%)	1.0×10^{7} (27%)
Farm land	1.2×10^{7} (2%)	1.6×10^{7} (3%)	1.7×10^{7} (3%)	4.4×10^{6} (37%)	7.4×10^{5} (5%)	5.1×10^{6} (30%)
Wetland	1.7×10^{7} (3%)	1.6×10^{7} (3%)	1.6×10^{7} (3%)	-1.7×10^{5} (1%)	-1.5×10^{5} (1%)	-3.2×10^{5} (2%)
Waterbody	6.7×10^{6} (1%)	7.4×10^{6} (1%)	7.5×10^{6} (1%)	7.8×10^{5} (12%)	1.2×10^{5} (2%)	9.0×10^{5} (12%)
Artificial surface	-	-	-	-	-	-
Total	5.6×10^{7}	5.3×10^{8}	5.3×10^{8}	-2.2×10^{7}	-3.7×10^{6}	-2.8×10^{6}

Where: Value in percentage indicated in brackets

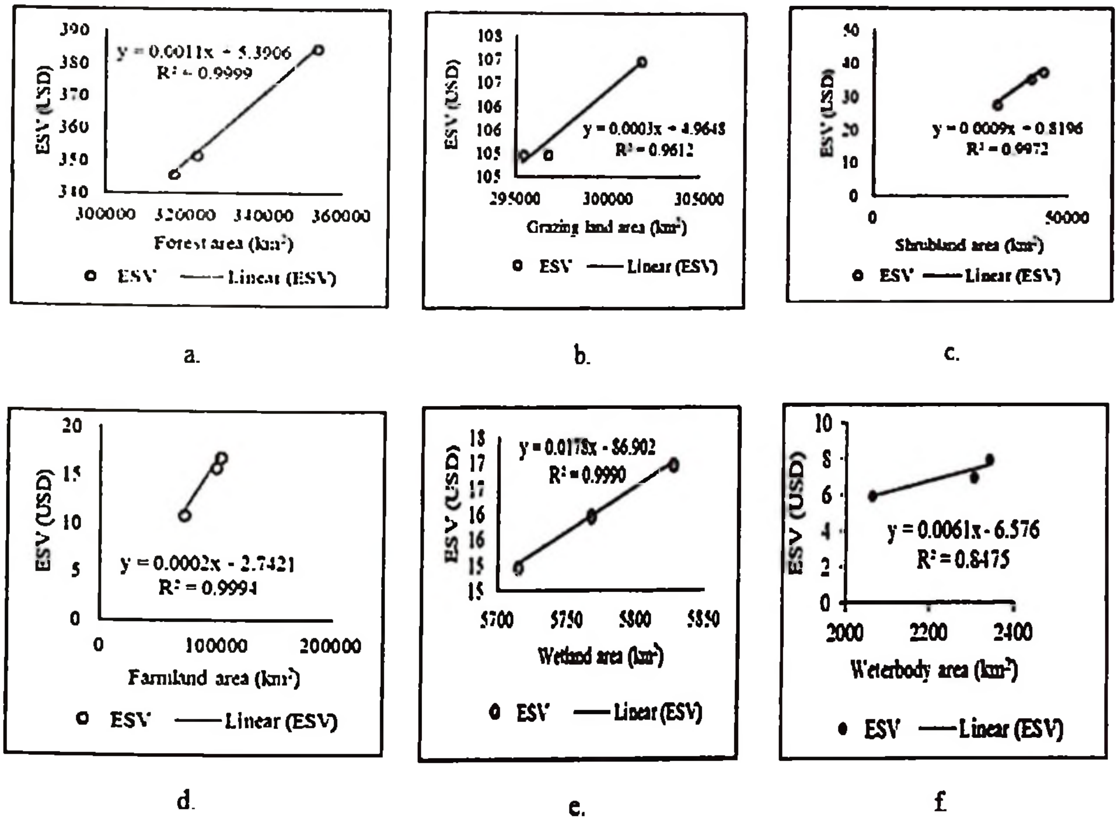


Figure 4- 12 : Correlation between LCLU and ESV changes: a. forest and ESV, b. grazing land and ESV, c. shrub land and ESV, d. farmland and ESV, e. wetland and ESV and f. water body and ESV

4.7.4. Human to Ecosystem Service Values

The assigned monetary values of each LCLU category and their total human to ESV varied across 2000, 2010 and 2016. The benefits of human welfare from coastal ecosystem services values declined across LCLU categories in the sixteen years. The total H-ESV ratio declined from 80 in 2000 to 64 in 2010, and then to 46 in 2016 (Table 4-25). Of these changes, forest and grazing land highly contributed on the total H-ESV decline. This trend illustrates a negative relationship between ESV and population across the study years.

4.7.5. Interviewed Population

The study interviewed 490 (20%) at Mbwewe, 540 (22%) at Kwaluhombo, 522 (21%) in Kwang'andu and 697 (28%) at Mpaji and 231 (9%) at Changalikwa villages. The village firewood collectors, crop agriculturists, charcoal business people, and livestock keepers provided the socioeconomic information. One district and one ward agriculture officer, two district and regional forest officers, one ward executive officer and eight village leaders were interviewed during the spoken surveys. The findings from these sample sub-categories were generalized as shown in (Figure 4-13).

Table 4- 25: Human -ESV across LCLU categories and years

LCLU type	Human-ESV across the study periods			Percentage gain/loss in H-ESV		
	2000	2010	2016	2010-2000	2016-2010	2016-2000
Forest	55.8	42.1	30.0	-24.5	-28.9	-46.3
Grazing land	15.5	12.6	9.1	-18.7	-27.8	-41.3
Shrub land	4.1	4.3	3.4	5.7	-22.2	-17.7
Farm land	1.7	2.0	1.5	13.3	-24.2	-14.1
Wetland	2.4	2.0	1.4	-18.2	-28.2	-41.2
Waterbody	1.0	0.9	0.7	-7.6	-26.3	-32.0
Artificial surface	-	-	-	-	-	-
Total	80.4	63.8	46.0	-20.6	-28.0	-42.9

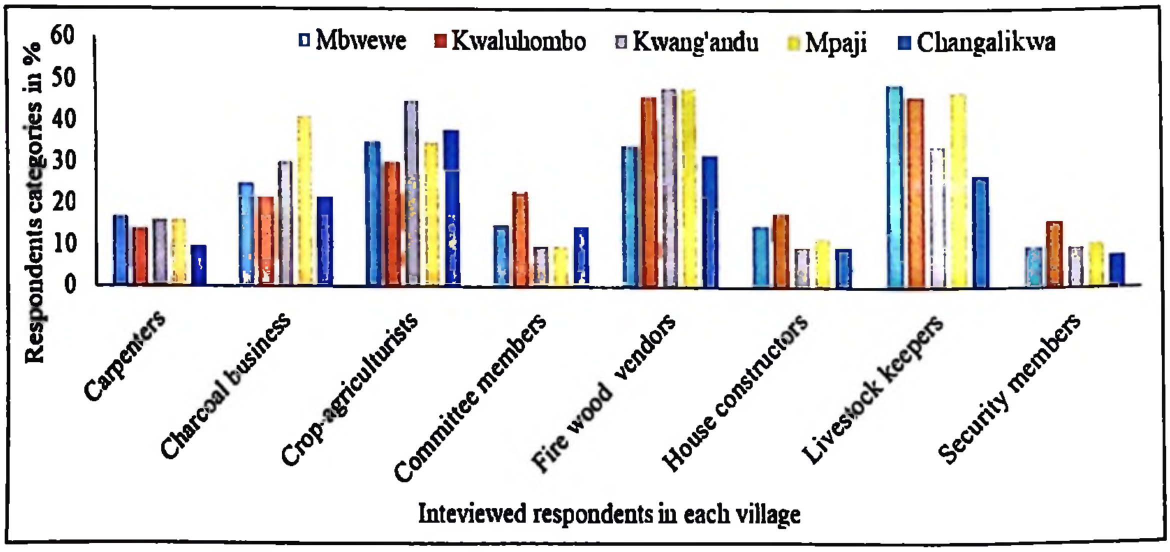
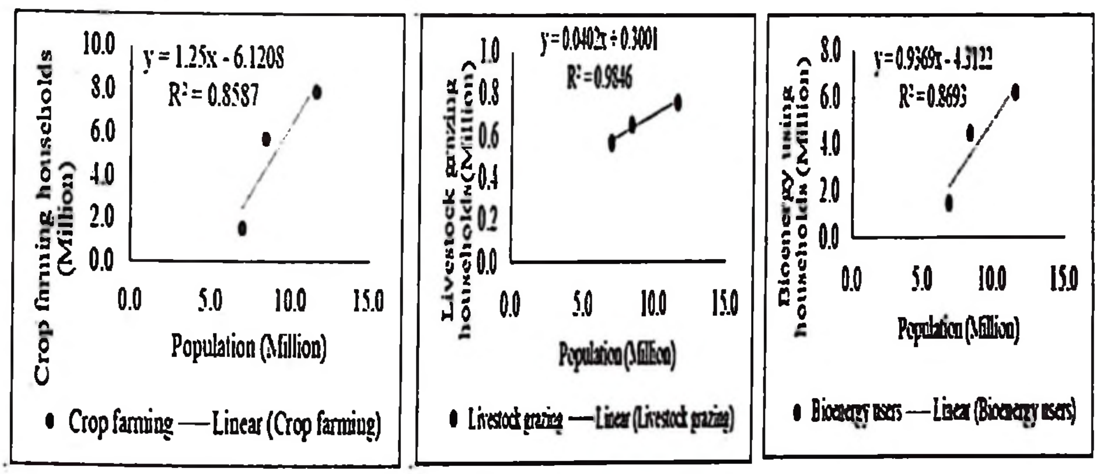


Figure 4- 13: Interviewed respondents from the representative study sites

Population data show an increase along the coastal zone of 23% of between 2000 and 2016. Indeed, during this time, some major socioeconomic activities increased tremendously. For example, there were a perfect positive correlation between population increase and households involved with crop farming (R^2 = 08587), livestock keeping (R^2 = 09846), and bioenergy use (R^2 = 0.9846) (Figure 4.14).



a. b. c.

Figure 4- 14: Correlation across population and the major drivers of LCLU change

4.7.6. Factors Causing LCLU Change Community Awareness

A major factor contributing to LCLU change was bioenergy (i.e. woods for cooking and heating), clearing virgin land for development of crop farming. Other activities identified by local community were uncontrolled livestock grazing, and collection of materials for construction (e.g. timber, poles and ropes). Interestingly, uncontrolled fire and urbanization ranked at the lowest than other LCLU change factors across all the study sites (Figure 3-15).

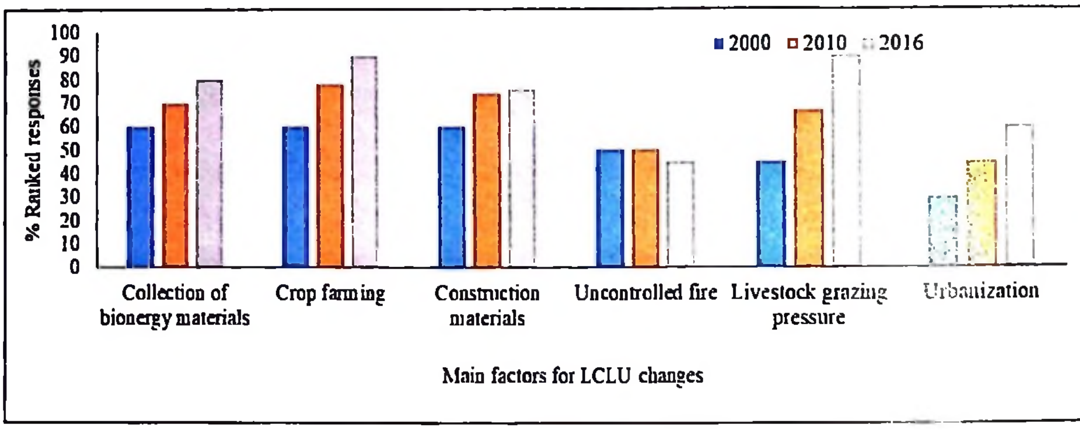


Figure 4- 15: Respondents' awareness on socioeconomic that cause LCLU change

4.8. Restoration Interventions on the Disturbed Sites

4.8.1. Households Size and Ages Groups for Restoration data

The household size across all the five villages dominated by age between 37-55 years. The second dominating age range was between 18-36 years except in Mpaji village. Furthermore, age class of above 55 years was at low percentage across the villages except in Changalikwa village where it was equal to that of between 18-36 but less than 37-55 years. Cumulatively, the age between 18-55 years was higher than above 55 years (Figure 4-16). The results show that productive members actively participate in tree planting and caring for the retained trees dominate the population.

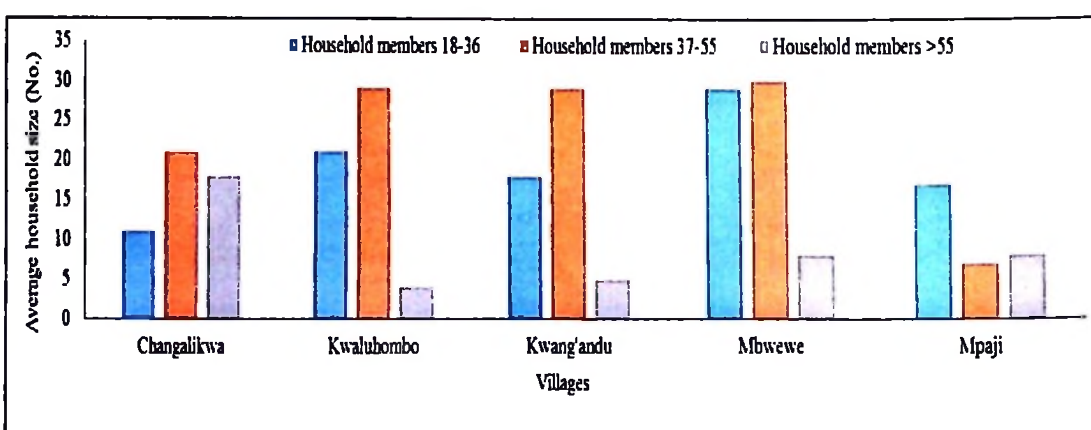


Figure 4- 16: Households average age's distribution

4.8.2. Trees per Households per Residential Duration

Planting of trees and retaining natural existing species recorded from each households and verifies in the field. Planting trees is a practice done at low pace than retaining naturally growing trees in the disturbed sites. On one hand, households had the average of 1-50 (33%) trees planted during the entire residential duration. On the other hand, there were higher number of trees retained as presented between 150 and above 200 per households in the disturbed or abandoned farmlands. On average, each household had higher number of retained trees than planted ones (Table 4-26).

Table 4- 26: Quantities of tree per households per residential duration

Number of trees	Planted trees frequency (percentage)	Retained trees frequency (percentage)
1-50	84 (33)	41 (16)
51-100	63 (25)	47 (18)
101-150	41 (16)	26 (10)
151-200	32 (13)	71 (28)
>200	35 (14)	71 (28)

Where bracketed values represent the percentage of tree numbers

4.8.3. Activities for Restoration of Disturbed Sites

Respondents showed their participation for restoration of the disturbed and/or degraded forest sites in several activities. The average participation rate and activities varied across villages with some activities being highly practiced on each village. For example, allowing natural trees to grow, reforestation and afforestation were higher than other activities (i.e. 15 - 38 %). The involvement of participants on activities such as site preparation for trees planting and plantation were averagely low (i.e. 7 - 22%) across the surveyed villages (Figure 4-17).

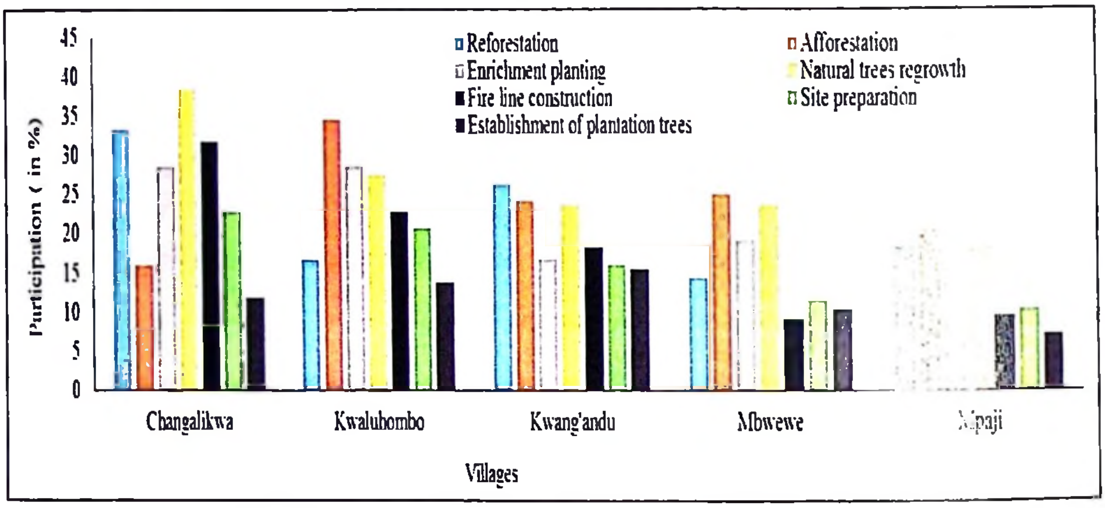


Figure 4- 17: Activities which households are involved for restoration of disturbed sites

4.8.4. Strategies for Controlling Forest Disturbances

The major strategies which were identified for controlling further forest disturbances were permitting guided harvesting by issuing licenses, providing incentives to forest resource users to forgone exploitation of forests resources and limiting utilization of some forest areas for any human activities accompanied by reallocating families (excluding) people to permit restoration process to naturally or artificially take place. Across the study area, the average values for exclusion was 45%, followed by certification at 38% and compensation at 16% (Figure 4-18).

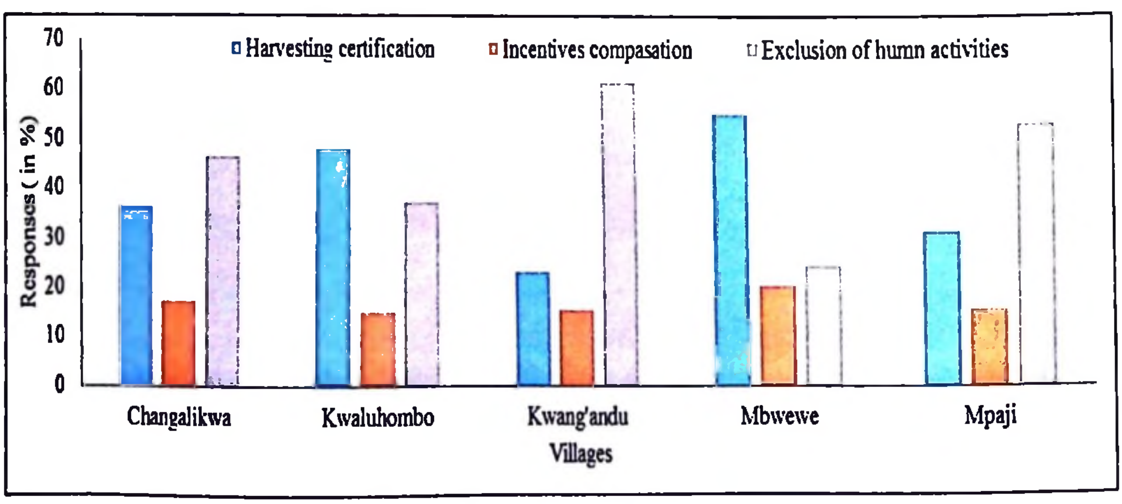


Figure 4- 18: Strategies for controlling expansion of forest disturbances

4.8.5. Drivers for Planting and Retaining of Trees

Driving factors for planting and or retaining trees dominated by keeping trees to gain volumes for timber harvesting, followed by improving soil health, which aligned with controlling soil erosion and fuel wood. Across the villages, protection of the environment, for food and medicinal values ranked the lowest (Figure 4-19).

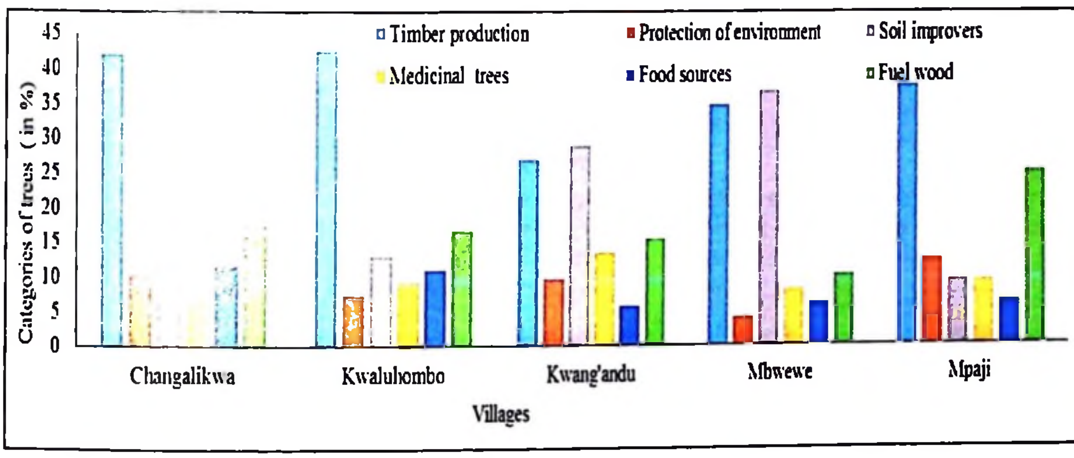


Figure 4- 19: Drivers for planting and retaining some categories of trees

4.8.6. Factors Influencing Tree Planting Practices

Restoration of disturbed forest site by trees planting challenged by many factors. The overall observation is that household income, climatic factors (citing shortage of rains) and poor enforcement of national and by-laws significantly hinder tree planting in the study area. Indeed, poor extension services because of lacking enough forest trained staff and inadequate supply of seeds and seedlings hinder trees planting options. Large family size identified as a significant opportunity to supply labor force in tree planting. Interestingly, across the villages, lack of incentives for restoration not considered as significantly affecting restoration. Other factors such as encroachment of human activities (crop farming and livestock grazing), land ownerships, residence duration and donor dependence showed a non-significant effects on tree planting (Table 4-27).

Table 4- 27: Tree planting and factors affecting tree planting for disturbed sites restoration

Xi	Y_1 (R^2= 0. 349)			
	Beta	SE	t	Sign. Level
Extension Services	.116	.251	1.856	.045*
Incentives support	.033	.251	.566	.572NS
Household income	.261	.260	4.296	.001***
Climatic factors	.551	.235	9.455	.001***
Encroachment	.109	.259	1.814	.001***
Household size	.107	.242	1.778	.007***
Land ownerships	.064	.243	1.115	.002***
Residence duration	.047	.283	.765	.445NS
Donor dependence syndrome	.020	.239	.378	.706NS
Law enforcement	.156	.196	2.977	.003***
Availability of planting materials	.078	.252	1.325	.016**

Xi = Independent variables (factors affecting tree planting)
Beta = Regression coefficients
Y_1 = Dependent variable (number of planted trees)
R^2 = Coefficient of determination (goodness of fit of the model)
SE = Standard error
t = Student's t-test
* = Statistically significant at 0.05 level of significance
** = Statistically significant at 0.01 level of significance
*** = Statistically significant at 0.001 level of significance
NS = Statistically non-significant at 0.05, 0.01 and 0.001 levels of significance

Table 4- 28: Factors affecting the retention of the naturally growing trees

Xi	Y_2 (R^2= 0.262)			
	Beta	SE	T	Sign. Level
Extension services	.018	.118	.260	.795NS
Incentives support	.061	.119	.961	.038*
Household income	.069	.116	1.105	.001***
Climatic factors	.234	.111	3.733	.001***
Encroachment	.307	.122	4.736	.001***
Household size	.005	.115	.081	.935NS
Land ownerships	.051	.111	.850	.396NS
Residence duration	.215	.134	3.235	.001***
Donor dependence syndrome	.022	.112	.389	.698NS
Law enforcement	.229	.093	4.044	.008**

Xi = Independent variables (factors affecting tree planting)
Beta = Regression coefficients
Y_2 = Dependent variable (number of planted trees)
R^2 = Coefficient of determination (goodness of fit of the model)
SE = Standard error
t = Student's t-test
* = Statistically significant at 0.05 level of significance
** = Statistically significant at 0.01 level of significance
*** = Statistically significant at 0.001 level of significance
NS = Statistically non-significant at 0.05, 0.01 and 0.001 levels of significance

4.8.7. Factors Influencing the Retention of Trees

Retaining trees in farms and or restoration sites emphasized as an approach to restore disturbed sites. Retaining approach affected significantly by encroachment due to the expansion of human activities (crop farming and livestock grazing), over-dependence on forests to supplement household income, lack of payment systems to compensate for restoration and non-use of forest resources. Also, factors like persistent drought, residence duration (in the sense that intruders and invaders affect natural systems) and lack of vigorous law enforcements significantly affect retaining of trees. Other factors like incentives support, household size, land ownerships and donor dependent syndrome have non-significant-effects on retaining trees for restoration (Table 4-28).

4.8.9. Correlation Restoration Trees and Influencing Factors

The results in (Table 4-29) show that there were positive correlation between tree planting and household income, household size, land ownerships, support from donor's projects, resident duration and availability of planting materials. In regards to retaining trees, lack of incentives support, poor climatic factors, encroachment by human activities and poor law enforcements reported to have significant impacts.

Table 4- 29: Correlation between trees planted and retained, and factors affecting these two interventions

6		Factors										
Trees R		ES	IS	HHI	CF	EN	HHS	LO	DDS	RD	LE	PMA
PT	PC	0.116	-0.089	0.306**	0.192**	-0.149*	0.257**	0.086	0.282**	0.085	-0.002	0.289**
	S	0.063	0.155	0.001	0.002	0.018	0.001	0.172	0.001	0.172	0.969	0.001
RT	PC	0.059	0.125*	0.076	0.355**	0.343**	0.081	0.077	0.040	0.171	0.218**	-
	S	0.351	0.046	0.229	0.001	0.001	0.200	0.220	0.530	0.006	0.001	-

*Correlation is significant at the 0.05 level (2-tailed); **Correlation is significant at the 0.01 level (2-tailed)

Where PT = planted trees, RT = retained trees, R = correlation, PC = Pearson correlation, S = Significance, ES = extension services, IS = incentives support, HHI = household income, CF = climatic factors, EN = encroachment by human activities, HHS = house hold size, LO = Land ownerships, RD = residence duration, DDS = donor dependent syndrome, LE = law enforcement PMA = availability of planting materials.

8.9.10. Restoration and Forest Trend in the Coastal Forests

Although efforts are in place to restore the disturbed forests, yet data show that forest gain is less than forest loss per unit time. The loss results into negative net loss for example from 2001 to 2017 (Figure 4.20 = forest loss and Figure 4-21=forests gain in some regions.

