EVALUATION OF THE QUALITY OF RECLAIMED SOILS AT ANGLOGOLD ASHANTI IDUAPRIEM MINES, GHANA

BY

JUSTICE KWESI BADU

(10507248)

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DECLARATION

I hereby declare that this thesis "Evaluation of the quality of reclaimed soils at AngloGold Ashanti Iduapriem Mines, Ghana" has been written by me and that it is the record of my own research work. It has neither in whole nor in part been presented for another degree elsewhere. The use of all materials from other sources have been properly and duly acknowledged.

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Justice Kwesi Badu
(Student)

20/10/2025
.....
Date

.....
Prof. G. N. N. Dowuona
(Principal Supervisor)

20/10/2025
.....
Date

.....
Prof. T. A. Adjadeh
(Co-Supervisor)

20/10/2025
.....
Date

.....
Prof. D. Neina
(Co-Supervisor)

20/10/2025
.....
Date

DEDICATION

I dedicate this PhD thesis to the loving memory of my mother, Maame Akosua Adum (aka Auntie Asi), who died when I was on the field collecting data for this research work.

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ABSTRACT

The assessment of reclamation status must include soil biogeochemical indicators that are sensitive to management. This study was conducted in the Iduapriem concession, Tarkwa to determine the (a) biodiversity of the study sites, (b) characteristics of the soil profiles of the reclaimed sites and the natural forests, (c) quality of rehabilitated mined soils. Therefore, six mined sites viz B1 Pit (Site B), ITSF (Site C), OTSF (Site D), B1S/E (Site E-9), B2&3 (Site E-18), and B1N (Site E-21) under four different modes and varied ages of reclamation were selected for the study. The Neung natural forest (Site A) was used as the control site for the study. For the biodiversity assessment, the floristic and earthworm composition in each site was estimated using the Shannon-Wiener, Margalef's, Pielou equitability, and the Czekanowski similarity coefficient indices. A 100 cm deep profile was dug at each site and samples were taken at 10 cm intervals. Each profile was described morphologically and their physicochemical characteristics were compared with the control site. Soils were also sampled from all selected sites at 0-20 cm depth using an auger for soil quality assessment. The collected soil samples were transferred to the laboratory and subjected to the physical, chemical, biological analyses and principal component analysis (PCA) to determine soil quality index. The highest Shannon-Wiener diversity index value of 3.13 was obtained at Site A (natural forest). However, at Site B had an index value of 2.64 which came close to Site A. Whereas Site C had an index value of 1.19, Site D had an index value of 1.62. Among the same mode of reclamation, Site E-9, Site E-18 and Site E-21 had an index value of 1.23, 1.82 and 1.76 respectively. Whereas a total number of 954 earthworms were collected from the study area, 30 were collected from Site A. A population of 540 earthworms were collected from Site B, 43 from

Site C, while Site D recorded 129 earthworms. However, Sites E-9, E-18 and E-21 produced 6, 5 and 201 earthworms, respectively. The Shannon-Wiener earthworm diversity index was in order of Site B (0.89) > Site E-18 (0.82) > Site A (0.58) > Site D (0.57) > Site C (0.48). However, the index value for Sites E-18 and E-21 was 0 for each site. The use of heavy mining equipment in transporting, dumping and spreading the reclamation materials, compacted the soils. This resulted in higher bulk densities, poor soil structure and limited root distribution within profiles of the reclaimed sites. However, the reverse was the case in the natural forest. Except for the tailing provenance reclaimed sites (Sites C and D) that had a neutral pH, the other reclaimed soils and the natural forest soil were in the range of strongly acid to extremely acid condition. The natural forest recorded the highest soil organic matter, total carbon, total nitrogen. However, in the reclaimed sites, it occurred in the following order: Site B > Site E-21 > Site E-9 > Site E-8 > Site D > Site C. Considering the vertical cross section of the individual profiles of the reclaimed soils, it is clear to conclude that the similar and uniform properties observed from the reclaimed profiles is due to cumulative impact of anthropogenic activities rather than pedogenic. The, PCA selected bulk density, aggregate stability, CF, Exch. acidity, total P, C: N, C_{mic} and SFD as the final MDS. However, the SQI for the natural forest was 0.687. Whereas in the reclaimed sites, SQI was 0.527 at Site B, Site C had 0.197 and Site D had a quality index of 0.310. At Site E, the value of SQI increased with reclamation age which occurred in the following order: Site E-21 (0.589) > Site E-18 (0.424) and Site E-9 (0.320). The mode of reclamation had significant impact on the physicochemical and biological properties of the reclaimed soils than the age of reclamation. Hence a better performance of the conventionally reclaimed site (Site B) than the haphazardly reclaimed sites. The findings of the study indicate that topsoil management and replacement is very crucial in the recovery of degraded mined lands.

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LIST OF ABBREVIATIONS

AAIL	AngloGold Ashanti Iduapriem Limited
AMD	Acid mine drainage
AWC	Available water capacity
B1 N	Block 1 North
B1 S/E	Block 1 South-East
B 2&3	Block 2 and 3
B1 Pit	Block 1 Pit
CEC	Cation exchange capacity
DHA	Dehydrogenase activity
ITSF	Interim tailings storage facility
MDS	Minimum data set
NF	Natural Forest
OTSF	Old tailings storage facility
PCA	Principal component analysis
SOC	Soil organic carbon
SOM	Soil organic carbon
SQI	Soil quality index
SSSA	Soil Science Society of America
WRD	Waste rock dump

CHAPTER ONE

GENERAL INTRODUCTION

1.1 Background

From its early and primitive antecedents to the present-day Ghana, mining has evolved gradually in the direction of increasing the number of companies and the complexities of the methodologies and the machinery used. Currently, there are fourteen large-scale gold, one bauxite and one manganese mining companies operating in Ghana (Ghana Chamber of Mines, 2018). Diamond, which used to be mined on large scale in Akwatia has dwindled and it is now confined to the artisanal and small-scale sector. However, all these mining companies practice surface mining, which includes open-pit or strip mining. Aside from the mineral resources mined on a large scale, the country also has other industrial minerals such as iron ore, mica, feldspar, quartz, silica, limestone, kaolin and salt which are mined on small-scale (Ghana Minerals Commission, 2006; Minerals and Mining Policy of Ghana, 2014; Ministry of Lands and Natural Resources, 2018). These minerals are situated within the western portion of the country, where the Proterozoic Birimian and Tarkwaian belts, in strips, intersect and extend due west to Senegal and Mauritania and north into Burkina Faso (Wright, 1985; Hilson, 2002a).