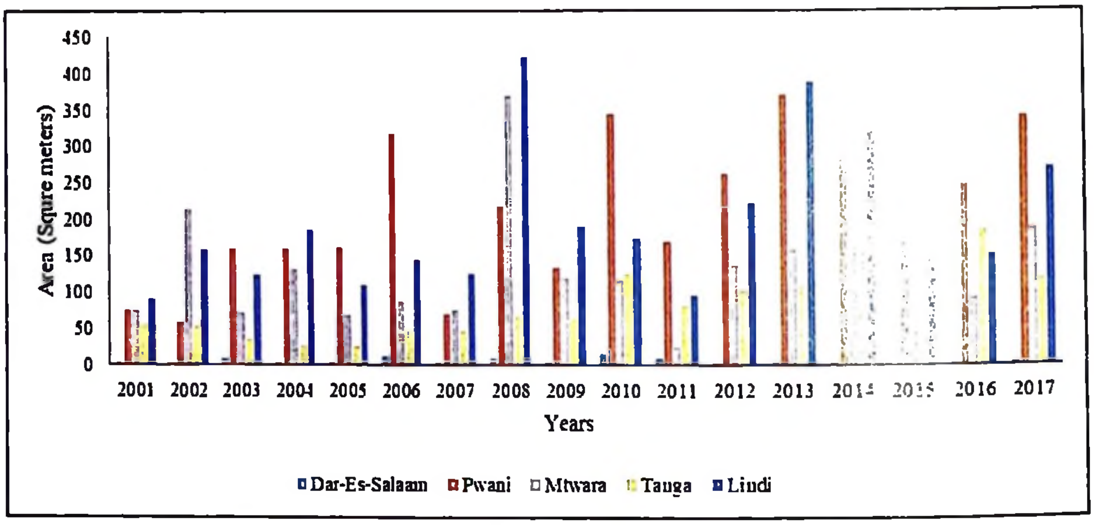


Figure 4- 20: Forest loss (Area × 10^6) along the coast areas of Tanzania (2001 to 2017)

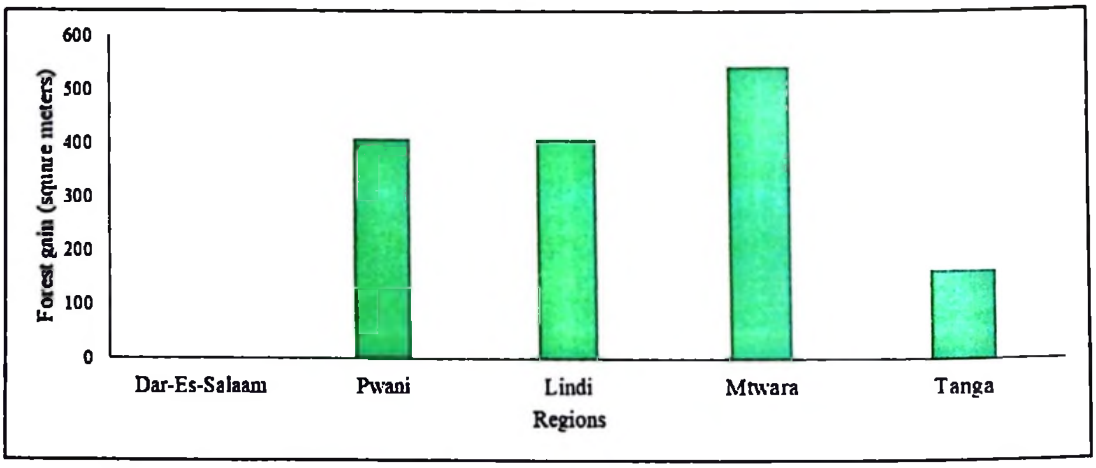


Figure 4- 21: Forest gains (Area × 10^6) along the coastal zone of Tanzania (2001 - 2017)

CHAPTER FIVE: DISCUSSION

5.1. Comparisons of Soil Physical Properties across Disturbed and Non-disturbed

5.1.1. Variation of EC across Land Uses

A higher quantity of EC observed in CFS followed by DGS and then ADS. Closed forest sites had higher CE but contained less amount of CEC as in DGS. This kind of variation can be used to support that intact soil patches (the control) are fertile agreeing (Sudduth et al. 2003; De Caires et al. 2014). Electrical conductivity in CFS was higher than that in other LU indicating that CFS contain higher amount of organic matter from vegetation residuals compared to other LU (Ryšan and Šařec 2008). The status of EC and CEC in ADS and DGS showed variations. These variations are significant to draw a conclusion that crop-agriculture largely contributes to disturb soil properties than livestock grazing as supported by Kane (2015).

It is possible that soluble bases depleted in cultivated land than intact and grazed sites. Cultivation of lands for crop production disturbs organic matter content and water holding capacity of the soil because the process involves removal of vegetation cover thus exposing soils into evaporation than livestock grazing. The electrical conductivity of dry soils is much lower than that of moist soil as in the CFS lands use (Ryšan and Šařec 2008). This variation is a good indicator for optimizing forest management choices if focusing on restoring soluble bases. In this view, priorities should be to restore forest sites disturbed by crop-cultivation followed by grazed area while protecting the CFS.

5.1.2. Correlation of EC within and across Land Uses

The positive correlation between CE and CEC within and across LU sites is a good indicator that EC is used to gauge the health of soils and thus provides information for management of different LU (Doolittle and Brevik 2014; Rabenberg and Kniffen 2014). The results shows some correlations, which varied across the LU sites as supported by (Bahrami et al. 2010). The trend of EC in CFS > DGS > ADS implies that degraded sites of coastal forests have low amount of soluble salts when compared to intact forest patches. A weak positive correlation observed between EC in ADS and DGS, indicating that the amount of EC across LU declines from CFS towards ADS. The correlation trend in ADS and DGS means that crop-agriculture affect soil properties such as salts than livestock grazing in the coastal forests.

Low amount of soluble bases in agriculture sites is supported by the fact that for more than fifty years farming activities took place in these sites without the addition of fertilizers (Silayo et al. 2006). There is low levels of EC in ADS and DGS as related to poor salts accumulation in ADS and DGS because in these land there had been vegetation destruction, which affects the complex integration of the primary natural resources: soil, water and vegetation (Islam and Weil 2000).

The impacts of human activities in soils of tropical forests is supported by that these forests have low ability to buffer especially when affected by human activities mainly agricultural practices (Islam and Weil 2000). Another reason for low EC in ADS and DGS is associated with low water content in these sites. Crop-agriculture and livestock grazing leave the land bare in some seasons of the year, exposing it to evaporation and drying (Ryšan and Šařec 2008). Therefore, agriculture stands a major contributor to affect CE and CEC across LU sites.

5.1.3. Electrical Conductivity and Soil Texture

The results show that EC increases as clays and silt percentage increase except in the sites dominated by sandy soils in CFS and ADS. These observations are also supported by other scholars Paillet et al. (2010); Doolittle and Brevik (2014). Across all the LU, there was a positive correlation between EC and clays followed by silts dominated sites agreeing with Sudduth et al. (2003). Sites with a relatively large quantity of clay sized minerals had large EC because water acts as colloidal particles, displacing a surface charge, and this charge is influenced by the mineralogy of the particles as well as the acidity and electrolyte composition of the water as supported by Makineci et al. (2015). The variation of EC and ST across LU sites indicates that although livestock grazing is pointed out, as among the activities contributing to disturb forest soils, the impacts of this activity in sand soils is not equated to those of sand soils in crop-agriculture sites.

One of the reasons is that grazed sites contain some kinds of vegetation left by browsers and grazers. These remnant vegetation help to conserve moisture to contribute in maintaining some quantities of soluble bases agreeing with Sudduth et al. (2003). This observation is generally contradicting with other scholars as soils containing high amount of clay have numerous, small water-filled pores that are quite continuous and usually conduct electricity better than sandier soils.

The main reasons for this contradiction are not documented and were not tested in this study, therefore this contradiction suggest that there might be intrinsic factors, which were not discovered in this study. However, in this work, an agreement just drawn from Sudduth et al. (2003), that EC is a good way to estimate soil texture of the surface layers.

5.1.4. Electrical Conductivity and Elevation

Although not significant, the results in this study showed that there was a negative correlation between elevation and EC across all the sites. These findings suggest that there are some variations of salts accumulation along the coastal forest ecological gradient. High elevations above 417 masl had low salts than the lower bottoms similarly to Charan et al. (2013). This condition suggests that regardless of the nature of the LU, different sites at lower bottoms have accumulated higher amount of salt ions ($Ca2+$, $Mg2+$ $K+$ and $Na+$ than the higher elevations in the order of CFS > DGS > ADS. The main reason to support this trend is that soluble salts are washed down the lower bottoms across the LU contrary with Makineci et al. (2015).

The main factor for EC variation against elevation is that higher elevations disturbed by crop production and livestock grazing pressure are exposed to excessive evaporation and drying. Excessive evaporation lowers the EC across all the LU sites at high elevations. With this trend, foresters and ecologists should be aware that management of forest ecosystems must address a full range of differences in soil properties across LU, ST and elevations because there is positive correlation between EC, CEC and ST in this study like in many other ecological gradients as documented in Charan et al. (2013).

5.1.5. Bulk Density Variation across Land Uses

The Bd in CFS was lower than that in ADS and DGS as also reported in Bahrami et al. (2010). These variations support that forest sites have high organic matter, compared to ADS and DGS. The variation between LU and Bd across the LU sites is associated with the collapse of soil aggregates and the clogging of voids and production of dusts in ADS and DGS (Certini 2005). It is noticeable that long term cultivation and livestock grazing in these areas have contributed to destroy soil organic matter and weakening the natural stability of soil aggregates hence making them susceptible to damage caused by water and wind (USDA 2008). Across LU sites, the Bd decreased against elevation, the trend that indicates that human activities taking place in the elevation above 417 masl affect more soil organic matter than in the lower bottoms.

A weak correlation between Bd and elevation in ADS showed that crop-agricultural activities increase the Bd, but the impacts of this activity is not significant because in this study, sand decreased with the increase in altitude in ADS hence lowering Bd at high elevations agreeing the results of Charan et al. (2013).

Indeed, the study found that at the higher elevation there is much compaction of soil because of livestock grazing pressure, occurring infrequently in the lower bottoms, dominated with crops cover in some seasons of the year. Since Bd is affected by soil structure and texture, organic matter, and climatic condition (temperature and rainfall) (USDA 2008), it is suggested that another study be conducted to determine how spatial and temporal variation of forests disturbances affect soil properties along the ecological gradient, and across different LU in tropical coastal forest of Tanzania beyond the scope of this study.

The difference in electrical conductivity, soil texture and bulk density across the land uses indicates that crop-agriculture and livestock grazing affect soil physical properties. The correlations across electrical conductivity, soil texture, cation exchange capacity and elevation established in this study can be used to indicate the status of soil fertility under different land uses in tropical coastal forest ecosystems. These variations suggest that in order to restore the disturbed sites, there must be plans to exclude crop-agriculture and livestock grazing. However, in a situation where crop-agriculture and livestock grazing are unavoidable for example because of the long terms existing dependence of local peoples' livelihoods with nature (as in this study local communities have been depending on this forest a period of more than 50 years), the best option could be integrating these activities in management plans.

In such management plans, there must be application of measures to decrease soil disturbance and increasing soil organic matter by practices such as introduction of cover crops, crop residues incorporations and reduced tillage. Indeed, livestock keepers must adopt grazing systems that minimize livestock traffic and adhere to recommended minimum grazing pressure to reduce and or prevent soil disturbances. For these measures to be practical and successful there must be a further investigation and settings that gives limits of crop-agriculture and livestock grazing activities within the carrying capacity of coastal tropical forests

5.2. Carbon, Nitrogen and Phosphorus across Intact Uses

This discussion presented the findings while considering that the study had some challenges from lack of baseline data on TN, TC and P status along the coastal zone of Tanzania. Hence, the discussion mainly focuses on the existing differences of soil nutrients and identifying the possible causes of variation under the hypothesis that disturbance type and associated cumulative severity affect the distribution and structure of forest vegetation. Disturbances in vegetation in turn affects inputs in soils subjected to disturbances (Amato et al. 2011). To establish the variation, this study used CFS as a control for comparison. These control sites were selected from maps and from the historical background of the experiences of the local people, which these sites have kept intact compared to those subjected into crop agriculture and livestock grazing. Indeed, the findings suggest that disturbances cause impacts on above ground and underground forest ecosystems hence causing differences in nutrients across different land uses (FAO 2009; URT 2015).

5.2.1. Variation of Total Nitrogen across Land Uses

Forests subjected to crop-agriculture and livestock grazing have different TN status. Closed forest sites contain a higher amount of TN than that found in ADS or DGS. It is explained that both crop-agriculture and livestock grazing contribute to making the soil susceptible to erosion and other processes which results into loss of nutrients. In this study, DGS had the lowest amount of TN across all the three LU. The low content of TN in DGS concurs with Golluscio et al. (2009); Xing et al. (2014) and Zhong et al. (2014).

The findings suggest that grazing disturbance reduces nitrogen mineralization, a process occurring in most of these lands when there is low moisture content following bare land exposure to solar radiation. These findings agree with other researches that low TN content in DGS is because livestock grazing decreases the input of organic matter and expose litter to photo-degradation (He et al. 2011; Salazar et al. 2000; Qu et al. 2016). Photo-degradation cause excessive loss of N from grazed land (Golluscio et al. 2009). However, the low content of TN in DGS is contrary to findings by Britton, Pearce, and Jones (2005); Bai et al. (2012). This controversial is that, it was expected DGS to contain higher amount of TN because of inputs from livestock excreta mainly urine as documented in Hoogendoorn et al. (2010).

From the results, it seems that the small amount of N inputs from excreta does not make up for the amount lost because of disturbed biomass. Low return of N from grazing animals explained in part by the fact that that livestock grazing in these forest sites is mainly a free-range system, under which animals are randomly grazed and therefore there is no guarantee of the return of nutrients from excreta in a particular piece of grazed land. The relatively low amount of TN in DGS as compared to that in ADS explained by the fact that while excessively grazed sites are left bare, cultivated farms have the advantage of harboring plant species in some seasons (crops and weeds) that check soil erosion and photo-degradation. In addition, there is partial recycling of nutrients from crop residues and weed decomposition on crop lands. Therefore, it is reasonable for DGS to contain relatively low amount of TN as compared to ADS and CFS (Zhong et al. 2014).

5.2.2. Variation of Total Carbon across Land Uses

Grazing within coastal forest ecosystems randomly done and so the current grazing system does not provide the redistribution of carbon and rapid increase in soil TC also reported by Kane (2015). The low amount of TC in ADS and DGS proves that most degraded and depleted soils of agro-ecosystems contain lower soil organic carbon pool than those under natural ecosystems, as is supported in Lal (2012). The low amount of TC in ADS as compared to that in any of the other LU practices indicates that farming activities are responsible for soil carbon reduction as supported in Syswerda et al. (2011); Kane (2015). In this view, farming accelerates soil heterotrophic activity and typically leads to between 50% to 70% carbon loss (Syswerda et al. 2011; Kane 2015). Indeed it is possible that low amount of carbon in ADS and DGS is limited by nutrients, predominantly nitrogen and phosphorus in addition to other environmental constraints (Wang et al. 2010).

Carbon depletion in the degraded coastal forest as represented by the Uzigua ecosystem contributed by clearing and burning vegetation for farm preparation. It reported that farming in this way has been a common practice for over 50 years since independence in 1960s. During all these years, crop production is characterized by conversion of forest into agricultural land without addition of mineral fertilizer or manure, so that the natural nutrients pool is depleted (Lal 2012). Although DGS showed a high amount of TC, yet it is below that found in CFS. This implies that grazing practices accelerated forest cover loss and hence affect the carbon sinks above and below ground (Syswerda et al. 2011).

The effect of livestock grazing in soil TC storage shows that herbivores may facilitate or depress TC deposition rates as compared to crop-agriculture and closed forests (He et al. 2011). The impacts of livestock grazing in TC show that there is a direct relationship between animal grazing activities above the ground and underground ecosystems as supported by Wardle et al. (2012). This relationship is explained by the difference in values of TC between CFS and DGS as supported by Golluscio et al. (2009); Britton, Pearce, and Jones (2005); Bai et al. (2012).

5.2.3. Variation of Phosphorus across Land Uses

Available phosphorus is among the important nutrients in any ecosystem as it plays roles in driving cellular energy cycles and building the molecules of DNA and RNA in plants (Buendia, Kleidon, and Porporato 2010). The difference for P across CFS, ADS and DGS imply that ecosystem disturbances cause positive or negative impacts on the availability of this nutrient (Block, Knoepp, and Fraterrigo 2012). The results showed low content of P in ADS and DGS, which is explained by the fact that conversion of forest land into

ADS and DGS reduces the amount of P because of the exposure of bare land to processes of soil runoff, erosion and percolation (Buendia, Kleidon, and Porporato 2010). However, Groppo et al. (2015); Schmitz et al. (2010) and Wardle et al. (2012) challenge the establishment of P decline in the livestock disturbed sites. This literature shows that livestock grazing supplements P by excretion and egestion processes contrary to the results in this work. Therefore, these findings suggest that forest disturbances affect above ground biomass and hence lower the amount of P in soils supporting the findings of Bai et al. (2012).

5.2.4. Carbon-Nitrogen Ratio across Land Uses

Carbon-nitrogen ratio, as an important factor for determination of the capability of soil and storage of carbon varied from CFS to ADS and DGS in the studies of UFR like in a study by Swangjang (2015). The variation in CN ratio is important in forest ecosystem health because carbon plays an important role in the energy content (carbohydrate) of plant species and production of CO_2 in soil ecosystem and nitrogen is essential for plant growth (Pausch and Kuzyakov 2012). This ratio plays a significant role in regulating soil organic matter mineralization (Swangjang 2015). Therefore, this ratio has implications in soil fertility.

The findings showed that soil CN ratio in the coastal forest decreased in the order CFS > ADS > DGS, possibly reflecting a higher degree of breakdown of humus stored in ADS and DGS as compared to CFS (Yang et al. 2010). However, these results contradict the trend in the CN ratio discovered by Zhang et al. 2016). The results portray that as breakdown of organic matter proceeds, those easily decomposed materials disappear and nitrogen immobilized in microbial biomass and decay products supporting (Yang et al. 2010). The process of breakdown and immobilization leaving behind more recalcitrant material characterized by slower decomposition rate because only fungi can break these materials (Zhang et al. 2016). These processes lowers the CN ratio in ADS and DGS than in CFS. The low CN ratio influences TN dynamics as it causes faster decomposition of organic matter and mineralization of nitrogen by microorganisms (Groppo et al. 2015). From these findings, the study states that the impacts of converting land from native forests to ADS and DGS have contributed to the degradation of microbial activities and thus forest disturbance considered to degrade the rate of organic matter, which is the main source of nutriments in soils (Zhang et al. 2016).

5.2.5. The Variation of TN, TC and P across Land Uses

The differences in soil TN, TC and P across CFS, ADS and P as presented in this study generally indicate that activities such as crop-agriculture and livestock grazing contribute to different alteration of nutrients depending on the differences in LU. The lower amount of TN and TC found in ADS and DGS as compared to CFS agrees with the findings in Groppo et al. (2015). However, the trend of P was contrary to Groppo et al. (2015). In this study, these three nutrients declined from CFS to ADS and DGS. From this trend, it is evident that human activities have a major contribution to variation in nutrient pools in forest ecosystems supporting the findings by Bai et al. (2012). Agriculture and livestock grazing activities affect the health status of soils by altering vegetation cover and the physical properties of soil (Eff, Eynolds, and Elnap 2005).

Lower content of the three nutrients in ADS and DGS across the study sites is the indicator that disturbed soils contained little organic matter because of inadequate vegetation life in the past years, which lead to a lack of humus and therefore low nutrient content (Abdu et al. 2010; Aubault et al. 2015). The differences in nutrients status between ADS and DGS show that, although all disturbances cause impacts in soil properties, there are some degrees of variation between the category of disturbances and any particular nutrient. For example, across all the nutrient states, DGS has the least amount of any of the nutrients except TC, which was higher than that in ADS. The differences in nutrients status across used to explain that any conversion of natural forest lands to artificial LU results into loss of nutrients such as TN and TC (Zhang et al. 2016).

5.2.6. Correlation of TN, TC and P across Land Uses

It was found a positive correlation between soil TN and TC in this study. Such correlation was significant especially in CAS and DGS than in other land uses. This relationship shows that TN variation goes hand in hand with TC spatially and quantitatively as supported by Groppo et al. (2015). These findings suggest that there are some degrees of TN decline in the same direction of TC in disturbed forest sites. These observations agree the existing documentations that loss of vegetation because of human activities such agriculture and livestock grazing affect bulk density, hence decomposition organic matter and mineralization of soils nutrients (Golluscio et al. 2009; Syswerda et al. 2011; Bai et al. 2012).

Such impacts contributed to the nature of variation correlation, which were obtained in ADS and DGS in this study. The study observed weak positive correlation between TN and TC in CFS, TN and P in CFS, TN and P in ADS, TN and P in DGS and, TC and P in DGS. These kinds of relationships show that variations existing between these elements in these land uses are partially independent. Weak to negative correlation between either TN and P or TC in CFS and ADS shows that TN or TC do not increase or decrease to the same direction in agreement with Bai et al. (2012). However, the weak correlation in variables is contrary with Block, Knoepp, and Fraterrigo (2012). This controversy could emanate from different ecological systems with differing climatic conditions such as temperatures and rainfall, which could affect TN, TC and P mineralization differently (Block, Knoepp, and Fraterrigo 2012).

In this work it is clearly established that the variation of nutrients across land uses provide useful information that clearing vegetation for various human activities contributes to physical losses of organic compounds from leaching and other processes that may alter the nutrient content of litter and returns to the soil and plants uptakes (Anzoni, Rofymow and Ackson 2010). The assumption that loss of vegetation above the ground influences soil fertility status supported by studies in Xuluc-Tolosaa, et al. (2003) as leaf litter aboveground is the main input of nutrients to the soil. Indeed, the amount of plant available nutrients affect natural and managed ecosystems largely (Anzoni et al. 2010). Therefore, processes that disturbs vegetation can also have a significant effect on nutrient cycles and nutrient limitation (Wang, Law and Pak 2010).

The variation of soil nutrients (TN, TC and P) content between closed forests sites compared to agriculture and livestock disturbed sites of Uzigua Forest Reserve are significant. Although these differences not directly defined as caused by forest disturbance, the findings used to establish relationships between the nutrient status of coastal forests and that of forest sites subjected to different forms of LU.

The established nutrient status partly suggests the interdependence of above and belowground component of forests. Crop-agriculture and livestock grazing affect aboveground biomass, which in turn affect the underground storage of nutrients. The study suggests that the restoration of aboveground (vegetation) and belowground (soil) forest ecosystems requires a holistic approach that requires sufficient knowledge about the interplay of these two components for sustainable forest management.

If crop-agriculture and livestock grazing are to be integral part of coastal forest management, it is suggested to carry further studies to devise crop-agriculture systems and grazing stocking rates that will sustain coastal forests. These activities should take place within limits, for example to take advantage of nutrient cycling between grazed animals, crop residues and forest ecosystems.

5.3. Soluble Bases across Disturbed and Intact Forests Sites

5.3.1. Variations of Soluble Bases across Land Uses

The tested hypotheses in this study show that there is significant variation of soluble bases, cation exchanges capacity and base saturation across forests sites subjected into different management practices. This variation supports the findings by other researchers that spatial nutrients variations contributed by land use management (Xia et al. 2015). From these findings, the researcher establish that those soluble bases vary because of different land uses and management supporting the findings by Yeshaneh (2015).

Across the study sites, agriculture and grazed disturbed sites have lost soluble bases, the conditions, which is associated with loss of vegetation (Lehmann et al. 2003). The effect of land use and management systems on soil fertility and chemical properties presented in this study is in agreement with some observations made by Pal et al. (2013) and Xia et al. (2015). The evidence that intact soil sites harbor higher bases than disturbed sites has been clearly observed in Ca, Mg and CEC where by these bases and CEC were high in CFS than in ADS and DGS unlike the K, Na and BS.

The variation shows that disturbances affect soluble bases differently across land uses. The significant differences of Ca and Mg in CFS and, DGS and ADS is a good indicator of impacts of disturbances on these two major soluble bases in the tropical coastal forests. The interpretation is that Ca and Mg highly get lost in disturbed than in the intact sites (Moreira and Fageria 2009).

Low amount of Ca and Mg in ADS than in DGS shows that converting land into farms and grazing use makes soil vulnerable to soil erosion and leaching and uptakes by crops (Vourlitis and Lobo, 2015; Johnson et al. 2008; Page and Mitchell 2008). Low amount of Ca and Mg in ADS and DGS partially shows that human activities in these land uses disturb nutrients through conversion of forests into other land uses. (Heineman et al. 2016) support low quantities of soluble bases in disturbed soils. Low quantity of soluble bases in disturbed sites indicate that cropped and grazed sites have lost nutrients. The loss is contributed by vegetation loss, whereby loss of vegetation have influenced soil chemical properties by influencing the distribution and concentrations of soluble bases unlike in the intact forest sites where trees and other vegetation contribute to increase exchangeable bases in the soils (Hobbie et al. 2005; Pal et al. 2013).

The study establish that low base elements in disturbed sites is partially explained by loss of vegetation or the removal of soil elements from the soil by crop harvests or livestock grazing and leaching (Page and Mitchell 2008). A combination of these three factors (i.e. clearing vegetation, crop harvests and grazing) affects the status of nutrients in the coastal forests; in turn, these factors affect forests ecosystems because of the interdependence between above and below ground forests ecosystem components. For example, variations of Ca, Mg, K and Na between CFS and the disturbed sites indicate that conversion of forests in other land uses results into release of nutrients locked in vegetation mainly in the form of woody (Heineman et al. 2016).

The wooden locked nutrients released into soils and animals where they are temporarily stored before getting lost (Moreira and Fageria 2009). Therefore, crop-agriculture and grazing disturb vegetation and litter hence soil nutrients in the tropics agreeing the results in Xia et al. (2015). Higher quantity of soluble bases in intact sites is a good indicator that undisturbed sites maintain nutrients circulation than the disturbed sites agreeing the findings of Quesada et al. (2012).

Indeed, the variation across soluble bases in response to disturbances shows that nutrients loss is not uniform throughout all soluble bases. For example, across the study, sites K and Na were low in all land uses compared to Ca and Mg. Low K and Na is the condition reported in the tropical forests because of the origin of the soils, high rainfall and high temperatures effects (Yawson et al. 2011).

These environmental factors when combined with crop-agriculture and livestock grazing pressure affect more K and Na in the tropics than other soluble bases (Yawson et al. 2011). Human activities accelerate the loss of K in the tropics, in turn low K affects carbohydrate and protein formation in forests trees. In this view, human activities cause K deficiency in forest ecosystems partly threatening the productivity of these forests (Gairola et al. 2012; Blanchet et al. 2017).

5.3.2. Correlation of Soluble Bases across Land Uses

Calcium had higher correlation with almost all other soluble especially in CFS. This correlation indicates that intact forest sites have the capacity to retain nutrients than disturbed sites in agreement with Moreira and Fageria (2009). Soluble bases such as Ca and Mg showed a positive and strong correlation across all the land uses except in ADS. The negatively correlation of Ca and Mg in ADS is in line with (Kabrick and Goyne, 2011).

The interplays of nutrients because of disturbances is used to indicate that certain activities accelerated loss of some nutrients. For example, the negative correlation of Ca and Mg in ADS than in any other land uses used to show that there are more declines in Mg than Ca in the disturbed forests sites supporting the findings in (Johnson et al. 2008). The main reason for high loss of Mg than Ca is that, the former base is vulnerable to leaching than the latter in disturbed sites (Sun et al. 2013). Because crop-agriculture and livestock grazing contribute to disturb forests sites by affecting vegetation and accelerating soil erosion and leaching, established that crop agriculture and livestock grazing contribute to loss of Mg than Ca through leaching (Hobbie et al. 2005; Kabrick and Goyne 2011). The positive and negative correlation findings on soluble bases in the intact forests and disturbed sites reported in (Kizza et al. 2013). Therefore, there is no uniformity in nutrients trends and dynamics other than variation across forests sites when exposed to different land use.

5.3.4. Soluble Bases, CEC and BS vs. Elevation Levels

The findings in this study show that base elements varied with elevation. There was significant variation for Mg in ADS, Ca, in ADS, CEC in ADS, and Na in CFS and BS in DGS. These variations show that Mg and Ca were low at high elevation (350 to 600m) across the study area meaning that the agricultural activities carried out at high elevations pose some potential risk for soluble bases depletion (Kabrick and Goyne 2011). There were more less variations of nutrients in CFS against elevation. A trend that can be associated with less leaching on nutrients in the CFS across different elevations. The variation of nutrients in DGS against elevation compared to other land uses was not significant. This little variation partially explained by that, grazed land contains some vegetation especially woods, which contribute to recycle soil nutrients, and partially returning these nutrients through animal feces (Lehmann et al. 2003; Khan et al. 2006; Heineman et al. 2016).

Although DGS had less variation of nutrients across the elevation, it is established that low nutrients availability at high elevation (350 to 600 m) contributed to limit vegetation growth, which upon grazing pressure; it results into loss of soil nutrients more than the lower bottoms. This limited supply of nutrients in-turn promotes nutrients insufficiency for wood production and thus livestock grazing continues to be among the factors affecting soluble bases in the study area like in other tropical forests (Vourlitis and Lobo 2015; Yeshaneh 2015).

5.3.5. Soluble Bases CEC, BS and UFR Sustainability

Although this study lacked baseline quantities of soluble bases, CEC and BS to make a comparison of whether the variation and quantities are sufficient or not to sustain UFR, still the current variation used to establish soluble bases and coastal forest. The available data on Ca and Mg, CEC and BS are useful in predicating sustainability of coastal forests relationships because these factors largely control forest ecosystems by affecting the distribution of plants in forests (Kabrick and Goyne 2011). Higher amount of Ca and Mg (for example) in CFS is a good indication that the uptake and recycling of these nutrients by trees and other vegetation is not in excess than the amount lost by leaching in disturbed soils (Kabrick and Goyne 2011). High amount of Ca and Mg in CFS is a good indicator that CFS health is promising because these two soluble bases are important in natural sustainability of forest ecosystems. In order that coastal forests maintain the forest capacity to retain nutrients, protection of forests and leaving trees regenerations must be in place.

It is important to protect vegetation in intact forests sites and restore disturbed sites to rejuvenate the lost nutrients and prevent further degradation of forests ecosystems. Protection and restoration must aim in improving the amount of soluble because these bases largely govern soil acidity and, consequently, plant species composition (Kabrick and Goyne 2011). Improvement on the composition of species in turn affects forest soils nutrients (Kabrick and Goyne 2011). These efforts will contribute into providing the function of runoff, soil and nutrient loss reduction and improve circulation of nutrients in coastal tropical forest (Kizza et al. 2013). A limitation in this study was that the interplays of nutrients variation on each not explored. Therefore, an understanding of how and reasons certain patterns of nutrients correlations opens another area for further research.

This study came up with a significant spatial chemical attributes variation of calcium, magnesium potassium as well as sodium, cation exchange capacity and base saturation across closed forest, crop agriculture and livestock grazing disturbances. These elements were significantly different among sites. From chemical variations of soluble bases as representative of soil chemical properties, it shows that disturbed forest sites have low nutrients than intact sites. These variations indicate that soils in the disturbed sites at high elevation ranging between 350 to 600m conserved for essential nutrients to maximize forest vegetation growth and development along the ecological gradient.

Improvements in forest ecosystems should not necessarily need addition of base elements, rather than avoiding further disturbances, protecting the intact sites and restoring disturbed sites by taking the advantage of natural forest capacities to recycle nutrients. It is suggested for further studies to identify soils, correlation of soil elements in the tropics in different land uses to establish a trend of nutrients at risk because of human activities mainly crop-agriculture and livestock grazing. Studies on forest nutrient mapping need continuous surveys to contribute into exploration of the above and below ground forest ecosystem nutrients pools and suggest possible remedies for sustainable coastal forests across different regions and landscapes in Tanzania.

5.4. Structure of Natural Forests in the Coastal Areas of Tanzania

5.4.1. Species Density across Land Uses

Like many other forests in the tropics, coastal forests are among the richest ecosystems in plant species (Devi and Yadava 2006). From the findings, there were significant differences in numbers and species composition across LU sites. These results confirmed the hypothesis that there are significant variations in stand parameters and diversity indices of regenerating trees between CFS and disturbed sites (ADS and DGS) at 5% level of significance. The variation of trees stand parameters and diversity indices across the plots indicates that tropical forests have a different natural ability to regenerate after disturbances and the exclusion of human activities (Sundarapandian and Swamy 2013).

The average number of trees recorded in this study was below that measured by Mligo (2015b). This difference is possible because many of the existing studies did not consider tree stand parameters and diversity values particularly from ADS and DGS seedlings and saplings. The variation plant density recorded in this study area shows that forests subjected to different disturbances regenerate differently as supported by Wekesa (2016). A larger number of seedlings and saplings in ADS than in CFS shows that disturbances have some beneficial effects, though in small spatial scales (Duah-gyamfi et al. 2014). Hessenmöller et al. (2013); Kalaba et al. (2013); Wekesa et al. (2016) and Lu et al. (2017) also support the beneficial effects of disturbances.

Also, it is possible to suggest that habitats modified by farming promote regeneration, and are measurable after some years of human activities exclusion (Kijazi et al. 2014; Navroodi 2015). The beneficial effects of disturbances observed in this study are contrary to other findings (Carnevale and Montagnini 2002; Hooper 2002; Kijazi et al. 2014). The dissimilarities in regeneration responses partially explained by the fact that different ecosystems respond differently to disturbances. Indeed, the recorded high population density of seedlings and saplings in ADS provides potential stock for future adult trees in these sites (Sundarapandian and Swamy 2013). Nevertheless, it is important to consider that not all regenerating trees will reach adult stages because they affected by multiple stresses (e.g. pest pressure, light, water and nutrient limitation).

In many cases, these stresses cause the gradual disappearance of some species in the process of growth and development (Majumdar and Datta 2014; Comita 2014; Amlin et al. 2014). The variations in this study across LU sites show that impacts of stresses in regeneration are neither equal, nor at the same rate (Golluscio et al. 2009; Cierjacks et al. 2008).

Livestock grazing sites had poorer seedlings and saplings density in comparison to ADS. This is explained by the fact that livestock grazing does not provide conducive microsites for regeneration. Poor regeneration in DGS is caused by livestock (e.g. cattle) compacting the soil, and consuming and trampling the regenerating trees (Cierjacks et al. 2008; Navroodi 2015). Livestock grazing sites incorporate seeds into deeper depths, which possibly hinders some seeds from germinating. On the other hand, cultivated sites allow the soil loosened and minimize the removal of seeds. Low seedling and sapling density in CFS indicates these sites have shifted from pioneer to non-pioneer communities (Duah-gyamfi et al. 2014).

The structure of forests in ADS and DGS are still under the former stage of succession, with the danger of losing many species because of further disturbance stresses (Comita 2014). Farming activities in ADS created microsites, which resulted in the rapid growth of seedlings. Thus, there was a more rapid recruitment and fast growth of pioneers in these sites than in DGS and CFS (Duah-gyamfi et al. 2014). Eludoyin (2016) supports high tree population density in ADS and DGS. It shows that seedlings and saplings contribute in larger proportions in many tropical forests than adult trees (Eludoyin 2016). However, there were some adult trees in ADS and DGS, which survived as remnants.

The main trees in these sites were *Afzelia quanzensis, Brachystergia spiciformis, Cynometra webberi,* and *Pterocarpus angolensis*. Interestingly, these trees passed into protected status by chance because their existence was not merely for protection or conservation values. Instead, they were kept to attain certain diameters and heights for timber production. Therefore, exclusion saved their lives, and thus they resumed their protected status and facilitated the parental stock roles (e.g. seed production) helping regeneration in ADS and DGS.

It is possible to find that the structure of forest developed from ADS and DGS after exclusion is dissimilar to the previously protected sites because human activities and natural disturbances affect the direction of forest structures (Bargali et al. 2013; Hessenmöller et al. 2013). The variation in forest stand structures and compositional settings for seedlings, saplings and adult trees reveled across LU sites indicates that human activities have either positive or negative impacts on the coastal forest ecosystems.

5.4.2. Species Diameter across Land Uses

The mean diameter of seedlings and saplings in ADS and DGS was below that found in CFS. In both LU sites, the average diameter was below that recorded in many tropical forests (Mligo 2015b). This variation in mean diameters between different studies can partially be justified by differing sample sizes, species compositions, and the age and degree of disturbances (Sundarapandian and Swamy 2013). In addition, a higher population density of seedlings and saplings than in CFS possibly causes the small diameters in ADS. It is mostly likely that a higher population creates a higher competition for the available resources, hence affecting the size of seedlings and saplings more than adult trees (Sundarapandian and Swamy 2013). The competition seems to be severe, especially when soil organic matter is a limiting factor as in the case of crop-agriculture and livestock grazing disturbed sites (Amlin et al. 2014).

5.4.3. Tree Basal Areas across Land Uses

The mean values of the basal areas showed a progressive increase from seedlings, to saplings, to adult trees similar in diameters across all the sites, as also noted by Shankar (2001). Therefore, BA was significantly different across LU sites. The BA in CFS was significantly greater as compared to ADS and DGS. The BA across all plots affected by differences in tree diameters. Tree species with large diameters contributed to the significant large BA especially in adult trees sub-category. The larger diameters of species within CFS, unlike in ADS and DGS, contributed to a greater mean value of BA in CFS. The BA variation in turn affected the volume of trees in all LU sites. The mean volume of trees increased from seedlings to adults in CFS unlike in ADS.

5.4.5. Species Volume across Land Uses

Volume variation between CFS and ADS indicates that, although ADS had a large number of young trees, their contribution to volume was less significant compared with a few adult trees in CFS. The large volume in DGS increased from seedlings to adult trees because this LU site contained the second largest number of adult trees. Across all LU sites, BA and heights at large affected the volume. That is why adult trees in CFS had a larger volume than seedlings and saplings (Mligo et al. (2009). The mean volume of adult trees obtained in this study is within the range reported by Mligo et al. (2009); Mligo (2015b). The mean volume for comparison with the existing studies mainly contributed by *Acacia brevispica, Combretum mole, Ficus stuhlmanii, Sclerocarya birrea, Sterculia abbreviate* and *Terminalia sambesiaca*, which were available across all LU sites more frequently than other species.

There were fewer contributions of species such as Cynometra *webberi, Dalbergia melanoxylone, Dalbergia nitidula, Dialium holtizii, Ficus sur, Hymenea verrucosa, Khaya anthotheca, Millicia excelsa, Millettia stuhlmannii, Pteleopsis myrtifolia, Pterocarpus angolensis* and *Pterocarpus rotundifolius* because these species are among the most overharvested for timber and construction poles.

5.4.6. Species Diversity across Land Uses

There were differences in species diversity across LU sites because species regenerate and exist depending on their differing abilities to survive in different environmental conditions (Bargali et al. 2013). The ADS and DGS showed a significantly lower variation in diversity indices, which is contrary to Hessenmöller et al. (2013). Low diversity in disturbed sites agrees with Guerrero and Bustamante (2007). The findings indicate that post-human disturbance regeneration differ from one LU site to another. It also shows that not all species have an equal capacity to regenerate (Jones et al. 2004).

These results confirm that human activities have modified UFR habitats, and thus the crop-agriculture and livestock grazing have imposed forest structural and diversity changes (Guerrero and Bustamante 2007). Low diversity in ADS and DGS partly explained by recognizing that some plants have a slower capacity to regenerate; hence, it is not possible to quantify their regeneration values three years after exclusion. However, my findings set a diversity baseline to describe regeneration potential in disturbed coastal forests (Devi and Yadava 2006; Duah-gyamfi et al. 2014). The findings also provide input for mapping future spatial and temporal coastal forest structures and diversity dynamics after exclusion. They also set forth the challenge that exclusion increases population density and/or trees diversity (Jones et al. 2004).

In regards to species richness, ADS and DGS had lower values compared with CFS sites contrary to Bargali et al. (2013). This contradiction possibly explained by the varying degrees of disturbance and kinds of species between different studies affecting the occurrence of certain species in a given location. However, low species richness used as a criterion to judge that Uzigua forest reserve should be counted among the degraded ecosystems along the coastal zone of Tanzania. This observation falls within the existing documentation that Tanzania coastal forests have lost at least 70 percent of their species (Howell et al. 2012).

The similarity index was below that in earlier studies (i.e. 5.06 and 5.40) for tropical forests (Devi and Yadava 2006). These differences are probably caused by variations in sampling methods, sample size, and measurements taken in the field, which in many cases have effects on results and comparisons (Jayakumar et al. 2011). Indeed, the deviation in similarity index obtained in this study is among the best confirmation that the Uzigua forest had been disturbed and degraded for the past 50 years. Human activity pressure has affected the biodiversity of coastal forests, even though some species have good potential to regenerate and thrive. For example, across all LU sites, Fabaceae (*Cordyla africana*, *Cynometra webberi*, *Dichrostachys cinerea*, *Erythrina abyssinica* and *Hymenea verrucosa*) affected similarity. These few species appeared more frequently than other species across LU sites. Therefore, disturbances seem to limit some species' ability to regenerate in different lands (Eales et al. 2016).

5.4.7. Important Values Index across Land Uses

The IVI was below that reported in Devi and Yadava (2006). The possible reasons for differences are that disturbance levels, geographical locations, and basal areas are different from one ecological system to another. However, some individual trees in CFS had high IVI just as in other tropical forests (Mligo 2015b). The IVI in the study sites show that disturbances have substantial impacts on different species. That is why trees such as *Tamarindus indica* in CFS had a value up to 20 %, while in ADS the highest value was only up to 14% for *Brachystegia boehmii, Combretum schumannii* and *Mimusops zeyheri*. In addition, DGS had up to 10% for *Combretum schumannii* and *Tamarindus indica*.

The IVI in the disturbed sites was below CFS possibly because of poor seed dispersal, competition, and low soil nutrient availability (Hooper et al. 2005). The IVI indicated that the impacts of disturbances affect individual species at different degrees, which in turn affect coastal forest structural settings and possibly affect some functions and services, agreeing with Bargali et al. (2013). However, the impacts of disturbances in services not covered in this study, thus opening another area for further investigation.

Crop-agriculture and livestock disturbed sites contributed to variability in numbers, basal area, volume, and species diversity and richness in the Uzigua Forest Reserve. Disturbed sites had differential successional regeneration consequences in trees' parameters and diversity. The differences in regeneration between and across land uses show that human disturbances in the coastal forests have positive or negative impacts. High population density of seedlings and saplings in the disturbed sites shows that the exclusion of human activities (agriculture and livestock grazing) enhances regeneration in quantity, but to a lesser extent in diversity. It shows that exclusion is a good management option as it permits natural and quick coastal forest recovery.

However, species diversity variances across land uses applied to comment that exclusion should take place even where the forest is already degraded. It is important that environmentalists, ecologists, foresters, livestock and agriculture practitioners are aware that there is a better regeneration response in crop-agriculture disturbed sites than in grazing lands. Therefore, attention be paid to the current overgrazing practices occurring in the coastal forests; otherwise, the ongoing pace of livestock grazing will continue to negatively affect forest ecosystems.

That said, it is possible that conclusions on species variations in crop-agriculture and livestock disturbed against closed forests sites three years after human activities exclusion provided only partial information. Therefore, further studies are required in the future to map coastal forest regeneration and dynamics for sustainable forest management.

5.5. Canonical Correlation between Vegetation and Soil Properties

5.5.1. Correlation between Stand and Soil Properties

The canonical correlation between sets of variables studied in this work reveals various outcomes. The significant canonical variation between above ground forest structure and soil properties across the study sites shows that tropical forests vary because of the floristic and environmental properties interactions (Nizam et al. 2013; Ichikogu 2014). The heterogeneity in correlation indicates that not all forest structures and diversity indices respond equally to soil parameters.

These results indicate that there are some direct and indirect relation between above and below ground forest ecosystems as documented in Wagg et al. (2014). From these findings, it is obvious that any disturbances on environment affect stand and soil physical properties. Indeed, these findings in this view supports Mbwambo et al. (2008) and Delang and Li (2013).

The ecological interpretation of the gradients represented by the canonical axes show that majority of plants positively correlated with soil properties supporting the findings in Ichikogu (2014). These results can be used to suggest that any alternation of soil physical properties in the tropical coastal forests affects species welfare, which in turn have influence on soil properties (i.e. bulk density , electric conductivity and soil texture in this work) in agreement with Joyi et al. (2015). From these findings, it can be predicted that any land use change, which affect the tree stand parameters has some impacts on soil nutrients (Martin et al. 2016; Guilherme et al. 2012). It is from this predicted and established reciprocal relations, the results revealed strong correlation of stand parameters in closed forest site than in the disturbed ones. Therefore, for proper management of coastal tropical forests, programs must consider the both below and above ground must consider ecosystems concurrently.

5.5.2. Correlation between Diversity and Soil Properties

There was positive correlation between diversity indices with soil chemical properties (soil nutrients) and soil physical properties as well as equitability and nutrients across land uses. These correlation values show that soil and above ground forest properties are characterized by the same dynamics directions in the coastal forests like in many other forest ecosystems (Gairola et al. 2012; Ichikogu 2014).

The positive correlation in Shannon index and soluble bases, Shannon and soil physical properties, equitability and soil physical properties, independent value index and soil physical properties are important in showing that each kind of forest diversity is affected by soil factors contrary to observations made in Nadeau and Sullivan (2015). This controversy is possibly resulting from variations in geographical locations and nature of vegetation. Regardless of this controversy, in this study, an establishment is that the relationships across soil properties and diversity indices can be used to indicate the direction of vegetation and soil interplays because vegetation influence the chemical and soil physical properties (Gairola et al. 2012).

The low correlations between trees stand parameters and soluble bases unlike that observed across carbon, nitrogen and phosphorus might be useful to predict that loss of vegetation affect more the non-soluble nutrients than soluble bases. For this prediction to qualify, it requires more studies because soil factors and or vegetation have some impacts on each other as documented in many tropical forests (Guilherme et al. 2012).

Interestingly, these variations can contribute into interpreting soil and diversity dynamics and complexity in agreement with Nizam et al. (2013) and Wagg et al. (2014). Conversely, the observation trees stand parameters had no significant correlation with soluble bases agree the results of Nadeau and Sullivan (2015). The implication of these findings in forest management is that some nutrients affected more than others did after disturbances. Moreover, it shows that different nutrients in different locale affected differently hence, production the variation of nutrients during and post disturbances requires temporally and spatially set assessments. Therefore it is hard to permanently establish nutrients status as supported in Huang et al. (2003) and Merganic et al. (2012). However, lack of correlation across tree density, heights, basal area and volume, and soluble bases should be considered with some precautions because tree growth in forests is highly influenced by elements such as Ca, Mg, K, Na concentration (Pal, Panwar and Bhardwaj 2013). Meaning that, any impacts on vegetation have impacts on soil soluble bases supporting Maynard et al. (2014).

Therefore, this study come up with the observation that more work needed to investigate the reasons for lack of correlation between tress stand parameters and some diversity indices (more specifically the equitability and independent value index) with soluble bases as were not discovered in this study. In this case, this studu fail to suggeetc the use of correlation between equaitability and simposns to explain and predict the interpplays between tropical coastal forests in relations to soluble bases status.

Generally, the correlations between vegetation and soil proerties established in this study indicate that disturbances cause changes on above ground species, which in turn have impacts on soil properties. The magsnitude of impacts nostly likely differe across a set of nutrients and prevailinglocale charactersitics. Therefore, the use the information on the relationship between above ground and soil properties to suggest management operations in forest is important but some precautions, which address a full range of above and below ground forests ecosystems welfare, are required. With this suggested remarks, certain parameters such higher Shannon-Weiner could be used as a good indicator of abundant regenerating vegetation in the disturbed sites after exclusion agreeing the results in (Gairola et al. 2012) unlike equitability or Simpsons index.

The canonical multivariate data analysis between forest structure (species variables) and soil properties (environmental variables) showed significant positive correlation across land uses The mean average shows that there is higher positive relationships in non-disturbed sites than the disturbed ones. The established correlations are the results of variations in forests ecosystem management, which bring forest disturbances emanating from crop-agriculture and livestock grazing. The correlations across tree stand parameters, diversity indices and soil properties established in this study set a ground, which are useful to make some predictions of forest structures and soil status dynamics in the tropical forest ecosystems. In addition, these correlations can also be used to inform foresters, environmentalists, agriculturists, livestock keepers and police makers that management efforts and plans of coastal forests must focus to address the below and above ground forests structures.

5.6. Socioeconomic Activities and ESV in Tanzania

This section narrates drivers for land cover and land use changes along the coastal ecosystems of Tanzania in consideration that human activities play a major role in altering the natural settings of biodiversity and ecosystems locally and globally (FRA 2018). Socioeconomic and demographic trends function as the major influencing factors on consumption patterns of the coastal zone in Tanzania. The socioeconomic activities influence land-use change and natural resource use, which in turn the changes affect the status of ecosystem services. More specifically, the drivers include lack of alternative sources of energy, population growth and lack of alternative livelihood activities. Other factors include overdependence on forest as the major income source of households, overdependence on agriculture and poor land use planning and implementation of the existing plans. Although the computed land use change did not show that each category has been depleted, yet there were significant land use changes. Drivers bring land cover and land use changes in turn these processes affect ecosystems and ecosystems service values as discussed in the following subsections.

5.6.1. Farming Activities and Changes in Shrub Land

The findings indicate that along the coastal zone of Tanzania, land cover and land use categories have changed over the past sixteen years. Significant gains in areas observed on farmland and shrub land as well as water body implies that the major driver of land LCLU change along the coastal zone is crop-agriculture. Obviously, crop farming that has increased in sixteen years is threatening the health status of the coastal zone, like many other Sub-Saharan African ecosystems, as supported by Scull et al. (2017) and Temesgen et al. (2018). The net increase in farmland indicates a substantial expansion of cultivated land for crop production. This expansion is supported by Ryan et al. (2016)

and Sonneveld, Keyzer and Ndiaye (2016), implying that farming is the persistent driver of land cover change globally.

Farmland expansion characterized mainly by small land holdings and commercial agriculture. Farmers produce paddy (*Orryza sativum*), maize (*Zea mays*), sesame (*Sesamum indicum*), cassava (*Mannihot esculentum*) and pineapples (*Ananas comosus*). Large-scale commercial agriculture is mainly for production of cashews (*Anacardium occidentale*), Sisal (*Agave* sp.), oranges (*Citrus* sp.) and mangoes (*Mangifera indica*). Peri-urban agriculture is the emerging practice mainly characterized by vegetable production. Production of these crops contribute to the conversion of natural vegetation into cultivated land and then into shrub land when the farms are abandoned.

Although the researcher did not compute the matrix of LCLU dynamics in order to locate areas of gains and losses (i.e. not indicated which LCLU category converted to which category and vice versa), the study can establish the interplay between farmland and shrub land expansion by using the existing literature (Foley et al. 2005). Thus, expansion of farmland related to shrub land development in agreement with Foley et al. (2005); Nkonya et al. (2013) and Rautiainen et al. (2016). Shrubs develop on the abandoned farmland because farmers practice shifting cultivation, especially with production of annual crops. The major reason for abandoning farms is that cropped land nutrients depleted since crop farmers across the coastal zone usually do not apply additional fertilizer. Shrubs development in disturbed sites supports the findings in Fetene et al. (2016).

As farms become infertile, they left for two, three or four years to regain fertility naturally. It is during the time of fallow that shrubs first overgrow on old farms before other vegetation as supported in Warinwa, Mwaura and Kiringe (2016). This succession stage implies that, it is possible to allow natural regeneration to take place on the degraded systems supporting findings in Rautiainen et al. (2016). Remarkably, shrubs are among the early successors of the disturbed farmland because they can survive in the degraded and nutrient poor soils (Kuenzer et al. 2011).

However, the correlating increase of shrubs with farmland expansion is contrary to the findings in Madriñán et al. (2012) and Wu et al. (2013). This controversial relationship implies that different geographical locations and climatic conditions permit different vegetation responses post land disturbances. Interestingly, it is possible that the abandoned farms are microsites to promote rejuvenation of forest species as supported by Rautiainen et al. (2016). The potential to rejuvenate is an important factor promoting many tropical ecosystems to succeed in regeneration (Bharathi and Prasad 2015).

5.6.2. Farming and Impacts on Wetlands and Waterbodies

Clearing land for crop cultivation affects vegetation and infiltration of water into the soil and ground water systems (Ryan et al. 2016). Expanded farming has impacts on wetlands and on increasing of water in open areas (mainly characterized as seasonal dams or floods) (Madriñán et al. 2012; Smith et al. 2014). However, the increase of waterbody in open or bare lands across the coastal zone is contrary to arguments of Blumstein and Thompson (2015). Yet, the study can establish that clearing land for any human purpose accelerates flooding in open areas as supported by Smail and Lewis (2009) and Ryan et al. (2016). This observation pronounced on the produced images and as well as supported by local communities and key informants that there is higher occurrence of floods on residential and farm area in the recent years.

Although there is no established matrix to map the direction of wetlands and waterbodies, the researcher used the existing literature to support the interplays between these two lands uses (Aighewi, Ishaque and Nosakhare 2014). For example, the shrinkage of wetland aggravates waterbodies accumulation (flooding in the open areas) because disturbances on wetlands reduces the capacity of wetlands to regulate flooding as supported by Chaudhary et al. (2017) and Temesgen et al. (2018). Coastal ecosystem management to protect wetlands and avoid further flooding along the coastal zone management should reexamine the danger of compromising the interplays between farming, loss of vegetation, wetlands and waterbody.

5.6.3. Population and Commercial Activities Dynamics

Gains or losses in LCLU categories such as farmland usage correlate well with human population changes along the coastal zone in agreement with Fetene et al. (2015) and Guerry et al. (2015). Population growth triggers changes on land cover whereby these changes occur parallel with higher conversion of large areas into housing and commercial activities (Blumstein and Thompson 2015). As in many tropical countries (Tanzania inclusive), population growth is also related to intensive and extensive clearing of land for crop agriculture as the major contributor to the households' income and economy (Foley et al. 2005; URT 2014; Sloan and Sayer 2015). Therefore, the findings confirm that there is a large-scale decline in forestland cover across the coastal zone of Tanzania like in many Sub-Saharan African areas. This decline indicates that population growth is significantly related to expansion of farming activities. In this view, the coastal zone is affected because there is a progressive transformation of land into cropland and built areas annually (Sonneveld, Keyzer and Ndiaye 2016; Scull et al. 2017).

Technically, expansion of some investments and human activities, for example, farmland does not necessarily mean increase in yield per unit area (Sonneveld, Keyzer and Ndiaye 2016). In some cases, land degradation is associated with declining yields in agreement with Sonneveld, Keyzer and Ndiaye (2016) and Borrelli et al. (2017). To compensate for the farmland nutrition deprivation, farmers opt to open new farms, consequently triggering further land degradation; this view is in agreement with Otsuka and Place (2014) findings. However, some findings do not support that population growth and urbanization promote the expansion of cultivated land (Madriñán et al. 2012; Quintas-Soriano et al. 2016).

This contradiction is useful to narrate that differences on the primary socioeconomic activities determine LCLU changes in a particular area. For example, the coastal zone of Tanzania influenced mainly by agricultural activities, while in highly developed coastal zones industrial activities or tourism dominates. Under these alternative livelihood activities, overdependence on farming activities to influence LCLU change is reduced (Quintas-Soriano et al. 2016). Therefore, identifying and promoting livelihood activities, which have minimum LCLU transformation, should be encouraged as a solution to protect and manage coastal ecosystems.

The reality is that, in a country or a zone where the major livelihood activity is crop agriculture, there is direct interplay between population growth and farmland expansion, which finally affects land cover.

5.6.4. Urbanization and LCLU Change

The urban environment has increased tremendously across the studied periods. Such an increase contributes in altering vegetation cover along the coastal ecosystems (Schmidt, Moore and Alber 2014; Blumstein and Thompson 2015; Yirsaw et al. 2017). The growth, expansion of urban, and exurban areas along the coastal zone of Tanzania are among the factors contributing to LCLU changes. Also, in these areas included are establishments of transportation systems and many other dispersed built-up sites supporting the findings in Maitima et al. (2009) and Temesgen et al. (2018).

The expansion of towns and other infrastructure correlates well with increased investments and developmental activities (Otsuka and Place 2014; Sloan and Sayer 2015). The major impacts of rapid expansion in urban settlements and commercial activities is the decline of forest areas and wetland shrinkage (Zhao et al. 2004; Zhang et al. 2015b; Warinwa, Mwaura and Kiringe 2016). Therefore, these findings are within the existing documentation that many tropical ecosystems suffer from urbanization supporting the conclusions of Wu et al. (2013), Xu et al. (2017) and Zhou et al. (2017). Some findings that support each other imply that locally and globally unplanned urbanization is a threat to coastal ecosystems (Zhou et al. 2017).

5.6.5. Grazing Land Use Change

Grazing land declined during the sixteen years. This land category was highly affected by livestock grazing pressure as also reported in other studies (Scull et al. 2017; and Temesgen et al. 2018). Records show an increase in the number of livestock mainly from Tanzania's inland towards the coastal zone (URT 2014). The major factors for the inland to coastal livestock movement include inadequate pasture and scarcity of water in other ecological zones emanating from land degradation and prolonged dry seasons in the inland areas of the country than the coastal zone. Indeed, the coastal zone is a livestock immigrant area because it harbors livestock fodder and has promising weather conditions, unlike many other inland zones (Maitima et al. 2009; UNEP 2015).

Moreover, livestock pressure on the coastal zone increased by the promising livestock market in Dar es Salaam city. This city is the major international market and is the prominent outlet of live animals and by-products. In addition, livestock keepers and livestock business people prefer either to keep domestic animals (mainly cattle) along the coastal zone permanently or temporally to capture market opportunities. Therefore, livestock-grazing pressure contributes to the decline of grazing land and deforestation in the coastal zone of Tanzania as reported in Fetene et al. (2016) and Maitima et al. (2009).

5.6.6. Wetland Change

Water draining activities contribute to the wetland decline along the coastal zone. These activities are mainly due to crop farming, commercial and residential development, and livestock grazing (Raburu and Kwena 2012). Modification of terrestrial ecosystems alters the ecohydrological processes as stated in Reeves and Champion (2004) and Duku et al. (2015). There is a clear relationship between shrinkage of wetland and farming activities (Duku et al. 2015). For example, crop production and livestock grazing take place within wetlands, hence creating the interference of the drainage systems (Duku et al. 2015). Farming activities and overgrazing in riparian areas reduce streamside vegetation and prevention of runoff while also lessening the wetlands filtration and recharge of water (Reeves and Champion 2004; Warinwa, Mwaura and Kiringe 2016).

Expansion of farms for crop production and livestock grazing in wetlands have intensified along the coastal zone in recent years because of rapid commercial investments (URT 2016; Zhang et al. 2015b). Moreover, some wetlands located in urban areas converted into built land and urban-agriculture contrary to observations made by Temesgen et al. (2018). This contradiction shows that the direction of wetland spatially and temporally change in the tropics. However, the findings suggest that wetlands in the coastal zone are on high pressure of degradation as reported in Aighewi, Ishaque and Nosakhare (2014). These wetlands need attention; otherwise, this land category will continue to shrink like other similar wetlands in the tropics (Chapungu and Hove 2013).

5.6.7. Forestland Change

Deforestation along the coastal zone produced by clearing land for farming activities, as well as developing settlements and infrastructure (Warinwa, Mwaura and Kiringe 2016). Activities such as collection of poles and timber for construction and harvesting of trees for fuel wood and charcoal prevail in the coastal zone and highly affect coastal forests like many forests globally (Keenan et al. 2015; Fetcne et al. 2015; Chaudhary et al. 2017).

Indeed, clearing land for salt extraction and sand mining identified by local community and key informants as other significant contributor to coastal forest loss. In addition, coastal aquaculture and livestock pressure threaten the coastal forests (Bryceson 2002; Luc van Hoof and Kraan 2017). In addition, there is higher community dependence on bioenergy at the expense of forest disturbances and degradation. Likewise, bioenergy dependence is prominent along the coastal zone because different households cannot afford to use gas or electricity for domestic cooking, heating and lighting (Temesgen and Wei 2018).

Furthermore, encroachment for crop cultivation into forests replaces natural vegetation with crops, consequently contributing to shrub invasion (Nkonya et al. 2013; Borrelli et al. 2017). All these factors are worsening the forest ecosystem's health along the coastal zone because of the net loss per unit time similar to many areas of Sub-Saharan Africa (Keenan et al. 2015; Temesgen et al. 2018).

In this view, farming activities affect forestland dynamics than other factors (Warinwa, Mwaura and Kiringe 2016). To overcome the ongoing forest loss, alternative livelihood activities to replace crop-agriculture needed. Otherwise, research on how agriculture can take place along the coastal zones without harming forests needed. These efforts will help to overcome the impacts of crop agriculture on the already disturbed ecosystem like many other tropical ecosystems (Temesgen et al. 2018).

5.6.8. Changes on H-ESV

Changes on LCLU have impacts on the total human welfare obtained from the ecosystem services. These changes have affected the total H-ESV too. Across sixteen years, H-ESV declined subject to population growth. This decline implies that as human population grow, human activities increase and aggravate some changes on LCLU categories, ESV and finally lead to the decline of the H-ESV supporting the findings in Fujita et al. (2013).

The loss in ESV and decline on H-ESV reported in this evaluation shows that if land use managements and the associated problems are not addressed properly, Tanzania will continue to lose benefits from ecosystem services, meaning that the current LCLU changes along the coastal zone need urgent attention. To address the current status of LCLU changes, ESV and H-ESV, efforts are required to slow down all processes which cause loss of each LCLU category, especially forest loss (Dale and Polasky 2007; Jantz and Manuel 2009; Han et al. 2016). Indeed, the advice is to allow natural regencration processes to take place along the coastal zone because ecosystems have the resilient capacity largely recover after disturbances (Rautiainen, Virtanen and Kauppi 2016).

5.6.9. Community Awareness on LCLU Changes

Across the study sites, local community and leaders showed that they are aware about the transformation of LCLU classes in the coastal zone. This observation supports the work by Chaudhary et al. (2017) that local communities are aware about changes occurring in their environments and drivers for changes. While agricultural activities frequently identified as the major drivers for LCLU changes, exploitation of forests resources for domestic bioenergy use is growing fast. This growth implies that overdependence on bioenergy is functioning as the current contributor of coastal forest loss, also globally reported in Jew et al. (2016) and Smith et al. (2014).

In recent years, urbanization related to higher demand of land for settlements and establishments of local investments in the expenses of harvesting of woods/trees for construction materials. Moreover, livestock grazing is becoming the threat to the coastal zone and is increasing at alarming pace. This trend automatically compromises the capacity of coastal ecosystems to offer ecological services for human wellbeing. Therefore, the decline of land cover because of land use changes along the coastal zone threatens the life of local communities because the livelihoods of the people depend directly on ecosystems 'services (Chaudhary et al. 2017).

Interestingly, uncontrolled fire is not significantly affecting coastal ecosystems in recent years. This implies that efforts have been in place to address fire problems in this zone as supported by group discussions and individual interviews. In this case, there are some improvements on some drivers of the coastal zone disturbances. Thus, efforts are needed to address the remaining disconcerting factors such as human population pressure, overdependence on natural coastal resources (because of lacking the alternative livelihood activities for income generation), and lack or poor implementation of land use plans supporting the findings of Luc van Hoof and Kraan (2017).

Indeed, it is important to consider that much work needed to manage and address at large the interplays between socioeconomic activities and ecosystems welfare along the coastal zone. Therefore, this study advises that there must be mechanisms that guide and give alternative livelihood activities as well as application of land use plans that safeguard coastal ecosystems as supported by Quintas-Soriano et al. (2016) and Chaudhary et al. (2017).

This study has highlighted the important links between LCLU change and impacts on ESV and H-ESV in the tropical coastal forests. The study conclude that land use changes in the coastal zone of Tanzania have transformed land cover to cultivated land, grazing lands, human settlements and other built area at the expense of natural vegetation mainly forest and grassland. The socioeconomic activities mainly crop farming, livestock grazing and harvesting of wood/trees as bioenergy sources all contribute to threaten the coastal ecosystems. These threats aggravated by a rapid population growth and expansion of socioeconomic activities along the coastal zone. Moreover, changes on LCLU have resulted into net loss of ESV and H-ESV.

Given the ongoing population pressure and socioeconomic activities in the coastal zone, it is likely that an increasing demand for land use will place heavy pressure on these ecosystems. Consequently, the capacity of coastal ecosystems to offer ecological functions and services to sustain life of human beings will be further impaired. Therefore, it is important to regulate and balance population and socioeconomic activities so that all changes that give net loss on LCLU, ESV and H-ESV circumvented.

5.7. Restoration Interventions of the Disturbed Sites

5.7.1. Methods for Restoration of Disturbed Sites

The findings confirm that socioeconomic and environmental factors have significant contribution to determine the status of the artificially planted and naturally regenerating trees across the study areas. In investments to restore coastal forests, efforts have been to promote artificial planting, which characterized by the use of a single species, or single applications. In supporting the tendency of a single species practice, local community and forest experts, proved that they are aware that many of the restoration methods focus on fast-growing tree species mainly for timber production and not biodiversity considerations agreeing the findings in Hjältén et al. (2017). These methods imply that restoration in this area put more focus to provide alternatives to forest resources but not addressing the entire ecological settings (Jacob et al. 2017).

It obvious that tree planting as a restoration techniques have ecological limitations as supported by Rayfield, Anand and Laurence (2005). In this view, ecological restoration is challenged because reforestation is done to restore the degraded areas to facilitate forests' provisions of only some goods and services contrary to the entire set of services that are expected from the naturally established systems (Lamb and Gilmour 2005).

5.7.2. Factors Affecting Restoration

While efforts are in place to promote artificial tree planting, yet this intervention hindered by factors such as poor extension services, lack of free and improved seedlings and seeds as well as inadequate land for planting trees. Local community emphasized that even if they are willing to plant trees; lack of improved materials is a challenge as supported by supported by Erickson et al. (2012).

The investigation shows that there are limited tree breeding projects to produce a lot of tree seeds and seedlings. This lack is affecting restoration programs as the consequences of using genetically improved forest plant material in most cases is important in improving the stand level because improved breeds grow faster with minimum impacts on the environmental health (Haapane et al. 2015). Lack of seedlings and seeds contribute into growing a few of tree species such as *Bambusa bambos, Dendrocalamus* sp. for conservation, construction and domestic utensils. *Acacia mangium, Melia azadirachta, Gmelina arborea* and *Khaya anthotheca* for production of timber, building poles and fuel woods. *Leucaena leucocephala* and *Senna siamea* for improving soil fertility while *Mangifera indica*, *Citruc* sp. and *Cocos nucifera* grown for production of fruits.

5.7.2.1. Climate Change and Restoration

Indeed, in line with the lack of improved breeds, is the issue of climatic condition (rainfall shortage). Persistent drought sets back the local community to plant trees because there is no possibilities for irrigation. Prolonged dry periods reported to characterize the coastal zone in recent years with pronounced effects on artificially planted trees than natural regenerating species. The restoration process challenged from the fact that not all species are capable to survive in the restoration process. This is because in the modified environmental attributes, only some species are capable of surviving and resisting to barriers as documented in (Cortines and Valcarcel 2009).

The best alternative could be harnessing the strength of natural species to succumb drought and produce species, which have the capacity to climatic adaptation (drought tolerant) (Haapane et al. 2015). Availability of genetically diverse and adapted seed and planting stock would help to overcome drought challenges and provide the foundation for healthy forests and ecosystems in the future as suggested in Erickson et al. (2012) and Haapane et al. (2015). Such trees could adapt to the changing climate by natural selection and in response to the current conditions (Erickson et al. 2012). Unfortunately, these materials are unavailable. Therefore, understanding the adaptive response of forest tree species to their local climates related to key adaptive traits such as survival, growth, and drought tolerance is opening a new area for further research. However, in this study it suggested that meeting forest restoration challenges relies on successful establishment of plant materials mainly seeds or seedlings (Dumroese et al. 2016).

5.7.2.2. Restoration and Enforcement

Furthermore, inadequate enforcement of laws and by-laws, and regulations hinders the process of planting trees in the study areas. Across the villages, the issues of legal compliance and punishments reported to be poorly implemented thus affecting restoration negatively. Thus local community gauge that law enforcements failed to encourage better practices, resulting in low involvement and lack of participation of local community in restoration practices as supported by Pinto et al. (2014). The findings in this work supports the observation made by Kaimowitz et al. (2003) that even though illegal forestry activities can be bad for rural livelihoods, so can enforcing the existing forestry laws, and doing it more effectively may make the problem even worse. In this case, forests activities such as tree planting and retention of natural trees based on households' willingness and not as abidance to the laws, by-laws or regulations.

5.7.2.3. Inadequate Knowledge on Restoration

In other cases, local community have no ideas on the types of restoration that can be implemented by defined laws that have been passed to support the conservation of biodiversity or to limit the spread of invasive species contrary to suggestion made in Beatty, Cox and Kuzee (2018). This controversy supported by the fact that the study area is characterized by weak forest extension systems as reported in (Kissinger, Herold and De Sy 2002; Durst 2009). Therefore, training and extension for smallholders, farmers, communities, small and medium-scale enterprises and forest officers improved in line with suggestion made in FAO (2010; 2015) and Franks et al. (2017). In addition, there must be efforts to strengthen the working-ship between local community and forest experts in activities such as forest species selection and planting (Beatty et al. 2018).

5.7.2.4. Exclusion as a Restoration Technique

In respect to conservation of the remaining trees and promotion of natural regeneration, the common practice has been exclusion of human activities. Exclusion used as a reactive measure in areas where disturbances and degradation have occurred. Exclusion of human activities are reported along the coastal zone as regeneration approach (passive restoration or natural recovery) where the only human intervention is to cease ongoing land uses such as grazing, agriculture or logging (Corbin and Holl (2012). Proactively, exclusion used to limit access and encroachments to forests ecosystems to promote natural regrowth or regeneration after ceasing the human activities (Lee et al. 2017; Virapongse 2017).

Interestingly, exclusion promotes local communities' adoption mechanisms to establish tree plantations on their farms. This adoption sets a non-natural forest dependence in some households although land ownerships continue to be a limiting factor. In this case, exclusion facilitates private forest ownership in line with the recommendation made by Lamb and Gilmour (2005). However, there has been variation in species response after exclusion across intact forest sites, agriculture disturbed and livestock grazed sites. This variation indicates that using only exclusion measures does not guarantee the sustainability of restoration. Thus, the variation show that disturbances create microsites, which determine the structure of new established forests consistently supported by previous findings in Kuuluvainen (2002) that all forest ecosystems characterized by disturbances.

Remarkably, the process of exclusion given time considerations to tolerate poor families living in forestlands claimed by the governments as clearly stated in Kaimowitz et al. (2003). Remarkably, some warning signs reported by local community that exclusion if not planned well, considered lethal. Because it contributes to transformation of disturbances and degradation to other forestlands. For example, the study found that farmers opted to open new farms elsewhere for crop production and livestock keeping. Excluded families had to find new sites for construction of infrastructure such as houses, schools, local roads etc. These new establishments resulted into the so-called "reallocated disturbances" to new areas. These transformations might have caused loss of forestland cover at the newly established settlements and farms. However, this study did not map the impacts of exclusion at the sites where the excluded community re-settled. Therefore, the anticipated impacts need investigation by another study beyond the coverage of this presented work.

In addition, exclusion affects the livelihood activities because people are supposed to terminate the long-term established life dependence on forests ecosystems, which is equal to loss of employment opportunities (Kaimowitz et al. 2003). The impact of exclusion might be large than the usually reported ones, because local communities are largely connected to forests even if the exactly number is unknown especially in the informal forestry activities (Kaimowitz et al. 2003). In addition, vulnerability is a result of the fact that different household socioeconomic characteristics, total household income, in particular found to be important determinants of the level of household reliance on forest resources (Widianingsih, Theilade and Pouliot 2016).

Therefore, is important to implement exclusion practices under good management setup because inadequate resource management capacity and poor governance are some of the problems that negatively affect natural resources. These impacts and the rural poor who depend on them (World Wildlife Fund (WWF) 2014). Notably, whether tree planting or retention of the naturally existing species, the purpose for restoration seems not well understood at local community level. Clearly, local community investments on restoration target on fuel wood and construction materials such as timber and for non-timber forest products. That is why the number of trees planted and/or reserved on the individual household land ownerships during the entire residential duration is very low. The ecological value of restoration left under the central or local government authorities and donor organizations. That means to achieve dual goals of restoration is ambiguous and fraught with contradictions (VonHedemann and Osborne 2016). Lacking a fully understanding of socioecological horizons of restoration is jeopardizing forest management efforts because increased understanding of forest disturbance and restoration is important in ensuring biodiversity conservation (Kuuluvainen 2002).

5.7.2.5. Awareness on the Aim of Restorations

Poor awareness about the aim of restoration and expected ecological value of forest, contribute the local community to get involved into destructive activities of forests ecosystems. This observation is contrary to the emphasize that forest management should be a local community-centered activity to ensures that biological resources are positively employed to help secure sustainable and desirable livelihoods (Lamb and Gilmour 2005). Village environmental committees used to improve community integration in community based forest restoration programs. These committees were useful in promoting restoration and ensuring community by-laws compliance as reported across the five surveys villages. The use of representative environmental committees in planning for restoration programs reported across the villages. However, these committees have no incentives (in kind or monetary) and facilities (e.g. transport) to perform their duties at fully capacity. As a result, they work under voluntary systems, which sometimes undermine their performance. The only promising incentives was a dividend from harvesting licenses levy, which is not covering the whole year. Therefore, it is important to devise mechanisms to improve working conditions of the committee members such as providing incentives and transport facilities.

5.7.2.6. Restoration Responsibilities

Nevertheless, majority of the locals consider that restoration responsibility is mainly resting on the hands of the committees, government or non-governmental authorities. As a result, village committees in some cases receives inadequate cooperation from majority of the local community. This poor cooperation across local communities undermines the advancements towards general community-based resource governance reported succcssful in many other ecological zones. Locals' participation in restoration programs is somewhat low agreeing the findings in Lee et al. (2017). Thus, to achieve a promising restoration and management of forests, there is a need to strengthen new communication strategies on restoration roles across different stakeholders as stated in Pinto et al. (2014) and Virapongse (2017).

5.7.2.7. Overdependence on Forest Resources

Another hindrance is that restoration interventions coincide with poor rural populations whose livelihoods depend on forests (Jacob et al. 2017). In these rural settings, people depend directly or indirectly on forest ecosystem services (e.g. for water, timber, and non-timber forest products) agreeing with the findings in Widianingsih, Theilade and Pouliot (2016).This dependence continue to create pressure rather than solutions to forest disturbances.

5.7.2.8. Motivation for Restoration

Excitingly, overdependence on forest resources shows that local communities define access to forest for timber and fuel woods as some kinds of incentives. Thus, access to benefits accrued from forests provide motivation for restoration in agreement with VonHedemann and Osborne (2016). In most cases local community continue to regard permitted harvesting of trees (for timber of charcoal) by licensing, motivates the participation in restoration programs. Even if these benefits are unquantified to present payments for ecosystem services, indirectly, they increase forest protection in agreement with VonHedemann and Osborne (2016). In this case, restoration interventions should not only focus on ecological roles but also the importance for human livelihoods and well-being (Widianingsih, Theilade and Pouliot 2016).

5.7.3. Suggestions to Improve Restoration Mechanisms

This study gives a general suggestion that there must mechanisms to ensure integrated social, environmental and economic landscapes, and emphasize on the production of multiple benefits from forests and participatory engagement of stakeholders in restoration as suggested by Virapongse (2017). Any compromise across livelihood, human-wellbeing and ecological restoration might hinder the entire process of restoration because people intensively depend on natural resources (Kärvemo et al. 2017) for their survival and economic purpose.

In ecological perspectives, the study discovered that there is no mechanisms to address factors, which hinder natural forest recovery and site requirements. There no processes for identifying and removing factors that inhibit dispersal and or establishments. Thus factors such as harsh microclimates, high seed predation, and competition with aggressive shrubs or grasses. Neglecting to consider these factors hinders the artificially and natural forest to regeneration and recovery of the disturbed sites. In addition, across the villages, restoration programs are not in the position to address the challenge related to invasions of alien species in the regenerating disturbed sites.

The issue of ecological considerations to fear invasion of non-native trees is often neglected (Jacob et al. 2017). Invasive species in response to forest disturbances and restoration is the area that demand further studies along the coastal zone of Tanzania as supported by Prach et al. (2014). Thus, this study strongly concur with Hartter et al. (2011) that an understanding of the structure and function of restoration interventions of tropical forest should be a routine. Since direct field observation found localized and non-uniform disturbed sites within and outside Uzigua Forest Reserve, it would be an ecological insight to plan for restoration of coastal forest by addressing not only large ecosystems but also the small fragments. These small fragments be under considerations because they might constitute a large fraction of seed banks and remnant forest species (Ribeiro et al. 2009). Therefore, the ecological response and functions of trees in the restored areas need further surveys.

The general evaluation of the restoration interventions in this study shows that restoration is complex and require a complex designs with multidimensional reestablishments that employ a combination of methods such as improving extension services, seed and seedlings availability, site preparation and control of non-native species to provide a more rapid and effective intervention strategies. In this view, it is important to gauge the interplays of restoration and efficacy of ecosystems in providing functions and services after restoration interventions. Therefore, holistic planning needed to address the social-ecological interplays along the coastal zones.

CHAPTER SIX:THESIS CONCLUSIONS AND RECOMMENDATIONS

6.1. Overview of Findings and Future Directions

6.1.1. Soil Chemical and Physical Properties in the coastal areas

Soil disturbances as defined in this study means any physical, biological, or chemical alteration of the soil caused by forestry operations (Curran et al. 2003). Human disturbance by activities such as plowing and logging greatly affect soil properties (Chen and Li 2003). Forest disturbance for example, strongly affects soil characteristics mainly soil volume, chemistry and texture. Impacts consequences are soil degradation, soil erosion and the destruction of species, biomass and biodiversity loss (Chu and Guo 2013). Forest disturbances start to affect species composition, which in turns affects soils nutrients (FAO, 2009). The impacts of vegetation destruction in soils nutrients pools is that, different plant species have different nutrient requirements and returns to soils (Chen and Li 2003). Disturbance in forest affects the ecological relationship between forest vegetation and forest soils (Caveliere et al. 1999; Brosofske, Chen and Crow, 2001; Eni, Iwara and Offiong 2012).

Human activities especially those involving clearance of forests vegetation pose soil to erosion, loss of organic matter and other necessary elements useful for vegetation growth. For example, a study by Chen and Li (2003) shows that soil nitrogen of different ecosystems is concentrated mostly at the top 10 cm depth hence any effects on this layer would affect soil nutrients in these ecosystems.

This effect supported in Chen and Li (2003) that soil nutrients such as phosphorus differences in different soil horizons may result from change of biological and geochemical processes at different depths after disturbances. However, in the same study, soil potassium was slightly higher in the disturbed than primary intact forest sites. Therefore, the findings in this work supports many existing literatures that disturbances on vegetation component of the ecosystems affects soil fertility.

6.1.2. Forest Structure under the Disturbed Coastal Forests Areas

In regards to the structure of the natural forests along the coastal zone of Tanzania, the examined tree structure and regeneration in the disturbed sites and intact forest sites is important in informing forest managers and land users that human induced disturbances bring forests structures and biodiversity dynamics (Averill et al. 1994; Kimaro and Lulandala 2013). The directions of forests in disturbed sites is not uniform because some disturbances are ecologically proved to be essential processes, necessary at some level of intensity and periodicity for the long-term sustainability and productivity of most, if not all forestry ecosystems (Averill et al. 1994). In this chapter, it shows that farming activities and livestock grazing disturbances have modified the coastal habitats. These modifications in turn provided favorable germinating conditions of some species especially in crop-agriculture disturbed sites than grazed sites (Kalaba et al. 2013). In this case, crop-agricultural disturbance is beneficial to vegetation regeneration. Indeed, the remarkable negative effects of disturbances as recorded in grazed sites suggest that human-related activities cause or exacerbate loss of capacity for species to regenerate (Averill et al. 1994).

The information on the structural attributes of forest stand presented in this work are important to foresters, ecologists and environmental mangers of coastal ecosystems. The information on forest disturbance and soil health expected to contribute in understanding and managing coastal ecosystems, more specifically forest ecosystems (Kijazi et al. 2014). This information is important in predicting the function of forest ecosystems because forest functioning is determined by taxonomic attributes of the vegetation and structural attributes and by the direct and indirect effects of environmental drivers including human disturbances (Poorter et al. 2015). Any activity whether natural or artificial, if it affects forest structural components, it results into affecting forests ecosystems functions and services.

6.1.3. Correlation between Vegetation and Soil Properties

In this piece of work, the relationships between forest disturbances and soil properties established. The section show that disturbances contribute to affect taxonomic, structural and soil attributes in forests ecosystems. Some variations recorded and expected to changes because the effects of disturbance can be short or long-term occurring over decades or centuries (Poorter et al. 2015). The variation across forests sites shows that forest disturbance affects an integral relationship between vegetation and soils (Eni, Iwara and Offiong 2012). This relationship is vital because soil gives vital support such as provision of moisture, nutrient and anchorage to vegetation while vegetation provides protective cover and nutrient maintenance. Disturbance can cause the elimination of species and result into major changes in species composition (Averill et al. 1994). In this study, it established that some disturbances bring changes in succession pathways, which are beneficial to maintain energy flow, nutrient cycling, species, genetic and structural diversity.

6.1.4. Socioeconomic Activities and Ecosystems Services

In this section, I examined land cover and land use change across the coastal zone of Tanzania. This work also identified the major activities contributing to disturb and degrade coastal forest ecosystem and other ecosystems as well as the impacts of land cover and land use changes on ecosystem services. The information on the interplays between land cover and land uses are important in informing ecosystems users and managers that the ongoing use of land along the coastal zone have impacts on the coastal ecosystems health (Luc van Hoof and Kraan 2017).

The results in this study show that human activities mainly crop- agriculture, livestock grazing, settlements, and urbanization cause impacts on the environment. What is happening along the coastal zone of Tanzania is an evidence that coastal ecosystems mainly forests are highly threatened locally and globally (Bryceson 2002). The degradation of coastal ecosystems mainly brought by rapid population pressure along the coastal zone (Szuster, Chen and Borger 2011).

The escalating demand from nature to meet rapid human population needs, have resulted into transformation on the patterns of interaction between ecosystems services and humans (Devisscher 2010). Indeed, in many cases disturbances cause disruption of ecosystem services, interruption of socioeconomic expectations (goods, services) and species replacement (Cork and Shelton 2000).

6.1.5. Restoration Interventions of the Disturbed Coastal Areas

In this section, I have presented that tree planting and retaining some naturally growing species focus to obtain forest resources to meet socioeconomic and livelihood activities and demands. Production of trees for construction materials and fuel woods supply drive the two restoration practices. Indirectly, these two interventions supports to protect maintain and restore the disturbed coastal areas (Waite et al. 2014).

The identified restoration approaches emulates human forest disturbances and land use drivers to enhance coastal forest ecosystems function and services as pointed out in (Turner, Donato and Romme 2013; Martinuzzi et al. 2015). However, restorations interventions face multiple factors, which range from socioeconomic to environmental bases. Because of multiple restoration set-backings, coastal ecosystem restoration as alternatives to protect coastal communities is in jeopardy. Therefore, the overall strategy should be to sustainably manage, conserve and restore coastal forest ecosystems to provide services to the people while maintaining the ecological integrity. To meet the socioeconomic and ecological demand, restorations approaches should be supported by making seed and seedlings available for establishment of tree distribution and woodlot planting programs, and subsequent use of the wood, or facilitation (under licensing schemes) of access to fuel wood stocks in protected areas in line with suggestion made in Shackleton et al. (2008).

6.2. Conclusions

This study gives a conclusion that forest disturbances in the tropical coastal forests are real. The impacts of these disturbances have a wide spread from soil, to forest stand and biodiversity structure and ecosystem services. The interplays across these components requires a well understood in planning for conservation and protection of coastal ecosystems. Any conservation measure must consider positive and negative impacts of socioeconomic drivers including population changes, land cover and land use dynamics, ecosystem service values, socioeconomic pressure mainly crop-agriculture and livestock grazing in the coastal forests. It is important that the positive roles of disturbances on species diversity and regeneration of tropical forests harnessed, promoted and considered in management plans and actions. Conservation efforts should focus on taking the advantage of disturbances mainly those emanating from crop-agriculture and livestock grazing in the coastal ecosystems especially where total exclusion of these two activities is not possible.

Moreover, to control the ongoing loss of coastal forests ecosystems, it is important to establish well-coordinated management approaches, which involves different sectors and socioeconomic activities along the coastal zone. Awareness raising and involvement of the public, and especially the communities of people living in the surrounding coastal forests areas, is of the utmost importance for the conservation and restoration of coastal forests. Initiatives such as education and campaigns on adjusting human activities and alternative livelihood activities promoted and demonstrated to reduce total forests dependence life. Further, education needed to encourage the local population to contribute into protection and restoration of tropical coastal forests.

6.3. Study Limitations and Areas for Further Studies

6.3.1. Study Limitations

The environmental conditions of the study sites at the beginning of disturbances and history of land cover and land use category changes were usually difficult to know with accuracy. This resulted in a risk of attributing wrongly to an effect of impacts of crop-agriculture and livestock grazing differences that are simply due to current variations amongst sites.

The estimation of temporal and spatial scale of land cover, land use changes and forest disturbances to establish a strong foundation for comparison across years was difficult to get and hence this study established a baseline in tropical coastal forests for future comparisons.

Moreover, there are some possible confusion between site conditions and time of exclusion. For example, agricultural activities are likely to be abandoned in less fertile sites earlier than in more fertile ones, livestock grazing is not uniformly distributed in a defined spatial and temporal design. Therefore, it is possible that this study has not provided the real situation in habitat variability in the initial stages of trees and shrubs regenerations especially after human activities exclusion.

Finally yet importantly, time constraints have imposed some trade-off in methodological choices. For example, the study would have collected panel data to improve statistical power. However, the necessity to sample a large number of samples for difference seasons and years from each land cover and land use prevented by time factor. However, these limitations did not affect the overall work as the researcher maximized the use of available resources and limited time by conducting intensive field surveys.

6.3.2. Areas for Further Studies

Besides the research plans mentioned in the previous chapters, this PhD research supports the need for further studies on several topics:

There is a general need to increase research on the soil and plant ecology of tropical coastal forests to get a better understanding on how tropical coastal forests' soils and forest structure differ from other tropical forests in response to different disturbances and management.

As mentioned earlier, several factors are likely to affect the interplays across the disturbed forests sites and the structure, functions and services of tropical coastal forests. The disturbance factors need further testing. Included in these tests should be studies on landscape changes and impacts on ecosystem services to establish the past, present, predict the future land cover, and land use changes.

It suggested assessing the relative carrying capacity of tropical coastal forest ecosystems to accommodate crop-agriculture and livestock keeping activities if these two activities cannot get a total exclusion in the management of tropical coastal forests. There must be efforts to discover and set the stocking rate to the management of grazed ecosystems. The research must produce the basic relationships between stocking rate and trees production, animal production, and species composition of vegetation communities.

Further studies are required to identify the mechanisms through which established trees influence regeneration in the tropical coastal forests. To identify the establishment of land cover and land use strategies, which promote species (in terms of the potential trade-off between growth and survival) and to understand the link between lands uses and function of coastal forests by considering variation between and within species and land uses.

Again, studies are suggested be conducted to explore the relationships between soil factors and spatial distribution of vegetation along the tropical coastal forest ecological gradient and establish the differences in their effects of soil heterogeneity on the structure of coastal forests and functional values.

In areas where the natural regeneration potential is poor, it is important to facilitate artificial regeneration. For successful artificial regeneration potential, this study suggest to conduct research, which aim to identify fast- growing, aggressive trees and shrubs, which will be appropriate for agro-forests in areas experiencing severe trees/shrub shortage. Such research developments must take into consideration that some species under some environmental conditions have the potential to be invasive and to spread to areas where they are not wanted. Therefore, any research findings need the introduction with care and the performance call for carefully monitoring.

References

[1] Abdu, A., Abd Kadir, M. Z., Ibrahim, Z., Hamzah, M. Z., Abdul Hamid, H., Hassan, A., and Majid, N. M. 2010. Properties of Soils in the Rehabilitated Degraded Tropical Lowland and Hill Dipterocarp Forests in Peninsular Malaysia. In 19th World Congress of Soil Science, Soil Solutions for a Changing World Brisbane. 23–26.

[2] Afrifa, A. B., Antwi-Boasiako, K., and Saaniano, Y. 2016. Natural Regeneration of Indigenous Tree Species in a Mixed Stand of Exotic Tree Species at a Reclaimed Mine Site in Ghana. *International Journal of Research in Agriculture and Forestry*, 3 (1): 36–41.

[3] Aighewi, I., Ishueaq, A., and Nosakhare, O. 2014. Geospatial Evaluation for Ecological Watershed Management II: Changes in Land Use-Land Cover Influence on Ecosystems Services. *Journal of Geographic Information System*, 6: 246–257

[4] Agherkakli, B., Najafi, A., and Sadeghi, S. H. 2010. Ground Based Operation Effects on Soil Disturbance by Steel Tracked Skidder in a Steep Slope of Forest. *Journal of Forest Science*, 2010 (6): 278–284.

[5] Ahmed, E., Ali, S., and Addisu, A. 2017. Structure and Regeneration Status of Woody Plants in the Hallideghie Wildlife Reserve, North East Ethiopia. *International Journal of Biodiversity and Conservation*, 9 (6): 200–211.

[6] Ahrends, A. 2005. *Patterns of Degradation in Lowland Coastal Forests in Coast Region;* A Thesis Submitted to the University of Greifswald in Candidacy for the Degree of "Diploma" (Master of Science German Equivalent). Ernst-Moritz-Arndt University of Greifswald. p.150.

[7] Alexander, A. 2012. Soil Compaction on Skid Trails after Selective Logging in Moist Evergreen Forest of Ghana. *Agriculture and Biology Journal of North America*, 3 (6): 262–264.

[8] Alexandre, Filho, R., Adams, C., Sergio, R., and Murrieta, S. 2013. The Impacts of Shifting Cultivation on Tropical Forest Soil: A Review. *Boletim Do Museu Paraense Emilio Goeldi:Ciencias Humanas* (90): 693–727.

[9] Al-hamdan, M. Z., Rickman, D. L., and Ye, J. 2012. Relationship Between Watershed Land-Cover/Land-Use Change and Water Turbidity Status of Tampa Bay Major Tributaries, Florida,USA. *Water Air Soil Pollution*. p.17.

[10] Amlin, G., Suratman, M. N., and Isa, N. M. 2014. Soil Chemical Analysis of Secondary Forest 30 Years after Logging Activities at Krau Wildlife Reserve, Pahang, Malaysia. *APCBEE Procedia*, 9: 75–81.

[11]Anna, C. 2009. The Forest, the Fire and the Fungi: Studying the Effects of Prescribed Burning on Mycorrhizal Fungi in Crater Lake National Park. *JFSP Briefs*. 61 (38):1-6.

[12] Aref, I. M., Atta, H. A., and Ghamde, A. R. M. 2011. Effect of Forest Fires on Tree Diversity and some Soil Properties. *International Journal of Agriculture and Biology*, 13 (5): 659–664.

[13] Armstrong, A. H., Shugart, H. H., and Temilola, E. 2011. Characterization of Community Composition and Forest Structure in a Madagascar Lowland Rainforest. *Tropical Conservation Science*, 4 (4): 428–444.

[14] Attiwill, P. M. 1994. The Disturbance of Forest Ecosystems: The Ecological Basis for Conservative Management. *Forest Ecology and Management*, 63 (2–3): 247–300.

[15] Amato, A. W. D., Fraver, S., Palik, B. J., Bradford, J. B., and Patty, L. 2011 Singular and Interactive Effects of Blowdown, Salvage Logging and Wildfire in Sub-Boreal Pine Systems. *Forest Ecology and Management*, 262 (11): 2070–2078.

[16] Anon. 2007. *West Africa Regional Network Meeting on Earth Observation and Environmental Change GOFC-GOLD Workshop on Requirements for Fire Early Warning Systems in Africa*. Accra, Ghana.p.4.

[17]Anyanwu J. C., Egbuche C. T., Amaku. G. E., Duruora J. O., Onwuagba, S. M. 2015. The Impact of Deforestation on Soil Conditions in Anambra State of Nigeria. *Agriculture, Forestry and Fisheries*. Special Issue: Environment and Applied Science Management in a Changing Global Climate. 4 (3-1) 64-69.

[18] Ares, A., Terry, T., Harrington, C., Devine, W., and Peter, D. 2007. Biomass Removal, Soil Compaction, and Vegetation Control Effects on Five-Year Growth. *Society of American Forests*, 53 (5): 600-610.

[19] Aubault, H., Webb, N. P., Strong, C. L., McTainsh, G. H., Leys, J. F., and Scanlan,

J. C. 2015. Grazing Impacts on the Susceptibility of Rangelands to Wind Erosion: The Effects of Stocking Rate, Stocking Strategy and Land Condition. *Aeolian Research*, (17): 89–99.

[20] Averill, R. D., Larson, L., Kaveland, J., Wargo, P. M., Williams, J., and Bellinger, M. 1994. *Disturbance Processes and Ecosystem Management*. Washington DC, USA. p.1-33.

[21] Axelsson, C., Jones, S., Haywood, A., Suarez, L., Wilkes, P., Woodgate, W., ... Mellora, A. 2012. Key Attributes for Monitoring and Assessment of Australian Forests: A Land Management Perspective. *CEUR Workshop Proceedings*, 1328. p. 9.

[22] Ayers, M. P., and Lombardero, M. J. 2000. Assessing the Consequences of Global Change for Forest Disturbance from Herbivores and Pathogens. *The Science of the Total Environment*, (262): 263–286.

[23] Baboo, B., Sagar, R., Bargali, S. S., and Verma, H. 2017. Tree Species Composition, Regeneration and Diversity of an Indian Dry Tropical Forest Protected Area. *Tropical Ecology*, 58 (2): 409–423.

[24] Backéus, I., Pettersson, B., Strömquist, L.,and Ruffo, C. 2006. Tree Communities and Structural Dynamics in Miombo (*Brachystegia-julbernardia*) Woodland, Tanzania. *Forest Ecology and Management*, 230 (1–3): 171–178.

[25] Bahamondez, C., and Thompson, I. D. 2016. Determining Forest Degradation, Ecosystem State and Resilience Using a Standard Stand Stocking Measurement Diagram: Theory into Practice. *Forestry*, 89 (3): 290–300.

[26] Bahrami, A., Emadodin, I., Atashi, M. R., and Bork, H. R. 2010. Land-Use Change and Soil Degradation : A Case Study , North of Iran Department of Soil Science , Tarbiat Modares University , Tehran , Iran . *Agriculture and Biology Journal of North America*, 1(4): 600–605.

[27] Bai, X., Queenborough, S. A., Wang, X., Zhang, J., Li, B., Yuan, Z., ... Hao, Z. 2012. Effects of Local Biotic Neighbors and Habitat Heterogeneity on Tree and Shrub Seedling Survival in an Old-Growth Temperate Forest. *Oecologia*, 170 (3): 755–765.

[28] Banda, T., Schwartz, M. W., and Caro, T. 2006. Woody Vegetation Structure and

Composition along a Protection Gradient in a Miombo Ecosystem of Western Tanzania. *Forest Ecology and Management*, 230 (1–3): 179–185.

[29] Bargali, K., Bisht, P., Khan, A., and Rawat, Y. S. 2013. Diversity and Regeneration Status of Tree Species at Nainital Catchment, Uttarakhand, India. *International Journal of Biodiversity and Conservation*, 5: 270–280.

[30] Barreto, M. B., Lo Monaco, S., Diaz, R., Barreto-Pittol, E., Lopez, L., and Peralba, M. do C. R. 2016. Soil Organic Carbon of Mangrove Forests (*Rhizophora and Avicennia*) of the Venezuelan Caribbean Coast. *Organic Geochemistry*, (100): 51–61.

[31] Baumgartner, P., and Cherlet J. 2016. Economics of Land Degradation and Improvement: An Introduction and Overview. In Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development, edited by E. Nkonya, A. Mirzabaev, and J. von Braun, Heidelberg, New York, NY: Springer International Publishing AG Switzerland.1-7

[32] Bhagabati, N. K., Ricketts, T., Sulistyawan, T.B. S., Conte, M., Ennaanay,D., Hadian,O., and Wolny, S. 2014. Ecosystem Services Reinforce Sumatran Tiger Conservation in Land Use Plans. *Biological Conservation* 169: 147–156.

[33] Beatty, C. R., Cox, N. A., and Kuzee, M. E. 2018. Biodiversity Guidelines For Forest Landscape Restoration Opportunities Assessments (First Edition). IUCN, Gland, Switzerland.p.43.

[34] Bell, F. W., Lamb, E. G., Sharma, M., Hunt, S., Anand, M., Dacosta, J., and Newmaster, S. G. 2016. Relative Influence of Climate, Soils, and Disturbance on Plant Species Richness in Northern Temperate and Boreal Forests. *Forest Ecology and Management*, (381): 93–105.

[35]Belsky, A. J., and Gelbard, J. L. 2000. Livestock Grazing and Weed Invasions in the Arid West. Bend (OR): *Oregon Natural Desert Association*. p.31

[36]Belnap, J., and Gillette, D. A.1998. Vulnerability of desert biological soil crusts to wind erosion: the influences of crust development, soil texture, and disturbance. *Journal of Arid Environments* 39:133-142.

[37] Berber, A. S., Tavşanoğlu, Ç., and Turgay, O. C. 2015. Effects Of Surface Fire On Soil Properties in a Mixed Chestnut-Beech-Pine Forest in Turkey. *Flamma*, 6 (2): 78–80.

[38] Bharathi, S., and Prasad, A. G. D. 2015. Regeneration status of tree composition in the sacred groves of Mercara , Central Western Ghats. *European Journal of Experimental Biology*, 5 (10): 29–40.

[39] Blanchet, G., Libohova, Z., Joost, S., Rossier, N., Schneider, A., Jeangros, B., and Sinaj, S. 2017. Spatial Variability of Potassium in Agricultural Soils of the Canton of Fribourg, Switzerland. *Geoderma*, (290): 107–121.

[40] Block, C. E., Knoepp, J. D., and Fraterrigo, J. M. 2012. Interactive Effects of Disturbance and Nitrogen Availability on Phosphorus Dynamics of Southern Appalachian Forests. *Biogeochemistry*, (112): 329–342.

[41 Blumstein, M., and Thompson, J. R. 2015. Land-Use Impacts on the Quantity and Configuration of Ecosystem Service Provisioning in Massachusetts, USA. *Journal of Applied Ecology*, 52 (4): 1009-1019.

[42] Bogale, T., Datiko, D., and Belachew, S. 2017. Structure and Natural Regeneration Status of Woody Plants of Berbere Afromontane Moist Forest, Bale Zone, South East Ethiopia. *Open Journal of Forestry*, (7): 352–371.

[43] Bonari, G., Acosta, A. T. R., and Angiolini, C. 2017. Mediterranean Coastal Pine Forest Stands: Understorey Distinctiveness or Not? *Forest Ecology and Management*, (391): 19–28.

[44] Borrelli, P., Robinson, D. A., Fleischer, L. R., Lugato, E., Ballabio, C., and Panagos, P. 2017. An Assessment of the Global Impact of 21st Century Land Use Change on Soil Erosion. *Nature Communications* 8 (1): 1–13.

[45] Brosofske, K. D., Chen,J., and Crow, T. R. 2001. Understory Vegetation and Site Factors: Implications for a Managed Wisconsin Landscape. *Forest Ecology and Management* 146 (1–3): 75–87.

[46] Bryceson, I. 2002. Coastal Aquaculture Developments In Tanzania : Sustainable and Non-Sustainable Experiences. *Western Indian Ocean Journal of Marine Sciences*, 1: 1–10.

[47] Bray, R., and Kurtz, L. 1945. Determination of Total, Organic, and Available Forms of Phosphorus in Soils. Soil Science. *Soil Science,* 59 (1): 39–46.

[48] Britton, A. J., Pearce, I. S. K., and Jones, B. 2005. Impacts of Grazing on Montane Heath Vegetation in Wales and Implications for the Restoration of Montane Areas. *Biological Conservation*, 125 (4): 515–524.

[49] Buendia, C., Kleidon, A., and Porporato, A. 2010. The Role of Tectonic Uplift, Climate and Vegetation in the Long-Term Terrestrial Phosphorous Cycle. *Biogeosciences*, (7): 301–333.

[50] Burt, R. 2004. *Soil Survey Laboratory Methods Manual;* Soil Survey Investigations Report No.42, Version 4.0 November 2004; USDA-Natural Resources Conservation Service (NRCS). Lincoln, Nebrasca.p.700.

[51] Cankaya, S., Balkaya, A., and Karaagac, O. 2010. Canonical Correlation Analysis for the Determination of Relationships Between Plant Characters and Yield Components in Red Pepper [*Capsicum annuum* L . var . conoides (Mill .) Irish] Genotypes. *Spanish Journal of Agricultural Research*, 8 (1): 67–73.

[52] Carnevale, N. J., and Montagnini, F. 2002. Facilitating Regeneration of Secondary Forests With the Use of Mixed and Pure Plantations of Indigenous Tree Species. *Forest Ecology and Management*, 163 (1–3): 217–227.

[53] Cathcart, M., Thomas, A., Harrell, T., and Lane, S. 2008. *Army Advanced Concept Workshop on Disturbed Soil Characterization and Exploitation*. Atlanta. 1-34.

[54] Catterall, C. P., Shoo, L. P., and Freebody, K. (2014). Natural Regeneration and Rainforest Restoration: Outcomes, Pathways and Management of Regrowth. *NERP Tropical Ecosystems Hub Factsheet*, 1–8.

[55] Cavelier, J., Aide, T. M., Dupuy, J. M., Eusse, A., and Santos, C. 1999. Long-Term Effects of Deforestation on Soil Properties and Vegetation in a Tropical Lowland Forest in Colombia, *Ecotropicos,* 12 (2): 57–68.

[56] Certini, G. 2005. Effects of Fire on Properties of Forest Soils : A Review. *Oecologia,* 1–10.

[57] Chapungu, L., and Hove, C. 2013. Human Perceptions on Degradation of Wetland Ecosystems : The Case of Magwenzi Wetland in Chivi District ; Zimbabwe. *Greener Journal of Geology and Earth Sciences*, 1 (1):13–22.

[58] Charan, G., Bharti, V. K., Jadhav, S. E., Kumar, S., Acharya, S., Kumar, P. Srivastava, R. B. 2013. Altitudinal Variations in Soil Physico-Chemical Properties at Cold Desert High Altitude. *Journal of Soil Science and Plant Nutrition*, 13(2):267–277.

[59] Chaudhary, S., Dago, T., Tshering, P., Kabir, U., Bandana .S., and Nakul, C. 2017. Impacts of Land Cover Change on a Mountain Ecosystem and Its Services: case Study from The Phobjikha Valley, Bhutan. *Ecosystem Health and Sustainability* 3: 1–12.

[60] Chazdon, R. L. 2012. Regeneração de Florestas Tropicais. *Boletim Do Museu Paraense Emílio Goeldi. Ciências Naturais*, 7 (3): 195–218.

[61] Chazdon, Robin L., Pedro H. S. Brancalion, Lars L., Aoife., Kathleen., Chetan., Julian., Ima., and Sarah, J.W. 2016. When is A Forest A Forest? Forest Concepts and Definitions in the Era of Forest and Landscape Restoration. *Ambio* 45 (5): 538–50.

[62] Chen, W., Moriya, K., Sakai, T., Koyama, L., and Cao, C. 2014. Monitoring of Post-Fire Forest Recovery Under Different Restoration Modes Based on Time Series Landsat Data. *European Journal of Remote Sensing*, 47 (1): 153–168.

[63] Chen, J., Lu, M., Chen, X., Chen, J., and Chen, L. 2013a. A Spectral Gradient Difference Based Approach for Land Cover Change Detection. *ISPRS Journal of Photogrammetry and Remote Sensing*, 85: 1–12.

[64] Chen, J., Wu, H., Li, S., Liao, A., He, C., and Peng, S. 2013b. Temporal Logic And Operation Relations Based Knowledge Representation for Land Cover Change Web Services. *ISPRS Journal of Photogrammetry and Remote Sensing*, 83: 140–150.

[65] Chen, X., and Li, B.L. 2003. Change in Soil Carbon and Nutrient Storage After Human Disturbance of a Primary Korean Pine Forest In Northeast China. *Forest Ecology and Management*, (186): 197–206.

[66] Christian, A. A., Lejoly, J., Hardy, O. J., Christian, A. A., Vleminckx, J., Drouet, T., ... Hardy, O. J. 2013. Species Responses to Edaphic Heterogeneity in Semi-Deciduous Forests From the Congo Basin by Deciduous Forests From the Congo Basin. *Greener Journal of Biological Sciences*, 3 (10): 365-374.

[67] Chu, T., and Guo, X. 2013. Remote Sensing Techniques in Monitoring Post-Fire Effects and Patterns of Forest Recovery in Boreal Forest Regions: A Review. *Remote Sensing*, 6 (1): 470–520.

[68] Cierjacks, A., Rühr, N. K., Wesche, K., and Hensen, I. 2008. Effects of Altitude and Livestock on the Regeneration of Two Tree Line Forming *Polylepis* Species in Ecuador. *Plant Ecology*, 194 (2): 207–221.

[69] Corbin, J. D., and Holl, K. D. (2012). Applied Nucleation as a Forest Restoration Strategy. *Forest Ecology and Management*, 265: 37–46.

[70] Comita, L. S. 2014. Drought as a Driver of Tropical Tree Species Regeneration Dynamics and Distribution Patterns. *Forests and Global Change*, Cambridge University Press, p. 48.

[71] Cork, S. J., and Shelton, D. 2000. The Nature and Value of Australia ' s Ecosystem Services : A Framework for Sustainable Environmental Solutions. In *Sustainable Environmental Solutions for Industry and Government Proceedings of the 3rd Queensland Environmental Conference*, 151–159.

[72] Cortines, E., and Valcarcel, R. 2009. Influence Of Pioneer-Species Combinations on Restoration of Disturbed Ecosystems in the Atlantic Forest, Rio de Janeiro, Brazil. *Revista Árvore*, 33 (5): 927–936.

[73] Costanza, R., D'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., ... van den Belt, M. 1997. The Value of the World's Ecosystem Services and Natural Capital. *Nature*, 387 (6630): 253–260.

[74] Curran, M. P., Heninger, R. L., Maynard, D. G., and Powers, R. F. 2003. Harvesting Effects on Soils, Tree Growth, and Long-Term Productivity. *Productivity of Western Forests:A Forest Products Focus*,3–15.

[75] Curtis, J. T. 1959. The Vegetation of Wisconsin: an Ordination of Plant Communities. University of Wisconsin Press, Madison, p. 640.

[76] Dale, V. H., and Polasky, S. 2007. Measures of the Effects of Agricultural Practices on Ecosystem Services, *Ecological Economics*, 64 (2): 286–296.

[77] Damian, T., Schroeder J. M., and Struwe, J. 2014. Regeneration Potential and Stand Structure of a Proposed Plantation Site in the Transition Zone of Ghana." *International Journal of Biodiversity and Conservation* 6 (3): 238-246.

[78] De Caires, S. A., Wuddivira, M. N., and Bekele, I. 2014. Assessing the Temporal Stability of Spatial Patterns of Soil Apparent Electrical Conductivity Using Geophysical Methods. *International Agrophysics*, 28 (4): 423–433.

[79] Deng, L., Sweeney, S., and Shangguan, Z. P. 2014. Grassland Responses to Grazing Disturbance: Plant Diversity Changes With Grazing Intensity in a Desert Steppe. *Grass and Forage Science*, 69 (3): 524–533.

[80] Defries, R., Rovero, F., Wright, P., Ahumada, J., Andelman, S., Brandon, K., ... Liu, J. 2010. From plot to landscape scale: Linking tropical biodiversity measurements across spatial scales. *Frontiers in Ecology and the Environment*, 8 (3): 153–160.

[81] Delang, C. O., and Li, W. M. 2013. Eological Succession On Fallowed Shifting Cultivation Fields. A Review of the Literature. In Forest Structure, *Briefs in Ecology*, p. 132.

[82] Deng, L., Sweeney, S., and Shangguan, Z. P. 2014. Grassland responses to grazing disturbance: Plant diversity changes with grazing intensity in a desert steppe. *Grass and Forage Science*, 69 (3): 524–533

[83] Demir, M., Makineci, E., and Yilmaz, E. 2007. Harvesting Impact on Herbaceous Understory, Forest Floor and Top Soil Properties on Skid Road in a Beech (Fagus orientalis Lipsky) Stand. *Journal of Environmental Biology*, 28(April), 427–432.

[84] Devi, S. L.,and Yadava, P. S. (200. Floristic Diversity Assessment and Vegetation Analysis of Tropical Semievergreen Forest of Manipur, North East India. *Tropical Ecology*, 47 (1): 89–98.

[85] Devisscher, T. 2010. *The Economics of Climate Change in Tanzania; Ecosystem-based Adaptation in Tanzania.*The Role of Ecosystem Services for Human Well-

Being and Climate Change; Ecosystems Report for the Economics of Climate in Tanzania Project. Dar es Salaam, Tanzania. p. 45.

[86] Dickinson, Y. 2014. Landscape Restoration Of A Forest with a Historically Mixed-Severity Fire Regime: What Was the Historical Landscape Pattern of Forest and Openings? *Forest Ecology and Management*, 331: 264–271.

[87]Dormaar, J.F, Willms, W.D. 1998. Effect of Forty-Four Years of Grazing on Fescue Grassland Soils. *Journal of Range Management* 52:122-126.

[88] Doolittle, J. A., and Brevik, E. C. 2014. The Use Of Electromagnetic Induction Techniques in Soils Studies. *Geoderma*, 223–225 (1): 33–45.

[89] Duah-gyamfi, A., Kyereh, B., Adam, K. A., Agyeman, V. K., and Swaine, M. D. 2014. Natural Regeneration Dynamics of Tree Seedlings on Skid Trails and Tree Gaps Following Selective Logging in a Tropical Moist Semi-Deciduous Forest in Ghana. *Journal of Forestry*, 4 (1): 49–57.

[90] Duku, H., Rathjens, H., Zwart,S. J., and L. Hein. 2015. Towards Ecosystem Accounting: a Comprehensive Approach to Modelling Multiple Hydrological Ecosystem Services. Hydrology and Earth System Sciences 19 (10): 4377–4396.

[91] Dumroese, R. K., Landis, T. D., Pinto, J. R., Haase, L., Wilkinson, K. W., and Davis, A. S. 2016. Meeting forest restoration challenges : Using the Target Plant Concept. Reforesta,1: 37–52.

[92] Durst, P. 2009. Asia-Pacific Forestry Sector Outlook Study II Working Paper Series Paper No. APFSOS II/WP/2009/22. Thailand Forestry Outlook Study. Food and Agriculture Organization of the United Nations Regional Office for Asia and the Pacific, Bangkok. p.106.

[93] Dyer, J. M. 2006. Revisiting the Deciduous Forests of Eastern North America. *BioScience*, 56 (4):341–352.

[94] Eales, J., Haddaway, N. R., Bernes, C., Cooke, S. J., Jonsson, B. G., Kouki, J., and Petrokofsky, G. 2016. What is the Effect of Prescribed Burning in Temperate and Boreal Forest on Biodiversity, Beyond Tree Regeneration, Pyrophilous and Saproxylic Species ? A Systematic Review Protocol. *Environmental Evidence*, 1–8.

[95] Eff, J. C. N., Eynolds, R. L. R., and Elnap, J. B. 2005. Multi-Decadal Impacts of Grazing On Soil Physical and Biogeochemical Properties in Southeast Utah. *Ecological Applications,* 15 (1): 87–95.

[96] Elliott, K. J., Harper, C. A., and Collins, B. 2011. Herbaceous Response to Type and Severity of Disturbance. In C. H. Greenberg (Ed.), *Sustaining Young Forest Communities, Managing Forest Ecosystems* (21): 97–119.

[97] Elmqvist, T., Pyykönen, M., Tengö, M., Rakotondrasoa, F., Rabakonandriahina, E., and Radimilahy, C. 2007. Patterns of Loss and Regeneration of Tropical Dry Forest in Madagascar: The Social Institutional Context. *PLoS ONE,* 2 (5): 1-14.

[98] Eludoyin, O. S. 2016. Understorey Vegetation Assessment in *Gmelina Arborea* (Roxb.) Plantations of Different Ages in Omo Forest Reserve, Southwestern Nigeria. Nature and Science, 14 (10): 92–101.

[99] Eni, D. D., A. I. Iwara, and Offiong, R.A. 2012. Analysis of Soil-Vegetation Interrelationships in a South-Southern Secondary Forest of Nigeria. *International Journal of Biology*, 3: 1–9.

[100]Erickson, V., Aubry, C., Berrang, P., Blush, T., Bower, A., Crane, B., ... Mahalovich, M. 2012. Genetic Resource Management and Climate Change: Genetic Options for Adapting National Forests to Climate Change. Forest Management Genetic Resource Management Program. USDA, Washington, DC. p. 24.

[101]Eva, H. D., Brink, A., and Simonetti, D. 2006. Monitoring Land Cover Dynamics in Sub-Saharan Africa. Institute for Environment and Sustainability, European Communities, Luxembourg. https://www.eldis.org/pdf.1-44. [Accessed on 18 /05/ 2017]

[102]Fang, Z., Bao, W., Yan, X.,and Liu, X. 2014. Understory Structure and Vascular Plant Diversity in Naturally Regenerated Deciduous Forests and Spruce Plantations on Similar Clear-Cuts: Implications for Forest Regeneration Strategy Selection. *Forests*, 5 (4). 715–743.

[103]FAO. 2006. *Global Forests Resorces Assesment 2005- Report on Fires in the North East Asian Region. Fire Management Working Paper 6.* (No. FM/6/E) (Vol. 3). Rome, Italy. p.1-22.

[104] FAO. 2009. *Case Studies on Measuring and Assessing Forest Degradation Impact of Developmental Projects in the Humid Evergreen Broad-leaved Forest:*Wasabi Pilot Project at Lamperi, Western Bhutan (Working Paper 164). Rome, Italy. p. 1-23.

[105]FAO. 2010. *Global Forest Resources Assessment; Main report* (Vol. 147). Rome, Italy. p. 350.

[106]FAO. 2010. Guidelines on Sustainable Forest Management in Drylands of Sub-Saharan Africa. Arid Zone Forests and Forestry Working Paper No. 1. Rome. p.58.

[107]FAO. 2011. *Abiotic Disturbances and their Influence on Forests Health. Forest Health and Biosecuity Woriking Paper FBS/35e; In proceedings of ForestSat 2005.* Rome Italy. p. 51

[108]FAO and JRC. 2012. Global Forest Land-Use Change 1990–2005, by Lindquist, E.J., D'Annunzio, R., Gerrand ,A., MacDicken, K., Achard, F., Beuchle, R., Brink,A., H.D. Eva,P. Mayaux, J. San-Miguel-Ayanz and H-J. Stibig. FAO Forestry Paper No. 169. Food and Agriculture Organization of the United Nations and European Commission Joint Research Centre. Rome, Italy, p. 53.

[109]FAO. 2015. Global Forest Resources Assessment 2015. Desk Reference. Rome, Italy. p. 244

[110] Fetene, A., Hilker, T., Yeshitela, K., Prasse, R., Cohen, W., and Yang, Z. 2015. Detecting Trends in Landuse and Landcover Change of Nech Sar. *Environmental Management*: p.11.

[111]Fleischner, T. L. 1994. Ecological Costs of Livestock Grazing in Western North America. *Conservation Biology* 8:629-644

[112]Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R.,... Snyder, P. K. 2005. Global Consequences of Land Use. *Science*, 309: 570–574.

[113]FRA. 2005. *Global Forest Resources Assessment 2005*. Rome Italy. p.1-5

[114]FRA.2018. Terms and Definitions. Global Forest Resources Assessment 2020. Working Paper No. 188. Rome, Italy. p. 1-32

[115]Frank, D. A., McNaughton, S. J., and Tracy, B. F. 1998. The Ecology of the Earth's Grazing Ecosystems. *BioScience* 48:513-521.

[116]Franklin, J. F., and Johnson, K. N. 2012. A Restoration Framework for Federal Forests in the Pacific Northwest. *Journal of Forestry*, 110 (8): 429–439.

[117]Francis, J., and Bryceson, I. 2001. Tanzanian Coastal And Marine Resources: Some Examples Illustrating Questions of Sustainable Use. In Lessons Learned: Case Studies in Sustainable Use. Dar es Salaam, Tanzania. Retrieved from http://scholar.google.com/pdf. 76–102 [Accessed on 06/06/2017.]

[118]Freilich, J. E.,Emlen, J. M., Duda, J. J., Freeman, D. C., and Cafaro, P. J. 2003. Ecological effects of ranching: a six-point critique. *BioScience* 52:759-765.

[119]Fujita, R., Lynham,J., Micheli, F., Feinberg, P. G., Bourillón, L., Sáenz-Arroyo,A and Markham,A. C. 2013. Ecomarkets for Conservation and Sustainable Development in the Coastal Zone. *Biological Reviews*, 88 (2): 273–286.

[120]Gairola, S., Sharma, C. M., Ghildiyal, S. K., and Suyal, S. 2012. Chemical Properties of Soils in Relation to Forest Composition in Moist Temperate Valley Slopes of Garhwal Himalaya, India. *Environmentalist*, 32 (4): 512–523.

[121]Gimenez, D., Kluchinski, D., Murphy, S., and Muldowny, L. S. 2010. *Assessment of Soil Disturbance on Farmland Presented to New Jersey State Agriculture Development Committee*. New Brunswick, NJ 08901-8525. p. 20.

[122]Giliba, R. A., Mafuru, C. S., Paul, M., Kayombo, C. J., Kashindye, A. M., Chirenje, L. I., and Musamba, E. B. 2011. Human Activities Influencing Deforestation on Meru Catchment Forest Reserve, Tanzania. *Journal of Human Ecology*, 33 (1): 17–20.

[123]Girma, A., and Mosandl, R. 2012. Structure and Potential Regeneration of Degraded Secondary Stands in Munessssa-Shashemene Forest, Ethiopia. *Journal of Tropical Forest Science* 24 (1): 46–53.

[124]Global Forest Watch (GFW) .2016. Data Digest: Tree Cover Extent in Tanzania. Retrieved from http://www.globalforestwatch.org/country/TZA [Accessed on 12 March 2016].

[125]Godoy, F. L., Tabor, K., Burgess, N. D., Mbilinyi, B. P., Kashaigili, J. J., and Steininger, M. K. 2011. Deforestation and CO_2 Emissions in Coastal Tanzania From 1990 to 2007. *Environmental Conservation*, 39 (1): 62–71.

[126]Goebel, P. C., Wyse, T. C., and Gregory Corace III. 2005. Determining Reference Ecosystem Conditions for Disturbed Landscapes within the Context of Contemporary Resource Management Issues. *Journal of Forestry,* 11 (1): 351–356.

[127]Goldmark, P. 2014. Eastern Washington Forest Health: Hazards, Accomplishments and Restoration Strategy. A Report to the Washington State Legislature, Washington DC, USA.p 65.

[128]Golluscio, R. A., Austin, A. T., García Martínez, G. C., Gonzalez-Polo, M., Sala, O. E., and Jackson, R. B. 2009. Sheep Grazing Decreases Organic Carbon and Nitrogen Pools in the Patagonian Steppe: Combination of Direct and Indirect Effects. *Ecosystems*, 12 (4): 686–697.

[129]Groppo, J. D., Lins, S. R. M., Camargo, P. B., Assad, E. D., Pinto, H. S., Martins, S. C., … Martinelli, L. A. 2015. Changes in Soil Carbon, Nitrogen and Phosphorus Due to Land-Use Changes in Brazil. *Biogeosciences*, (2050): 4765–4780.

[130]Grossmann, E. B., and Mladenoff, D. J. 2008. Farms, Fires, and Forestry: Disturbance Legacies in the Soils of the Northwest Wisconsin (USA) Sand Plain. *Forest Ecology and Management*, 256 (4): 827–836.

[131]Guerrero, P. C., and Bustamante, R. O. 2007. Can Native Tree Species Regenerate in Pinus Radiata Plantations in Chile?. Evidence From Field and Laboratory Experiments. *Forest Ecology and Management*, 253 (1–3): 97–102.

[132]Guerry, A. D., Polasky, S., Lubchenco, J., Chaplin- Kramer, R., Daily, G. C. Griffin, and Vira. B. 2015. Natural Capital and Ecosystem Services Informing Decisions: From Promise to Practice. *PNAS,* 112 (24): 7348–7355.

[133]Guilherme, F. A. G., Ferreira, T. O., Assis, M. A., Torrado, P. V., and Morellato, L. P. C. 2012. Soil Profile, Relief Features and Their Relation to Structure and Distribution of Brazilian Atlantic Rain Forest Trees. *Scientia Agricola*, 69 (1): 61–69.

[134]Gxasheka, M., Beyene, S. T., Mlisa, N. L., and Lesoli, M. 2017. Farmers' Perceptions of Vegetation Change, Rangeland Condition and Degradation in Three

Communal Grasslands of South Africa. *Tropical Ecology*, 58 (2): 217–228.

[135]Haapane, M., Jansson, G., Nielsen, U. B., Steffenrem, A., and Stener, L.-G. 2015. The Status of Tree Breeding and its Potential for Improving Biomass Production. A Review of Breeding Activities and Genetic Gains in Scandinavia and Finland. (L.G. Stener, Ed.). Uppsala Science Park, 751 83 Uppsala: SKOGFORSK.p.56.

[136]Hames, A. S. 2009. Comparison of Four Classification Methods to Extract Land Use and Land Cover from Raw Satellite Images for Some Remote Arid Areas , Kingdom of Saudi Arabia. *Earth Sci.*, 20 (1): 167–191.

[137]Han, Z., Song, W., and Deng, X. 2016. Responses of Ecosystem Service to Land Use Change in Qinghai Province. *Energies*, 9 (4):1-16.

[138]Halter, R. 2016. *The Use Versus Availability of Wood Extraction at the Baga II Forest Reserve Border Adjacent to Kizanda Village in the West Usambara Mountains , Tanzania.* Independent Study Project (ISP) Collection. Paper 2350. Independent Study Project (ISP) Collection. Dar es Salaam, Tanzania. p. 25.

[139]Hartter, J., Ryan, S. J., Southworth, J., and Chapman, C. A. 2011. Landscapes as Continuous Entities: Forest Disturbance and Recovery in the Albertine Rift Landscape. *Landscape Ecology*, 26 (6): 877–890.

[140]Haugo, R., Zanger, C., DeMeo, T., Ringo, C., Shlisky, A., Blankenship, K.,... Stem, M. 2015. A New Approach to Evaluate Forest Structure Restoration Needs Across Oregon and Washington, USA. *Forest Ecology and Management*, 335: 37–50.

[141]Hessenmöller, D., Elsenhans, A. S., and Schulze, E. D. 2013. Sampling Forest Tree Regeneration with a Transect Approach. *Annals of Forest Research*, 56 (1): 3–14.

[142]He, N. P., Zhang, Y. H., Yu, Q., Chen, Q. S., Pan, Q. M., Zhang, G. M., and Han, X. G. 2011. Grazing Intensity Impacts Soil Carbon and Nitrogen Storage of Continental Steppe. *Ecosphere*, 2 (1): 1-10.

[143]Heineman, K. D., Turner, B. L., Heineman, K. D., Turner, B. L., and Dalling, J. W. 2016. Variation in Wood Nutrients Along a Tropical Soil Fertility Gradient. *New Phytologist*, (March). p.15.

[144]Hitimana, J., Kiyiapi, J. L., and Njunge, J. T. 2004. Forest Structure Characteristics

in Disturbed and Undisturbed Sites of Mt. Elgon Moist Lower Montane Forest, Western Kenya. *Forest Ecology and Management*, 194 (1–3): 269–291.

[145]Hjältén, J., Hägglund, R., Löfroth, T., Roberge, J. M., Dynesius, M., and Olsson, J. 2017. Forest Restoration by Burning and Gap Cutting of Voluntary Set-Asides Yield Distinct Immediate Effects on Saproxylic Beetles. *Biodiversity and Conservation*, 26 (7): 1623–1640.

[146]Hobbie, S. E., Eissenstat, D. M., Chadwick, O. A., Hale, C. M., and Mark, G. 2005. Linking Litter Calcium , Earthworms and Soil Properties : A Common Garden Test With 14 Tree Species. *Ecology Letters*, (8): 811–818.

[147]Holl, Karen D. 2017. Research Directions in Tropical Forest Restoration. *Annals of the Missouri Botanical Garden*, 102 (2): 237–250.

[148]Holl, Karen D., John Leighton Reid, José Miguel Chaves-Fallas, Federico Oviedo-Brenes, and Rakan A. Zahawi. 2017. Local Tropical Forest Restoration Strategies Affect Tree Recruitment More Strongly Than Does Landscape Forest Cover. *Journal of Applied Ecology* 54 (4): 1091–1099.

[149]Holland, E.A., Parton, W. J., Detling, J. K., Coppock, D. L. 1992. Physiological responses of plant populations to herbivory and other consequences of ecosystem nutrient flow. *American Naturalist* 140:685-706.

[150]Hoogendoorn, C. J., Betteridge, K., Costall, D. A., and Ledgard, S. F. 2010. Nitrogen Concentration in The Urine of Cattle, Sheep and Deer Grazing; A Common Ryegrass/Cocksfoot/White Clover Pasture. *New Zealand Journal of Agricultural Research*, 53 (3): 235–243.

[151]Hooper, E., Legendre, P., and Condit, R. 2005. Barriers to Forest Regeneration of Deforested and Abandoned Land in Panama. *Journal of Applied Ecology*, (7): 1165–1174.

[152]Howell, K. M., Msuya, C. A., Mligo, C., Werema, C., Kihaule, P., Honorati, M. and Suleiman, H. O. 2012. *Biodiversity Surveys of Poorly Known Coastal Forests of Southeastern Tanzania and Zanzibar*. Dar es Salaam, Tanzania. p.171

[153]Hu, H., Liu, W., and Cao, M. 2008. Impact of land use and land cover changes on ecosystem services in Menglun, Xishuangbanna, Southwest China. *Environmental Monitoring and Assessment*, 146 (1–3): 147–156.

[154]Huang, W., Pohjonen, V., Johansson, S.,and Nashanda, M. 2003. Species Diversity, Forest Structure and Species Composition in Tanzanian Tropical Forests. *Forest Ecology and Management*, 173 (14): 11-24.

[155]Hubbard, R.K., Newton, G.L., Mill, G.M. 2004. Water quality and the grazing animal. *Journal of Animal Science* 82:E255-E263.

[156]Ichikogu, V. I. 2014. Relationships between Soil and Vegetation Structural Properties of Secondary Forest Regenerating in Degraded Rubber Plantation in the Niger Delta Area of Nigeria. *An International Multidisciplinary Journal, Ethiopia*, 8 (34): 123–139.

[157]IPCC.2003. Good Practice Guidance for Land Use, Land-Use Change and Forestry [Penman, J.,Gytarsky,M., Hiraishi,T.,Krug,T., Kruger,D.,Pipatti,R., Buendia, L., Miwa, K., Ngara,T., Tanabe,K.,Wagner. F (eds.)]. IPCC / OECD / IEA / IGES, Hayama, Japan, p.580.

[158]Ishizuka, S., Iswandi, A., Nakajima, Y., Yonemura, S., Tsuruta, H., and Murdiyarso, D. 2005. The Variation of Greenhouse Gas Emissions From Soils of Various Land-Use / Cover Types In Jambi Province, Indonesia. *Nutrient Cycling in Agroecosystems*, 71 (1993): 17–32.

[159]Islam, K. R., and Weil, R. R. 2000. Land Use Effects on Soil Quality in a Tropical Forest Ecosystem of Bangladesh, *Agriculture, Ecosystems & Environment*, 79 (1): 9–16.

[160]IUCN. 2012. *IUCN Red List Categories and Criteria:Version 3.1.Second edition.* Gland, Switzerland and Cambridge, UK. p. 32.

[161]Jacob, A. L., Lechowicz, M. J., and Chapman, C. A. 2017. Non-Native Fruit Trees Facilitate Colonization of Native Forest on Abandoned Farmland. *Restoration Ecology*, 25 (2): 211–219.

[162]Jantz, D. C., and Manuel, J. J. 2009. Estimating Impacts of Population Growth on Ecosystem Services for the Community of Albemarle County and Charlottesville ,

VA. Population, Virginia. http://cvilletomorrow.typepad.com/pdf: p. 76. [Accessed on 10/06/2017].

[163]Jayakumar, S., Kim, S. S., and Heo, J. 2011. Floristic Inventory and Diversity Assessment - A Critical Review. In *Proceedings of the International Academy of Ecology and Environmental Sciences*. Puducherry, India. (1): 151–168.

[164]Jew, E. K. K., Dougill, A. J., Sallu, S. M., O'Connell, J., and Benton, T. G. 2016. Miombo Woodland under Threat: Consequences for Tree Diversity and Carbon Storage. *Forest Ecology and Management*, 361: 144–153.

[165]Johnson, D. W., Todd, D. E., Trettin, C. F., and Mulholland, P. J. 2008. Decadal Changes in Potassium , Calcium, and Magnesium in a Deciduous, Forest Soil. *Soil Sci. Soc. Am. J.*, (72): 1795–1805.

[166]Jones, H. P., Jones, P. C., Barbier, E. B., Blackburn, R. C., Benayas, J. M. R., Holl, K. D., ... Jones, H. P. 2018. Restoration and Repair of Earth's Damaged Ecosystems. *Proc.R.Soc.B*, 285:1–8.

[167]Jones, E. R., Wishnie, M. H., Deago, J., Sautu, A., and Cerezo, A. 2004. Facilitating Natural Regeneration in *Saccharum spontaneum* (L.) Grasslands Within the Panama Canal Watershed: Effects of Tree Species and Tree Structure on Vegetation Recruitment Patterns. *Forest Ecology and Management*, 191 (1–3): 171–183.

[168]Joyi, O., Utanga, M. M., Dindi, O. O., Ynekulu, J. a, and Ahman, E. B. 2015. The Effect of Forest Fragmentation on Tree Species Abundance and Diversity in the Eastern Arc Mountains of Tanzania. *Applied Ecology and Environmental Research*, 13 (2): 307–324.

[169]Kabrick, J. M., and Goyne, K. W. 2011. Landscape Determinants of Exchangeable Calcium and Magnesium in Ozark Highland Forest Soils. *Soil Sci. Soc. Am. J*, 75 (1): 164–180.

[170]Kalaba, F. K., Quinn, C. H., Dougill, A. J., and Vinya, R. 2013. Floristic Composition, Species Diversity and Carbon Storage in Charcoal and Agriculture Fallows and Management Implications in Miombo Woodlands of Zambia. *Forest Ecology and Management*, (304): 99–109.

[171]Kane, D. 2015. *Carbon Sequestration Potential on Agricultural Lands: A Review of*

Current Science and Available Practices. National Sustainable Agriculture Coalition Breakthrough Strategies and Solutions, LLC. p. 35.

[172]Kaimowitz, D., Enforcement, F. L., Diversity, B., Partnership, F., Witness, G., International, T., and Forest, G. 2003. Forest Law Enforcement and Rural Livelihoods. *International Forestry Review*, 5 (3): 199–210.

[173]Kärvemo, S., Björkman, C., Johansson, T., Weslien, J., and Hjältén, J. 2017. Forest Restoration as a Double-Edged Sword: The Conflict Between Biodiversity Conservation and Pest Control. *Journal of Applied Ecology*, 54 (6): 1658–1668.

[174]Kaspari, M., and Yanoviak, S. P. 2008. Biogeography Of Litter Depth in Tropical Forests : Evaluating the Phosphorus Growth Rate Hypothesis. *Functional Ecology*, 22. 919–923.

[175]Kauano, É. E., Cardoso, F. C. G., Torezan, J. M. D., and Marques, M. C. M. 2013. Micro- a Meso-Scale Factors Affect the Restoration of Atlantic Forest. *Natureza Conservacao*, 11 (2): 145–151.

[176]Kebede, A. S., Brown, S. and Nicholls, R. J. 2010. Synthesis Report: The Implications of Climate Change and Sea-Level Rise in Tanzania – The coastal zones. Global Climate Adaptation Partnership (GCAP), (November),.http://economics-of-cc-in-tanzania.org/images/Tanzania_Synthesis-Report Dar es Salaam. p. 32.

[177]Keenan, R. J., Reams, G. A., Achard, F., de Freitas, J. V., Grainger, A., and Lindquist, E. 2015. Dynamics of Global Forest Area: Results From the FAO Global Forest Resources Assessment 2015. *Forest Ecology and Management*, 352: 9–20.

[178]Kissinger, G., M. Herold, V. De Sy. Drivers of Deforestation and Forest Degradation: A Synthesis Report for REDD+ Policymakers. Lexeme Consulting, Vancouver Canada. p. 48.

[179]Khan, Z. I., Hussain, A., Ashraf, M., and Mcdowell, L. R. 2006. Mineral Status of Soils and Forages in Southwestern Punjab-Pakistan: Micro-minerals. *Asian-Aust.J.Anim.Sci.*, 19 (8): 1139–1147.

[180]Kideghesho, J. R. 2015. Realities on Deforestation in Tanzania-Trends, Drivers, Implications and the Way Forward. Precious Forest- Precious Earth. InTech. p. 21-47.

[181]Kijazi, M., Lusambo, L., Matura, S., and Seki, H. 2014. *Green Economy in Biosphere Reserves; Biodiversity Survey in East Usambara Biosphere Reserve Tanga Region, Tanzania*. Dar es Salaam, Tanzania. p. 85.

[182]Kimaro, J.,and Lulandala, L. 2013. Forest Cover and Land Use Change in Ngumburuni. *Journal of Environment and Ecology*, 4 (2): 113–125.

[183]Kindu, M., Schneider, T., Teketay, D., Knoke, T. (2016). Changes of Ecosystem Service Values in Response to Land Use/Land Cover Dynamics in Munessa-Shashemene landscape of the Ethiopian Highlands. *Sci. Total Environ.*, 547: 137–147.

[184]Kirby, K. R., and Potvin, C. 2007. Variation in Carbon Storage Among Tree Species : Implications for the Management of a Small-Scale Carbon Sink Project. *Forest Ecology and Management* 2 (246): 208–221.

[185]Kjeldahl, J. Z. 1883. A New Method for the Determination of Nitrogen in Organic Bodies. *Analytical Chemistry*, 22: p. 36

[186]Kizza, C. L., Majaliwa, J. G. M., Nakileza, B., Eilu, G., Bahat, I., Kansiime, F., and Wilson, J. 2013. Soil And Nutrient Losses Along the Chronosequential Forest Recovery Gradient in Mabira Forest Reserve, Uganda. *African Journal of Agricultural Research*, 8 (1): 77–85.

[187] Kubiszewski, I., R. Costanza, S. Anderson, and P. Sutton. 2017. The Future Value of Ecosystem Services: Global Scenarios and National Implications. *Ecosystem Services*, 26: 289–301.

[188]Kuenzer, C., Bluemel, A., Gebhardt, S., Quoc, T. V., and Dech, S. 2011. Remote Sensing of Mangrove Ecosystems: *Remote Sensing*, 3 (12): 878-928.

[189]Kuuluvainen, T. 2002. Disturbance Dynamics in Boreal Forests: Defining the Ecological Basis of Restoration and Management of Biodiversity. *Silva Fennica*, 36 (1): 5–11.

[190]Lacey, J.R., Wallander, R., and Olson-Rutz, K. 1992. Recovery, Germinability, and Viability of Leafy Spurge *(Euphorbia Esula)* Seeds Ingested By Sheep and Goats. *Weed Technology* 6:599-602

[191]Laiho, R., Penttilä, T., and Lainest, J. 2004. Variation in Soil Nutrient Concentrations and Bulk Density Within Peatland. *Silva Fennica*, 38 (1): 29–41.

[192]Lal, R. 2012. Enhancing Eco-efficiency in Agro-ecosystems. *Crop Sci.*, 50: 120–131.

[193]Lamb, D., and Gilmour, D. 2005. Rehabilitation and Restoration of Degraded Forests. *Restoration Ecology*, 13 (3): 578–579.

[194]Lee, S. M., Cha, J. G., and Moon, H. G. 2017. Study on the Diagnosis of Disturbed Forest Ecosystem in the Republic of Korea: In Case of Daegwallyeong and Chupungryeong. *Journal of Ecology and Environment*, 41 (1): 1–9.

[195]Legendre, P. 2008. Studying Beta Diversity: Ecological Variation Partitioning by Multiple Regression and Canonical Analysis. *Journal of Plant Ecology*, 1 (1): 3–8.

[196]Lehmann, J., Pereira, J., Steiner, C., Nehls, T., Zech, W., and Glaser, B. 2003. Nutrient Availability and Leaching in an Archaeological Anthrosol and a Ferralsol of the Central Amazon Basin : Fertilizer, Manure and Charcoal Amendments. *Plant and Soil*, (249): 343–357.

[198]Leps, J., and Smilauer, P. 2003. *Multivariate Analysis of Ecological Data using CANOCO*. New York: Cambridge University Press. p. 269.

[199]Li, R. Q., Dong, M., Cui, J. Y., Zhang, L. L., Cui, Q. G., and He, W. M. 2007. Quantification of the Impact of Land-Use Changes on Ecosystem Services: A Case Study in Pingbian County, China. *Environmental Monitoring and Assessment*, 128 (1–3): 503–510.

[200]Ligate, E. J., Wu, C., and Chen, C. 2017. The Status of Forest Ecosystem Services and Their Management : The Case of Uzigua Forest Reserve in Tanzanian Coastal Forests. *Natural Resources and Conservation*, 5 (2): 21–32.

[201]Lin, D., Lai, J., Yang, B., Song, P., Li, N., Ren, H., and Ma, K. 2015. Forest Biomass Recovery After Different Anthropogenic Disturbances: Relative Importance of Changes in Stand Structure and Wood Density. *European Journal of Forest Research*, 134 (5): 769–780.

[202]Lindsell, J. A., Lee, D. C., Powell, V. J., and Gemita, E. 2015. Availability of Large Seed-Dispersers for Restoration of Degraded Tropical Forest. *Tropical Conservation Science*, 8 (1):17–27.

[203]Lü, X. T., Yin, J. X., Jepsen, M. R., and Tang, J. W. 2010. Ecosystem Carbon Storage and Partitioning in a Tropical Seasonal Forest in Southwestern China. *Forest Ecology and Management*, 260 (10): 1798–1803.

[204] Yang, L., Ranjitkar,S., Harrison, R. D., Xu,J., Ou,X., Ma,X., and He, J. 2017. Selection of Native Tree Species for Subtropical Forest Restoration in Southwest China. *PLoS ONE* 12 (1): 1–15.

[205]Van Hoof,L., and Kraan, M. 2017. Mission Report Tanzania; Scoping Mission Marine Fisheries Tanzania. Wageningen: p. 66.

[206]Madriñán, M. J. M., Al-Hamdan M. Z., Rickman D. L., and Ye, J. 2012. Relationship between Watershed Land Cover and Land-Use Change and Water Turbidity Status of Tampa Bay Major Tributaries, Florida, USA. *Water Air Soil Pollution*, 223 (5): 1–17.

[207]Maitima, J. M., Mugatha,S. M., Reid ,R. S., Gachimbi L. N., Majule,A., Lyaruu,A., and S. Mugisha. 2009. The Linkages between Land Use Change, Land Degradation and Biodiversity across East Africa. *African Journal of Environmental Science and Technology* 3 (10): 310–325

[208]Majumdar, K., and Datta, B. K. 2014. Vegetation Types, Dominant Compostion, Wood Plant Diversity and Stand Structure in Trishna Wildlife Sancuary of Nothereast India. *Journal of Environmental Biology*, (36): 409–418.

[209]Makineci, E., Demir, M., and Kartaloglu, M. 2015. Acidity (pH) and Electrical Conductivity Changes in Runoff Water From Ditches of Paved and Unpaved Forest Roads. *Baltic Forestry*, 170–175.

[210]Mandal, G., and Joshi, S. P. 2014. Journal of Asia-Pacific Biodiversity Analysis of Vegetation Dynamics and Phytodiversity From Three Dry Deciduous Forests of Doon Valley, Western Himalaya, India. *Journal of Asia-Pacific Biodiversity*, 7(3): 292–304.

[211]Manzoni, S., Trofymow, J. A., Jackson, R. B., and Porporato, A. 2010. Stoichiometric Controls on Carbon , Nitrogen, and Phosphorus Dynamics in Decomposing Litter. *Ecological Monographs*, 80 (1): 89–106.

[212]Marc, L., Babu, S., and Hamilton, K. 2005. Ecosystem Conditions and Human Well-being. In D. Rapport and L. Zhang (Eds.), *Millenium Ecosystem Assessment. Ecosystems and HumanWell-being: Current State and Trends*, Washington, DC: p.123–164.

[213]Martinuzzi, S., Gavier-Pizarro, G. I., Lugo, A. E., & Radeloff, V. C. 2015. Future Land-Use Changes and the Potential for Novelty in Ecosystems of the United States. *Ecosystems*, 18 (8): 1332–1342.

[214]Martin, Š., Daniel, V., Aytekin, E., and Radim, M. 2016. The Effect of Coppice Management on the Structure, Tree Growth and Soil Nutrients in Temperate Turkey. *Journal of Forest Science*, 61 (1): 27–34.

[215]Maynard, D. G., Paré, D., Thiffault, E., Lafleur, B., Hogg, K. E., and Kishchuk, B. 2014. How Do Natural Disturbances and Human Activities Affect Soils and Tree Nutrition And Growth in the Canadian Boreal Forest? *Environmental Reviews*, 22 (2):161–178.

[216]Mbwambo, L., Valkonen, S., and Kuutti, V. 2008. *Structure and dynamics of miombo woodland stands at Kitulangalo Forest Reserve, Tanzania:*Working Papers of the Finnish Forest Research Institute (98):10–19.

[217]McElhinny, C., Gibbons, P., Brack, C., and Bauhus, J. 2005. Forest and Woodland Stand Structural Complexity: Its Definition and Measurement. *Forest Ecology and Management*, 218 (1–3): 1–24.

[218]Merganic, J., Merganicova, K., Marusak, R., and Audolenska, V. 2012. Plant Diversity of Forests. In Forest Ecosystems – More than Just Trees, p. 28.

[219]Mganga, N. D., and Lyaruu, H. V. M. 2016. Plant Species Diversity in Western Tanzania : Comparison between Frequently Burnt and Fire Suppressed Forests. *Int. J. Pure App. Biosci.* 4 (3): 28–44.

[220]Millenium Ecosystem Assessment. 2003. *Ecosystems and Human Well-being; A Framework for Assessment*. ISLAND PRESS ,Washington, DC: p.155.

[221]Millenium Ecosystem Assessment. 2005. *Ecosystems and Human Well-Being: Biodiversity Synthesis*. World Resources Institute, Washington, DC: p.100.

[222]Mishra, B. P., and Tripathi, S. K. 2015. Recovery of Plant Diversity and Soil Nutrients During Stand Development in Subtropical Forests of Mizoram, Northeast India. *Biodiversitas,* 16 (2): 205–212.

[223]Mligo, C. 2015a. Conservation of Plant Biodiversity Of Namatimbili Forest in the Southern Coastal Forests of Tanzania. *International Journal of Biodiversity and Conservation*, 7 (3): 148–172.

[224]Mligo, C. 2015b. Conservation of Plant Species Diversity Based on Richness and Evenness Criteria in the Coastal Forests of Tanzania. *Journal of Environment and Ecology*, 6 (1): 1–20.

[225]Mligo, C., Lyaruu, H., and Ndangalasi, H. 2009. Vegetation Community Structure , Composition and Distribution Pattern in the Zaraninge Forest, *Journal of East African Natural History*, 98 (2): 223–239.

[226]Mligo, C., Lyaruu, H., and Ndangalasi, H. 2011. The Effect of Anthropogenic Disturbances on Population Structure and Regeneration of *Scorodophloeus Fischeri* and *Manilkara Sulcata* in Coastal Forests of Tanzania. *Southern Forests: A Journal of Forest Science*, 73 (1): 33–40.

[227]Moore, B., and Allard, G. 2008. *Climate Change Impacts on Forest Health,; Forest Health and Biosecurity Working Papers FBS/34E*. Rome, Italy. Retrieved from http://www.fao.org/pdf. p. 4. [Accessed on 10/05/2017].

[228]Moreira, A., and Fageria, N. K. 2009. Soil Chemical Attributes of Amazonas State , Brazil. *Communications in Soil Science and Plant Analysis,* (40): 2912–2925.

[229]Mori, A. S. 2011. Ecosystem Management Based on Natural Disturbances: Hierarchical Context and Non-Equilibrium Paradigm. *Journal of Applied Ecology*, 48 (2): 280–292.

[230]Mukul, S. A., Herbohn, J., Firn, J., Group, T. F., Sciences, F., Forests, T., and Sciences, B. 2016. Co-Benefits of Biodiversity and Carbon From Regenerating Secondary Forests in the Philippines Uplands: Implications for Forest Landscape Restoration. *Biotropica*, (1): 1–16.

[231]Muluneh, A. 2010. *Synthesis of research on land use and land cover.* UNU-Land Restoration Training Programme, Reykjavik, Iceland. p. 21.

[232]Miraemiliana,M., Chang,S.,Abu,A.,Yap,H., and Yong, K. 2017. Automated Classification of Tropical Shrub Species: A Hybrid of Leaf Shape and Machine Learning Approach." *Peer J* (5): 1-23.

[233]Enrique,M.,Calle,Z.,Uribe,F.,Calle,A and Solorio, B. 2011. Native Trees and Shrubs for the Productive Rehabilitation of Tropical Cattle Ranching Lands. *Forest Ecology and Management* 261 (10): 1654–1663.

[234]Nadeau, M. B., and Sullivan, T. P. 2015. Relationship Between Plant Biodiversity and Soil Fertility in a Mature Tropical Forest, Costa Rica. *International Journal of Forestry Research*, (2015): 1–13.

[235]Nasi, R., and Van Vliet, N. 2009. *Case Studies on Measuring and Assessing Forest Degradation - Defaunation and Forest Degradation in Central African Logging \rConcessions: How to Measure the Impacts of Bush Meat Hunting on the Ecosystem. Forest Resources Assessment Working Paper.* Rome Italy.p.12.

[236]Nave, L. E., Vance, E. D., Swanston, C. W., and Curtis, P. S. 2010. Harvest Impacts on Soil Carbon Storage in Temperate Forests. *Forest Ecology and Management*, 259: 857–866.

[237]Navroodi, I. H. 2015. Effects of Livestock Exclusion on Forest Tree Regeneration (Case Study : Ramsar District 1 – Iran). *Journal of Forests Science*, 2015 (1): 1–6.

[238]Ndang'ang'a, P. K., Eshiamwata, G., Ngari, A., Pius, E., John, J., Arinaitwe, J., and Mbwana, S. 2008. *Status Report for the Eastern Arc Mountains and Coastal forests of Kenya and Tanzania Region, 2007,Instituting a standardized sustainable biodiversity monitoring system in the Eastern Arc Mountains and Coastal forests of Kenya and Tanzania region.* Dar es Salaam, Tanzania. p.74.

[239]NGCC. 2014. 30-Meter Global and Cover Dataset: GlobeLand30 Product Description. National Geomatics Center of China, Beijing, China. http://www.globallandcover.com/home/Enbackground.aspx/pdf. p. 25. [Accessed on 15/06/2016].

[240]Nizam, M. S., Jeffri, A. R., and Latiff, A. 2013. Structure of Tree Communities and

their Association With Soil Properties in Two Fan-Palm Dominated Forests of East Coast Peninsular Malaysia. *Tropical Ecology*, 54 (2): 213–226.

[241]Nkonya, E., J. Koo, E. Kato, and Z. Guo. 2013. Trends and Patterns of Land Use Change and International Aid in Sub-Saharan Africa. WIDER Working Paper, No. 2013/ 110, WIDER, Helsinki. 33. http://wwwwider.unu.edu/pdf. [Accessed on 15/03/ 2017]

[242]Otsuka, K., and F. Place. 2014. Changes in Land Tenure and Agricultural Intensification in Sub-Saharan Africa. World Institute for Development Economics Research. Working Paper, Helsinki, Finland, 051: 18.

[243]Paillet, Y., Cassagne, N., Brun, J. J., Paillet, Y., Cassagne, N., and Monitoring, J. J. B. 2010. Monitoring Forest Soil Properties With Electrical Resistivity. *Biology and Fertility of Soils, Springer Verlag*, 46 (5). p2. 0.

[244]Pal, S., Panwar, P., and Bhardwaj, D. R. 2013. Soil Quality Under Forest Compared to Other Land-Uses in Acid Soil of North Western Himalaya , India. *Ann. For. Res.*, 56(1): 187–198.

[245]Patra, B., Dey, S. K., and Das, M. T. 2015. Forest Management Practices for Conservation of Biodiversity: An Indian Perspective. *International Journal of Environmental Biology*, 5 (4): 93–98.

[246]Pan, Y., Chen, J. M., Birdsey, R., McCullough, K., He, L., and Deng, F. 2011. Age Structure and Disturbance Legacy of North American Forests. *Biogeosciences*, 8 (3): 715–732.

[247]Parras-Alcántara, L., Lozano-García, B., and Galán-Espejo, A. 2015. Soil Organic Carbon Along an Altitudinal Gradient in the Despeñaperros Natural Park, Southern Spain. *Solid Earth*,(6): 125–134.

[248]Pausch, J., and Kuzyakov, Y. 2012. Soil Organic Carbon Decomposition From Recently Added and Older Sources Estimated by 13C Values of CO_2 and Organic Matter. *Soil Biology and Biochemistry*, (55): 40–47.

[249]Peña-Alzate, S., and Barriga, J. E. C. 2017. Approaching the Concepts of Ecosystems Resilience and Stability Through Spatiotemporal System Dynamics and Agent-Based Modelling. *Revista Facultad de Ingeniería, Universidad de Antioquia*,

(84): 84–96.

[250]Peres, C. A., Barlow, J., and Laurance, W. F. 2006. Detecting Anthropogenic Disturbance in Tropical Forests. *Trends in Ecology and Evolution*, 21 (5): 227–229.

[251]Petz, K., Schulp, C. J. E., Zanden, E. H. Van Der, Veerkamp, C., Schelhaas, M., Nabuurs, G., and Hengeveld, G. 2016. *Indicators and Modelling of Land Use, Land Management and Ecosystem Services. Methodological documentation.* Hague (2386): p. 109.

[252]Pfeifer, M., Platts, P. J., Burgess, N. D., Swetnam, R. D., Willcock, S., Lewis, S. L., and Marchant, R. 2013. Land Use Change and Carbon Fluxes in East Africa Quantified Using Earth Observation Data and Field Measurements. *Environmental Conservation*, 40 (3): 241–252.

[253]Philip, M.S. 1983. Measuring Trees and Forests. A Textbook Written for Students in Africa, University of Dar es Salaam Press, Dar es Salaam, p. 338.

[254]Phiri, D., and Morgenroth, J. 2017. Developments in Landsat Land Cover Classification Methods : A Review. *Remote Sensing*, 9 (967):1–25.

[255]Pinto, S. R., Melo, F., Tabarelli, M., Padovesi, A., Carrascosa, H., Calmon, M., ... César, R. G. 2014. Governing and Delivering a Biome-Wide Restoration Initiative: The Case of Atlantic Forest Restoration Pact in Brazil. *Forests*, 5: 2212–2229.

[256]Polster, D. F. 2016. Natural Processes for The Restoration of Drastically Disturbed Sites. *JASMR*, 5 (2): 77–90.

[257]Poorter, L., van der Sande, M. T., Thompson, J., Arets, E. J. M. M., Alarcn, Alvarez-Sanchez, J., ... Pea-Claros, M. 2015. Diversity Enhances Carbon Storage in Tropical Forests. *Global Ecology and Biogeography*, 24 (11):1314–1328.

[258]Potter, C. 2014. Global Assessment Oo Damage to Coastal Ecosystem Vegetation From Tropical Storms. Remote Sensing Letters, 5 (4): 315–322.

[259]Powell, L L, G Zurita, J. D. Wolfe, E.I. Johnson and Stouffer, P.C. 2015. Changes in Habitat Use at Rain Forest Edges Through Succession: A case Study of Understory Birds in the Brazilian Amazon. *Biotropica* (47): 723–32.

[260]Prach, K., Řehounková, K., Lencová, K., Jírová, A., Konvalinková, P., Mudrák, O., ... Pyšek, P. 2014. Vegetation Succession in Restoration of Disturbed Sites in Central Europe: The Direction of Succession and Species Richness Across 19 Seres. *Applied Vegetation Science*, 17(2): 193–200.

[261]Preston, M. A. 2006. *Anthropogenic disturbance of forests , its effects on primates , and conservation in West Usambara , Tanzania; Doctoral thesis*. Office of Graduate Studies, University of California. p. 160.

[262]Pulla, S., Riotte, J., Suresh, H. S., Dattaraja, H. S., and Sukumar, R. 2016. Controls of Soil Spatial Variability in a Dry Tropical Forest. *PLoS ONE*, 11 (4):1–20.

[263]Qu, T., Du, W., Yuan, X., Yang, Z., Liu, D., and Wang, D. 2016. Impacts of Grazing Intensity and Plant Community Composition on Soil Bacterial Community Diversity in a Steppe Grassland. *PLoS ONE*, 11 (7): 1–16.

[264]Quesada, C. A., Phillips, O.L., Schwarz, M., Czimczik, C. I., Baker, T. R., Patino,S., ...,Vieira, I. 2012. Basin-Wide Variations in Amazon Forest Structure and Function are Mediated by Both Soils and Climate,*Biogeosciences* (9): 2203–2246.

[265]Quintas-Soriano, C., Castro, A. J., Castro,H.,and M. García-Llorente. 2016. Impacts of Land Use Change on Ecosystem Services and Implications for Human Well-Being in Spanish Drylands. *Land Use Policy*, 54: 534–548.

[266]Rabenberg, G., and Kniffen, C. 2014. Electrical Conductivity: The Pulse of the Soil. *Acres U.S.A*, 41 (4).p.2.

[267]Raburu, P. O., and Kwena, F. 2012. *Community Based Approach to the Management of Nyando Wetland , Lake Victoria Basin , Kenya. McPowl Media Limited.* Nairobi, Kenya: Mcpowl Media Ltd. p.157.

[268]Rautiainen, A., Virtanen, T., and Kauppi. P. E. 2016. Land Cover Change on the Isthmus of Karelia 1939–2005 Agricultural Abandonment and Natural Succession. *Environmental Science and Policy*, 55: 127–134.

[269]Razavi, S. M., Mattaji, A., Rahmani, R., and Naghavi, F. 2012. The Assessment of

Plant Species Importance Value (SIV) in Beech (*Fagus orientalis*) Forests of Iran (A Case study : Nav District 2 of Asalem , Guilan Province), 3 (2): 433–439.

[270]Redford, K.H., and Fearn, E. 2007. Protected Areas and Human Displacement: A Conservation Perspective. Working Paper No. 29, Wildlife Conservation Society, NY: p. 152.

[271]Rezapour, S. 2014. Response of Some Soil Attributes to Different Land Use Types in Calcareous Soils With Mediterranean Type Climate in North-West of Iran. *Environ Earth Sci*, (71). 2199–2210.

[272]Reeves, P. N., and Champion, P. D. 2004. *Effects of Livestock Grazing on Wetlands:Literature Review.Environment Waikato Technical Report 2004/6.* Hamilton East: Doccument No. 928468-v2: p 27.

[273]Ribeiro, M. C., Metzger, J. P., Martensen, A. C., Ponzoni, F. J., and Hirota, M. M. 2009. The Brazilian Atlantic Forest: How Much is Left, and How is the Remaining Forest Distributed? Implications for Conservation. *Biological Conservation,* 142 (6): 1141–1153.

[274]Runsten, L., Ravilious, C., Kashindye, A., Giliba, R., Hailakwahi, V., Kashaga, L. R. A., … Miles, L. 2013. *Using spatial information to support decisions on safeguards and multiple benefits for REDD+ in Tanzania*. Dar es Salaam, Tanzania.1-9

[275]Ryšan, L., and Šařec, O. 2008. Research of Correlation Between Electric Soil Conductivity and Yield Based on the Use of GPS Technology, 2008 (3): 136–147.

[276]Ryan, C. M., R. Pritchard, I. McNicol, M. Owen, J. A. Fisher, and C. Lehmann. 2016. Ecosystem Services from Southern African Woodlands and their Future Under Global Change. *Phil Transactions R Social*, 371: 1–16.

[277]Santha, S. D. 2015. Coastal Resources, Ecosystem Services and Disaster Risk Reduction; Analysis of Social and Environmental Vulnerability along the Coast of India." Global Assessment Report on Disaster Risk Reduction 2015. http://www.preven tionweb.net/pdf. p.120. [Accessed on 27/05/ 2017].

[278]Schieltz, J. M., and Daniel I. R. 2016. Evidence Based Review: Positive Versus Negative Effects of Livestock Grazing on Wildlife. What Do we Really Know? Environmental Research Letters, IOP Publishing, 11 (11): 1–18.

[279]Sáez-Plaza, P., Michałowski, T., Navas, M. J., Asucro, A. G., and Wybraniec, S. 2013. An Overview of the Kjeldahl Method of Nitrogen Determination. Part I. Early History, Chemistry of the Procedure, and Titrimetric Finish. *Critical Reviews in Analytical Chemistry*, 43 (4):178–223.

[280]Sapkota, I. P., Tigabu, M., and Odén, P. C. 2009. Spatial Distribution, Advanced Regeneration and Stand Structure of Nepalese Sal (*Shorea robusta*) Forests Subject to Disturbances of Different Intensities. *Forest Ecology and Management*, 7(9): 1966–1975.

[281]Salazar, F., Alfaro, M., Ledgard, S., Iraira, S., and Teuber, N. 2000. Effect of the Stocking Rate and Land Slope on Nitrogen Losses to Water on a Grazed. *J. Soil Sci. Plant Nutr.*, 11 (3): 98–109.

[282]Sarmadian, F., Keshavarzi, A., and Malekian, A. 2010. Continuous Mapping of Topsoil Calcium Carbonate Using Geostatistical Techniques in a Semi-Arid Region. *Australian Journal of Crop Science*, 4 (8): 603–608.

[283]Shackleton, C., Shackleton, S., Gambiza, J., Nel, E., Rowntree, K., & Urquhart, P. 2008. Links between Ecosystem Services and Poverty Alleviation: Situation Analysis for Arid and Semi-Arid Lands in Southern Africa. Ecosystem Services and Poverty Reduction Research Programme: DFID, NERC, ESRC Cape Town, South Africa. p.187.

[284]Shankar, U. 2001. A case of high tree diversity in a sal (Shorea robusta) dominated lowland forest of Eastern Himalaya : Floristic Composition, Regeneration and Conservation. *Current Science*, 81 (7): 776–786.

[285]Shannon, C. E., and Wiener, W. 1963. *The mathematical theory of communication.* University of Illinois Press, Urbana. p.126.

[286]Schmidt, J. P., R. Moore, and M. Alber. 2014. Integrating Ecosystem Services and Local Government Finances into Land Use Planning: A Case Study of Coastal Georgia. *Landscape and Urban Planning* 122: 56–67.

[287]Schmitz, O. J., Hawlena, D., and Trussell, G. C. 2010. Predator Control of Ecosystem Nutrient Dynamics. *Ecology Letters*, (13):1199–1209

[288]Scull, P., Cardelus C. L., Klepeis, P., Woods, C. L., Frankl ,A., and J. Nyssen. 2017. The Resilience of Ethiopian Church Forests: Interpreting Aerial Photographs, 1938-2015. *Land Degradation & Development,* 28 (2): 450–458.

[289]Shelukindo, H. B., Msanya, B. M., Singh, B. R., and Munishi, P. K. T. 2014. Soil Organic Carbon Stocks in the Dominant Soils of the Miombo Woodland Ecosystem of Kitonga Forest Reserve, Iringa, Tanzania. *International Journal of Agricultural Policy and Research*, 2 (4): 167–177.

[290]Silayo, D. A., Tarimo, M. C. T., Kweka, A. R. E., and Muganda, A. M. 2006. Impacts of Human Induced Activities on Species Composition and Diversity in Miombo Woodlands of Bagamoyo District, Tanzania. *Journal of Korean Association of African Studies,* (2): 223–243.

[291]Simpson, E. H. 1949. Measurement of Diversity. Nature 163: 688.

[292]Sloan, S., and Sayer, J. A. 2015. Forest Resources Assessment of 2015 Shows Positive Global Trends But Forest Loss and Degradation Persist in Poor Tropical Countries. *Forest Ecology and Management,* 352: 134–145.

[293]Song, X. P., Huang, C., Sexton, J. O., Channan, S., and Townshend, J. R. 2014. Annual Detection of Forest Cover Loss Using Time Series Satellite Measurements of Percent Tree Cover. Remote Sensing 6 (9): 8878–8903.

[294]Smail, R. A., and Lewis, D. J. 2009. Forest-Land Conversion, Ecosystem Services, and Economic Issues for Policy: a Review. Portland, OR: U.S. Dept. of Agriculture, Forestry Services, Pacific Northwest Research Station-797, 40. Portland.

[295]Smith, P., Bustamante, M., Ahammad, H., Clark, H., Dong, H., Elsiddig, E. A., ... Tubiello, F. N. (2014). Agriculture, Forestry and Other Land Use (AFOLU). In T. Z. and J. C. M. Edenhofer, O., R. Pichs-Madruga, Y. Sokona, E. Farahani, S. Kadner, K. Seyboth, A. Adler, I. Baum, S. Brunner, P. Eickemeier, B. Kriemann, J. [189]Savolainen, S. Schlömer, C. von Stechow (Ed.), *In: Climate Change 2014: mitigation of Climate Change. Contribution of Working Group III to the Fift Assessment Report of the Intergovernmental Panel on Climate Change.* Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. 811–922.

[296]Sonneveld, B. G. J. S., Keyzer,M. A., and D. Ndiaye. 2016. Quantifying the Impact of Land Degradation on Crop Production: The Case of Senegal. *Solid Earth,* 7 (1): 93– 103.

[297]Soulé, P. T., White, P. B., and Van De Gevel, S. L. 2012. Succession and Disturbance in an Endangered Red Spruce–Fraser Fir Forest in the Southern Appalachian Mountains, North Carolina, USA. Endangered Species, 18:17–25.

[298]Stanturf, J. A., Goodrick, S. L., and Outcalt, K. W. 2007. Disturbance and coastal forests: A Strategic Approach to Forest Management in Hurricane Impact Zones. *Forest Ecology and Management*, 250 (1–2): 119–135.

[299]Sturtevant, B. R., Miranda, B. R., Wolter, P. T., James, P. M. A., Fortin, M. J., and Townsend, P. A. 2014. Forest Recovery Patterns In Response to Divergent Disturbance Regimes in the Border Lakes Region of Minnesota (USA) and Ontario (Canada). *Forest Ecology and Management*, 313:199–211.

[300]Sudduth, K. A., Kitchen, N. R., Bollero, G. A., Bullock, D. G., and Wiebold, W. J. 2003. Comparison of Electromagnetic Induction and Direct Sensing of Soil Electrical Conductivity. *Agron. J.*, 95: 472–482.

[301]Sun, X., Kay, A. D., Kang, H., Small, G. E., Liu, G., Zhou, X.,...Liu, C. 2013. Correlated Biogeographic Variation of Magnesium Across Trophic Levels in A Terrestrial Food Chain. *PLoS One*, 11: 1-8.

[302]Sundarapandian, S., and Swamy, P. S. 2013. Short-Term Population Dynamics of Tree Species in Tropical Forests at Kodayar in the Western Ghats of Tamil Nadu , India. In W. Zhang (Ed.), *Proceedings of the International Academy of Ecology and Environmental Sciences,* Madurai , India: *International Academy of Ecology and Environmental Sciences*, 3: 191–207.

[303]Swangjang, K. 2015. Soil Carbon and Nitrogen Ratio in Different Land Use. In *International Conference on Advances in Environment Research*, (87): 36–40.

[304]Syswerda, S. P., Corbin, A. T., Mokma, D. L., Kravchenko, A. N., and Robertson, G. P. 2011. Agricultural Management and Soil Carbon Storage in Surface vs . Deep Layers. *Soil Biology and Biochemistry*, 75 (1): 92–101.

[305]Szuster, B. W., Chen, Q., and Borger, M. 2011. A Comparison of Classification Techniques to Support Land Cover and Land Use Analysis in Tropical Coastal Zones. *Applied Geography*, 31 (2): 525–532.

[306]Tadesse, S A, and Kotler, B. P. 2013. The Impacts of Humans and Livestock Encroachments on the Habitats of Mountain Nyala (*Tragelaphus buxtoni*) in Munessa, Ethiopia. *International Journal of Biodiversity and Conservation*, 5 (9): 572–83.

[307]Tang, Z., Shi, C., and Bi, K. 2014. Impacts of Land Use and Land Cover Change on Ecosystem Service Values in Maqu County. *Environmental Science and Management Journal*, 13 (10): 2697–2705.

[308]Tanah, V., and Hutan, J. 2015. Soil Properties and Variation between Three Forest Types in Tropical Watershed Forest of Chini Lake , Peninsular Malaysia. *Sains Malaysiana*, 43 (11): 1635–1643.

[309]Temesgen, H., Wu, W., Shi, X., and Yirsaw, E. 2018. Variation in Ecosystem Service Values in an Agroforestry Dominated Landscape in Ethiopia : Implications for Land Use and Conservation Policy. *Sustainability*, 10 (1126): 1–20.

[310]Thomas, E., Jalonen, R., Loo, J., Boshier, D., Gallo, L., Cavers, S., … Bozzano, M. 2014. Genetic Considerations in Ecosystem Restoration Using Native Tree Species. Forest *Ecology and Management*, 333: 66–75.

[311]Thompson, I., Mackey, B., McNulty, S., and Mosseler, A. 2009. *Forest Resilience, Biodiversity, and Climate Change. A synthesis of the biodiversity/resilience/stability relationship in forest ecosystems* (Technical Series No. 43). *CBT Technical Series*. Montreal. p. 67.

[312]Thompson, J. R., Lambert, K. F., Foster, D. R., Broadbent, E. N., Blumstein, M., Zambrano, A. M. A., and Fan, Y. 2016. Four Land-Use Scenarios and their Consequences for Forest Ecosystems and Services they Provide. *Ecosphere*, 7: 1–22.

[313]Tomppo, E., Malimbwi, R., Katila, M., Mäkisara, K., Henttonen, H. M., Chamuya, N.,… Otieno, J. 2014. A Sampling Design for a Large Area Forest Inventory, *Can. J. For. Res*, 18: 931–948.

[314]Turner, M. G., Donato, D. C., & Romme, W. H. 2013. Consequences of Spatial

Heterogeneity for Ecosystem Services in Changing Forest Landscapes: Priorities for Future Research. *Landscape Ecology*, 28 (6): 1081–1097.

[315]UNDP. 2004. Extending the Coastal Forest Protected Area Subsystem in Tanzania:PIMS No: 2760 Proposal ID: 00058855, Project ID: 00073328. *Extending the Coastal Forest Area System in Tanzania*. p.107.

[316]UNEP. 2015. The Economics of Land Degradation in Africa, ELD Initiative. Bonn, Germany, www.eld-initiative.org/pdf/ p. 155. [Accessed on 17/05/2018].

[317]URT. 2013. *Sustainable Forest Management in a Changing Climate, FAO-Finland Forestry Programme- Tanzania; A fire Baseline for Tanzania*. Dar es Salaam, Tanzania. p. 63.

[318]URT. 2014. *The United Republic of Tanzania Basic Demographic and Socio-Economic Profile Report:Tanzania Mainland*. National Bureau of Statistics, Dar es Salaam, Tanzania: p. 245.

[319]URT. 2015. *National Forest Resources Monitoring and Assessment of Tanzania Mainland;Ministry of Natural Resources and Tourism, Tanzania Forest Services Agency in Collaboration With the Government of Finland and Food and Agriculture Organization (FAO) of the United Nat*. Dar es Salaam, Tanzania.p.124.

[320]URT. 2016. Tanzania in Figures, National Bureau of Statistics, Dar es Salaam, Tanzania, p.106.

[321]USDA. 2008. *Soil Quality Indicators. USDA Natural Resources Conservation Service*. Washington, DC. p.1-2.

[322]Ustuner, M., Sanli, F. B., and Dixon, B. 2015. Application of Support Vector Machines for Land Use Classification Using High-Resolution Rapid-Eye Images: A Sensitivity Analysis. *European Journal of Remote Sensing*, (48): 403–422.

[323]Van der Ploeg, S., R. S. De Groot, and Y. Wang. 2010. The TEEB Valuation Database: Overview of Structure, Data and Results, 63. Wageningen, the Netherlands: Foundation for Sustainable Development.

[324]Virapongse, A. 2017. Smallholders and Forest Landscape Restoration in Upland Northern Thailand. *International Forestry Review* 19 (4): 102–119.

[325]VonHedemann, N., and Osborne, T. 2016. State Forestry Incentives and

Community Stewardship: A Political Ecology of Payments and Compensation for Ecosystem Services in Guatemala' s Highlands. *Journal of Latin American Geography*, 15 (1): p. 29.

[326]Vourlitis, G. L., and Lobo, F. D. A. 2015. Low Soil Nutrient Availability Can Limit Tree Growth and Forest Expansion into Savanna Because Nutrient Availability May be Insufficient for Wood Production. *Plant Soil*, (389): 307–321.

[327]Wagg, C., Bender, S. F., Widmer, F., and van der Heijden, M. G. A. 2014. Soil Biodiversity and Soil Community Composition Determine Ecosystem Multifunctionality. Proceedings of the National Academy of Sciences, 111(14): 5266–5270.

[328]Waite, R., Burke, L., Gray, E., van Beukering, P., Brander, L., McKenzie, E., ... Tompkins, E.. 2014. Coastal Capital: Ecosystem Valuation for Decision Making in the Caribbean. Washington, DC: World Resources Institute. Retrived from http://www.wri.org/coastal-capital. pdf. p.78 [Accessed on 21/05/2017].

[329]Wallander, R. T., Olson, B. E., and Lacey, J. R. 1995. Spotted Knapweed Seed Viability After Passing Through Sheep and Mule Deer. *Journal of Range Management* 48:145-149.

[330]Walkley, A., and Black, I. A. 1934. An Examination of the Digestion Method for Determining Organic Carbon in Soils: Effect of Variations in Digestion Conditions and of Inorganic Soil Constituents. *Soil Sci.*, (63): 251–263.

[331]Wang, Y. P., Law, R. M., and Pak, B. 2010. A Global Model of Carbon, Nitrogen and Phosphorus Cycles for the Terrestrial Biosphere. *Biogeosciences*, 7: 2261–2282.

[332]Warinwa, F., F. Mwaura., and J. Kiringe. 2016. Land Cover Dynamics in the Kirisia Forest Ecosystem, Samburu County, Kenya. *Advances in Remote Sensing*, (5): 168–182.

[333]Wardle, D. A., Bellingham, P. J., Fukami, T., and Bonner, K. I. 2012. Soil-Mediated Indirect Impacts of an Invasive Predator on Plant Growth. *Biol.Lett*,(8): 574–577.

[334]Widianingsih, N. N., Theilade, I., and Pouliot, M. 2016. Contribution of Forest

Restoration to Rural Livelihoods and Household Income in Indonesia. *Sustainability*, 8 (835): 1-22

[335]Webb, E. L., Bult, van de,M., Chutipong, W., and Kabir, M. E. 2007. *Species composition and forest structure of tree communities in four permanent forest monitoring plots in the Ta"u unit of the National Park of American Samoa*. 3190 Maile Way. Technical report No.138. p. 40.

[336]Wekesa, C., Leley, N., Maranga, E., Kirui, B., Muturi, G., Mbuvi, M., and Chikamai, B. 2016. Effects of Forest Disturbance on Vegetation Structure and Above-Ground Carbon in Three Isolated Forest Patches of Taita Hills. *Open Journal of Forestry*, 6: 142–161.

[337]Wenhua, L. 2004. Degradation and Restoration of Forest Ecosystems in China. *Forest Ecology and Management*, 201 (1): 33–41.

[338]Willcock, S., Hooftman, D., Sitas, N., Farrell, P. O., Hudson, M. D., Reyers, B., and J. M. Bullock. 2016. Do Ecosystem Service Maps and Models Meet Stakeholders' Needs? A Preliminary Survey across Sub- Saharan Africa. *Ecosystem Services* 18: 110–117.

[339]Whitworth, A., Downie, R., May, Von,R., Villacampa, J., and Macleod, R. 2016. How Much Potential Biodiversity and Conservation Value Can a Regenerating Rainforest Provide? A "Best - Case Scenario" Approach From the Peruvian Amazon . *Tropical Conservation Science*, 9 (1): 224–245.

[340]World Bank. 2016. Managing Coasts With Natural Solutions: Guidelines for Measuring and Valuing the Coastal Protection Services of Mangroves and Coral Reefs. In: M. W. Beck and G-M. Lange, (eds) Wealth Accounting and the Valuation of Ecosystem Services Partnership (WAVES), World Bank, Washington, DC, p.167.

[341]World Wildlife Fund (WWF). 2014. Tanzania's Wildlife Management Areas A 2012 Status Report, Dar es Salaam. p. 70.

[342]Wu, K., Ye, X., Qi, Z., and Zhang, H. 2013. Impacts of Land Use / Land Cover Change And Socioeconomic Development on Regional Ecosystem Services: The Case of Fast-Growing Hangzhou Metropolitan Area, China. *Cities*, (31): 276–284.

[343]Xia, S., Chen, J., Schaefer, D., and Detto, M. 2015. Scale-Dependent Soil Macronutrient Heterogeneity Reveals Effects of Litterfall in a Tropical Rainforest. *Plant Soil*, (391): 51–61.

[344]Xing, W. U., Zongshan, L. I., Bojie, F. U., Fei, L. U., Dongbo, W., Huifeng, L. I. U., and Guohua, L. I. U. 2014. Effects of Grazing Exclusion on Soil Carbon and Nitrogen Storage in Semi-Arid Grassland in Inner Mongolia, China. *Chin. Geogra. Sci*, 24 (4): 479–487.

[345]Xu, X., Guishan,Y., Yan, T., Xuguang ,T., Hong ,J., Xiaoxiang ,S., Qianlai ,Z., and Hengpeng ,L., 2017. Impacts of Land Use Changes on Net Ecosystem Production in the Taihu Lake Basin of China from 1985 to 2010. *Journal of Geophysical Research* 122 (3): 690–707.

[346]Xuluc-Tolosaa, F. J., Vestera, H. F. M., Ramı'rez-Marcialb, N., Castellanos-Alboresb, J., and Lawrencec, D. 2003. Leaf Litter Decomposition of Tree Species in Three Successional Phases of Tropical Dry Secondary Forest in Campeche, Mexico. *Forest Ecology and Management*, 174: 401–412.

[347]Yang, Y. H., Fang, J. Y., Guo, D. L., Ji, C. J., and Ma, W. H. 2010. Vertical Patterns of Soil Carbon, Nitrogen and Carbon : Nitrogen Stoichiometry in Tibetan Grasslands. *Biogeosciences*, (7): 1-27.

[348]Yavitt, J. B., Harms, K. E., Garcia, M. N., Wright, S. J., He, F., and Mirabello, M. J.2009. Spatial Heterogeneity of Soil Chemical Properties in a Lowland Tropical Moist Forest, Panama. *Australian Journal of Soil Research*, (47): 674–687.

[349]Yawson, D. O., Kwakye, P. K., Armah, F. A., and Frimpong, K. A. 2011. The Dynamics of Potassium (K) in Representative Soil Series of Ghana. *ARPN Journal of Agricultural and Biological Science*, 6 (1): 48–55.

[350]Yengoh, G. T., Dent, D., Olsson, L., Tengberg, A. E., and Tucker, C. J. 2014. *The use of the Normalized Difference Vegetation Index (NDVI) to Assess Land Degradation at Multiple Scales: A Review of the Current Status, Future Trends, and Practical Considerations*. Lund University Center for Sustainability Studies

(LUCSUS), and The Scientific and Technical Advisory Panel of the Global Environment Facility (STAP/GEF). Lund, Sweden. p.47.

[351]Yeshaneh, G. T. 2015. Assessment of Soil Fertility Variation in Different Land Uses And Management Practices in Maybar Watershed. *International Journal of Environmental Bioremediation and Biodegradation*, 3 (1): 15–22.

[352]Yirdaw, E., Tigabu, M., and Monge, A. 2017. Rehabilitation of Degraded Dryland Ecosystems – Review. *Silva Fennica*, 51 (1): 1–32.

[353]Yirsaw, E., Wu,W., Temesgen,H., and Bekele.B., 2016. Effect of Temporal Land Use and Land Cover Changes on Ecosystem Services Value in Coastal Area of China: The Case of Su-Xi-Chang Region. *Applied Ecology and Environmental Research* 14 (3): 409–422.

[354]Yirsaw, E.,Wu,W., Shi,X., Temesgen, H., and Bekele .B., 2017. Land Use and Land Cover Change Modeling and the Prediction of Subsequent Changes in Ecosystem Service Values in the Coastal Area of China, the Su-Xi-Chang Region. Sustainability, 9: 1–17.

[355]York, N., Science, E., Drive, F., and York, N. 2008. Influences of a calcium Gradient on Soil Inorganic Nitrogen in the Adirondack Mountains, New York, 18 (7): 1604–1614.

[356]Yun-guo, L. I. U., Xiao-xia, Z., Li, X. U., and Da-lun, T. 2011. Impacts of Land-Use Change on Ecosystem Service Value in Changsha , China. *J. Cent. South Univ. Technol.*, (18): 420–428.

[357]Zhang, H., John, R., Peng, Z., Yuan, J., Chu, C., Du, G., and Zhou, S. 2012. The Relationship between Species Richness and Evenness in Plant Communities Along a Successional Gradient : A Study from Sub-Alpine Meadows of the Eastern Qinghai-Tibetan Plateau, China. *PLoS ONE*, 7 (11): 1–9.

[358]Zhang, W., Liao, A., Peng, S., Zheng, X., and Li, M. 2015. Check and modification of GlobeLand 30 with MODIS NDVI. In H. L. Chunhua Shen, Weiping Yang (Ed.),

Proc. SPIE 9675, AOPC: Image Processing and Analysis, Beijing, China: National Geomatics Center of China: October (8). p. 22.

[359]Zhang, W., Lu, Z., Yang, K., and Zhu, J. 2016a. Impacts of Conversion From Secondary Forests to Larch Plantations on the Structure and Function of Microbial Communities. *Applied Soil Ecology*, (2568). p. 11.

[360]Zhang, Y., Zhao, L., Liu, J., Liu, Y., and Li, C. 2016b. The Impact of Land Cover Change on Ecosystem Service Values in Urban Agglomerations Along the Coast of the Bohai Rim, China. *Sustainability*, 7 (8): 10365–10387.

[361]Zhao, B., Kreuter, U., Li, B., Ma, Z., Chen, J., and Nakagoshi, N. 2004. An Ecosystem Service Value Assessment of Land-Use Change on Chongming Island , China. *Land Use Policy*, (21): 139–148.

[362]Zhao, F. R., Meng, R., Huang, C., Zhao, M., Zhao, F. A., Gong, P.,...Zhu, Z. 2016. Long-Term Post-Disturbance Forest Recovery In the Greater Yellowstone Ecosystem Analyzed Using Landsat Time Series Stack. *Remote Sensing*, 8 (11): 11.

[363]Zhong, L., Du, R., Ding, K., Kang, X., Li, F. Y., Bowatte, S.,... Wang, S. 2014. Effects of Grazing on N_2O Production Potential and Abundance of Nitrifying and Denitrifying Microbial Communities in Meadow-Steppe Grassland In Northern China. *Soil Biology and Biochemistry*, (69): 1–10.

[364]Ziter, C., Graves, R. A., and Turner, M. G. 2017. How Do Land-Use Legacies Affect Ecosystem Services in United States Cultural Landscapes? *Landscape Ecology*, 32 (11): 2205–2218.

[365]Zhou, J., Sun, L., Zang, S. Y., Wang,K., Zhao ,J. Y., Li ,Z. X., and Liu.X. R. 2017. Effects of the Land Use Change on Ecosystem Service Values. *Global Journal of Environmental Science and Management* 3 (2): 121–130.

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Published Papers during Study Program

1.1. International Peer Reviewed Journals

Ligate, E. J., Chen, C. and Wu, C. 2018. Evaluation of tropical coastal land cover and land use changes and their impacts on ecosystem service values. *Ecosystem Health and Sustainability*, 4(8), 188–204.

Ligate, E. J., Chen, C. and Wu, C. 2018. Investigation of tropical coastal forest regeneration after farming and livestock grazing exclusion. *Journal of Forestry Research*. Received: 12 July 2017 / Accepted: 6 February 2018 available at https://doi.org/10.1007/s11676-018-0792-5.

Ligate, E. J., Chen, C. and Wu, C. 2018. Estimation of Carbon Stock in the Regenerating Tree Species of the Intact and Disturbed Forest Sites in Tanzania, *International Journal of Environment and Climate Change*, 8 (2): 80-95.

Ligate, E. J., Chen, C. and Wu, C. 2018. Evaluation of Soil Fertility Status Based on CEC and Variation across Disturbed and Intact Tropical Coastal Forests Sites in Tanzania. *Asian Journal of Environment and Ecology*, 6 (2): 1-12.

Ligate, E. J., Chen, C. and Wu, C. 2017. The Status of Forest Ecosystem Services and Their Management : The Case of Uzigua Forest Reserve in Tanzanian Coastal Forests. *Natural Resources and Conservation*, 5 (2): 21–32.

Msola, D., **Ligate, E. J**, Chen, C. and Wu, C. 2017. Contribution of Tanzania Southern Highlands Forest Diversity to Household Income and Food Supplements: The Case of Mufindi District in Tanzania. *Journal of Geography, Environment and Earth Science International*, 9(4), 1–12.

Ligate, E. J., Kitila, M.M, Chen, C. and Wu, C. 2017. Impacts of Salt Water Intrusion on Maize (*Zea mays*) and Rice (*Oryza sativa*) Production under Climate Change Scenarios in Bagamoyo District-Tanzania. *Universal Journal of Agricultural Research*, 5(2), 148–158.

1.2.Submitted Papers

Ligate, E. J., Chen, C. and Wu, C. 2018. Evaluation of carbon, nitrogen and phosphorus status across forest sites subjected to farming and livestock grazing disturbances in the tropics.

Applied and Environmental Soil Science. Ref. No. 1815048 (Submitted in March 2018).

Ligate, E. J., Chen, C. and Wu, C. 2017. Mapped electrical conductivity in relation to soil texture and bulk density across intact forest, crop-agriculture and livestock disturbed sites: The case of Uzigua coastal forest in Tanzania, *International Journal of Environmental Sciences* (Submitted in July 2018).

Prepared Manuscript For Submission

Ligate, E. J., Chen, C. and Wu, C. (undated) Multivariate Correlation between Soil Properties and Tree Structures of the Disturbed Coastal Forests Ecosystems in Tanzania

Other Academic Achievements during the Program

2018.11.14: Awarded a certificate of joining as an **Academic Editor** of Advances in Research Journal from 2018 to 2022.

2017-2018: Awarded Eleven (11) Certificates of Excellence in Reviewing Manuscripts submitted for Publications in International Peer Reviewed Journals [1. Advances in Research, 2. Asian Journal of Advances in Agricultural Research, 3. Asian Journal of Environment and Ecology, 4. Geography, Environment and Earth Sciences International, 5. Experimental Agriculture International and 6. Current Journal of Applied Sciences and Technology].

Annex 1: Households Questionnaire

Introduction

My Name is Elly J. Ligate, a Tanzanian PhD student (Reg. # 2151930001) at Fujian Agriculture and Forestry University, Fuzhou, Fujian- China. I would like to invite you for this interview, which will take about 30 to 90 minutes. This survey aim to collect data, which will contribute to produce a PhD thesis for partial fulfilment of completion of a PhD in Ecology as offered by Fujian Agriculture and Forestry University- China.

Consent section

In this interview, I would like to learn from your experience about land uses, coastal forests resources use and management, household's socioeconomic activities, which influence the structure and services of coastal forests and restoration interventions of coastal forest ecosystems. This interview is neither examination nor test and thus there is no wrong answer (s) for any question. Feel free to answer the questions and in some cases where you are not feeling okay, be free to skip or suspend the interview. The information collected through this interview will be kept confidential. Indeed, the results from all the interviews will be generalized in the report where there will be no any name of identity of the respondents. Before we proceed I would like to ask if you are willing to participate: Are you willing or not?________________For further contacts use: Phone: +8617720805052; Email address: ligateelly@yahoo.com

Section 1: Demographic Information

1. Sex of respondent a. Male b. Female	2. Head of the household a. Male b. Female	3. Age of respondents a. 18 to 36 b. 37 to 54 c. 54 to 72 d. 73 to 90 e. Above 90	4. Level of Education a. None b. Primary c. Secondary education d. College education e. University education f. Any other training

5. Household size a. 1-5 members b. 6-10 members c. More thatn10 members	6. Livelihoods and income a. Forests resources b. Agriculture c. Livestock d. Business e. Food security	7. Source of food per house hold a. Food grown on land owned and cultivated by household b. Food grown on land but not owned by household c. Food purchased from market d. Food from forests (mushroom, fruits, bush meat)	8.How long have you lived in this area a. < 5yrs b. 5-10 yrs c. 11-15 yrs d. 16-20 yrs e. 21-25 yrs f. 26-30 yrs

Section 2: Information about availability of forest resources and utilization

1.Forests resources	2. Forests resources	3. User Rights	4.Types of energy sources
a. Forests food sources b. Herbs c. Spices d. Traditional medicinal plants	a. Fire woods b. Construction material c. Plant medicines d. Charcoal e. Honey f. Fodder g. Bush meat h. Wood carving	a. Open access b. No right c. Customary right d. Rent e. Product lease	a. Firewood b. Charcoal c. Gas d. Kerosene e. Electricity f. Other…………

5. Energy source for Cooking	6. Energy source for Lighting	7. Energy source for Heating	8. Where are users of forests resources come from?
a. Firewood b. Charcoal c. Gas d. Kerosene e. Electricity f. Other…	a. Firewood b. Charcoal c. Gas d. Kerosene e. Electricity f. Other…	a. Firewood b. Charcoal c. Gas d. Kerosene e. Electricity f. Other…	a. From within the village b. From other villages c. From other districts

8. Have you come across any problems related to:- a. The use of land b. The use of water c. Use of forests resources	9. Impacts of forests degradation a. Climate change b. Loss of water sources c. Loss of windbreak d. Soil erosion e. Loss of natural beauty for ecotourism		

Section 3: Forest management and restoration practices

	1	2	3	4
1. How your life depends on forest resources?	Not at all	A little	Somehow	Highly dependent
2. In your opinion what is the trend of forest your area?	Not changed at all	There is small changes	Changed somehow	Changed largely
3. How human activities influence forest changes?	Not at all	Very little effect	Somehow affect	Largely affect
4. Are there any impacts of forest disturbances on your life?	No at all	Very little impacts	There are somehow impacts	There is high impacts
5. How would you find yourself without forests?	Nothing	A little vulnerable	Somehow venerable	Highly vulnerable
6. Have you participated in forest	Never	A little	Somehow	Highly

restoration activities in any way?				participated
7. Do you consider that restoration rejuvenates forests?	Not at all	A little	Somehow	Highly rejuvenates
8. How do you rank your involvement in forest management?	Not involved at all	Very low	Involved somehow	Involved fully
9. Does local community know forest restoration responsibility?	Not at all	Lightly know	Somehow know	Highly know
10. Are you aware about any land use plan for human activities and forest conservation in your village?	Not at all	A little aware	Somehow aware	Highly aware
11. Are you aware with laws and regulations enforcements on forest management?	Not at all	Lightly aware	Somehow aware	Highly aware
12. Is there any incentives for participating in forest restoration activities	Not at all	A little	Somehow	High incentives
13. Are the tree planting materials available	Not at all	A little	Somehow	Highly available
14. Does ownership of land influence forest restoration efforts?	Not at all	A little	Somehow	Highly influence

Section 4: Guiding questions to discuss about forest management

1. What is the trend of forest in your area?
2. What activities contribute to deforestation in your area?
3. How are you involved in restoration of the disturbed sites of forests?
4. Why there is continuous deforestation in your area?
5. What efforts should be in place to overcome forest disturbance and deforestation?
6. What livelihood activities should be promoted besides those forestry resources based ones?
7. What is the management plan is in place for managing forests in these villages?
8. Who should be involved in forests management to control deforestation?
9. What tree species are mostly harvested?
10. What are the major reasons behind number 9 above?
11. What kind of tree species are useful in restoration of disturbed sites?
12. What should be the alternatives to tree use from forest?
13. How land ownerships and tree planting experienced in your village?
14. Are there any rules and regulations that govern harvesting and restoration of forests resources at the village?
15. What are the penalties?
16. How human exclusion affects forests and livelihood of the locals?

--The End--

Annex 2: Check List for Key Informants and FGD

1. How local community, leaders and experts gauge the trend of land cover and land use change, and ecosystems disturbances along the coastal zone of Tanzania?
2. How sets of socioeconomic activities have taken place and what sets have changed over the past sixteen years to affect coastal ecosystems in Tanzania?
3. Why the degradation of coastal ecosystems increased in the past sixteen years?
4. What strategies are in place to address forest disturbances along the coastal zone of Tanzania?
5. What factors affect coastal forest restoration interventions in Tanzania?

----------The End----------

Annex 3: Field Photo

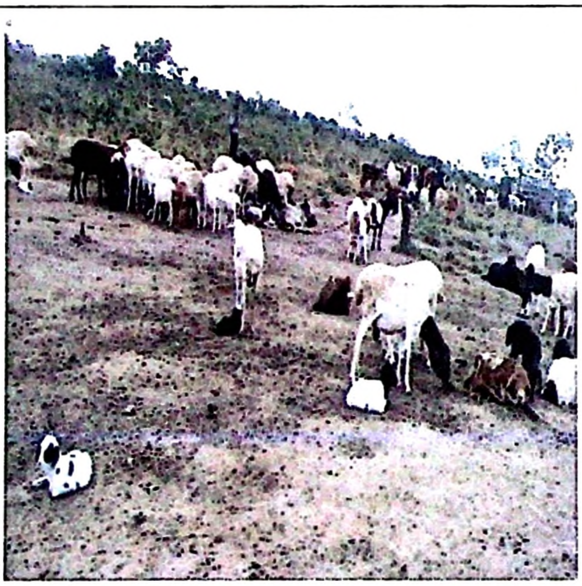
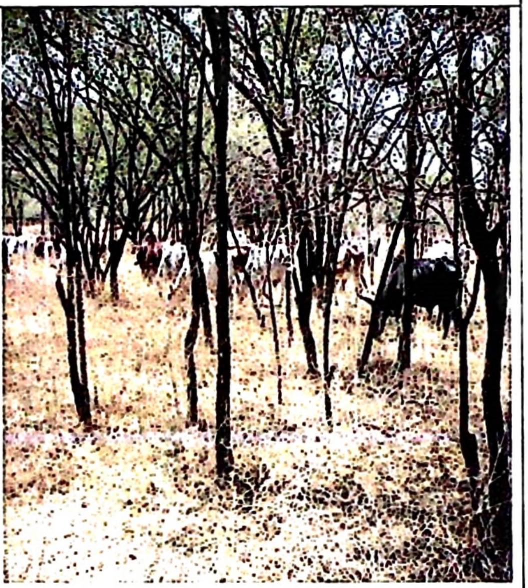
a. Interview photo at Kwamduma

b. Interview photo at Mpaji

c. Interview hoto at Kwaruhombo

d. Focus group discussion

Annex 3: Continue

	
e. Triangulation workshops members	f. Abandoned farm at Mbwewe
	
g. Grazing sheep at Mpaji	h. Grazing cattle at Mpaji

Annex 3: Continue

i. Burn area at Changalikwa village

j. Tree cut down for farm preparation at Changalikwa

k. Salt evaporation pans at Bagamoyo

l. Wooden charcoal ruminants at Kwang'andu

NB. These Photos were taken during data collection in 2016-2017.