According to the Ministry of Lands and Natural Resources, mining companies have been granted concessions on about 30% of Ghana's landmass (World Rainforest Movement, 2008) to extract precious minerals. Today, Ghana is Africa's largest gold producer, with production increasing by 12% in 2018 to more than 4.8 million ounces, surpassing South Africa's output of 4.2 million ounces for the first time (Ghana Chamber of Mines, 2018). The mining industry largely contributes to the Ghanaian economy through payments of corporate taxes, royalties, fees, and income taxes on both salaries and wages of employees and dividends declared (UNDP, 2018; GSS, 2019). The

industry also accounts for around 5.2% of the country's GDP and provides approximately 40% of total goods exports, with gold accounting for over 90% of total mineral exports (Ghana Minerals Commission, 2006; Aryee, 2012). The mining sector also provides direct and indirect jobs to people within their catchment areas. As part of their corporate social responsibility (CSR) to communities in which they operate, mining companies initiate and fund some local development projects, in areas like health, education, roads, business development, water, and sanitation. Despite these roles the mining sector plays in the country's socioeconomic growth, gold mining in Ghana is replete with reports of the destructive nature of mining operations in almost all parts of Ghana. These disturbing trends are not limited to the past, but currently prevalent due to the mode of operation. Modern mining is a highly sophisticated operation, which involves the simultaneous manipulation of the mining environment which unavoidably causes substantial destruction to the environment and the land on which it takes place (Bai et al. 2006; Topp et al. 2010). Land degradation emanating from mining activities and their consequential soil, water and air pollution have been reported by many authors.

On a global scale, mining and its allied activities have generated around 10% of land destruction (Oldeman, 1994; Koranteng, 2007). A study conducted by Akabzaa and Darimani (2001) revealed that over 70% of landmass in the Tarkwa area has been leased as concessions to mining companies where vast areas of land have been destroyed to pave way to surface mining activities. Schueler et al. (2011) claimed that surface mining in the Western Region of Ghana alone has resulted in 58% deforestation and 45% loss of agricultural land. A situation that is causing displacement of communities, the downtrend in agriculture in the area while fueling conflicts and agitations over land use between the custodians of the lands and mining companies. Within the Iduapriem concession, vast areas of arable land have been converted to mining pits, tailing storage facility,

waste rock dumping sites and host of reconstructed landscapes.

Surface mining is an anthropogenic activity that alters the natural soil environment (McSweeney and Jansen, 1984) culminating in a change in land use from the natural ecosystems to a mining landscape and then to a post-mining landscape (Macdonald et al. 2015). The focus of modern large-scale mining projects is now shifting to surface mining (IFC/World Bank 2007). According to Ramani (2012), over 95% of non-metallic minerals, over 90% of metallic minerals, and over 60% of coal are mined via surface mining. Heavy earth moving machines and blasting generate over 83% of ore and waste materials per year. Literature reveals that, pre-and post-mining landscapes often differ dramatically from each other, primarily in terms of shape and form, biogeochemical and pedological properties, land use capability as well as flora and fauna composition. Surface mining destroys vegetation, removes soil and overburden, changes the landform, and disturbs surface and underground hydrology (Amegbey, 2001; Herath et al. 2009; Shrestha and Lal. 2011), causing subsidence and erosion (Hilson, 2002b., Hamanaka et al. 2014) of the area in which it takes place.

Soil disturbances caused by surface mining especially open-pit systems inevitably results in loss of soil strength and structure (Jansen, 1981), higher bulk density due to compaction (Shukla et al. 2003), decrease in soil organic matter content (Lal et al. 1998), while disrupting the aesthetics of the landscape. The phenomenon of acid mine drainage (AMD) and heavy metals emanating from mining activities contaminate the soil ecology, surface and subsurface water, and the downstream aquatic ecosystem (Nemeron, 1978). According to the Chamber of Mines of South Africa (2008), the devastating effects of surface mining on the soil, vegetation, fauna, hydrologic regimes at the surface and subsurface make gold mining a prime contributor of human-caused ecosystem degradation. However, Shrestha and Lal (2011) noted the degradation effects are mitigated by

reclamation. Therefore, reclamation strategies should bring about improvements in soil quality (Tripathi et al. 2016) and the ecosystem. These are fundamentals in assessing the performance of the reclamation trajectories.

Asensio et al. (2013); Frimpong et al. (2014); and Mukhopadhyay et al. (2016) underscore the importance of frequently assessing the success and the ability of reclaimed soils to support crops and forest production. The success of every reclamation process can be assessed in various ways however, soil quality is the most crucial indices of returning the structure and functions of ecosystems after mining (Liu et al. 2017). The assessment of soil quality is crucial in the soil reclamation process (Zhao et al. 2013) and sustainable land use management (McGrath and Zhang, 2003).

1.2 Problem statement

Ghana is endowed with many minerals and other geological materials of economic value. These are natural gifts that may be exploited, marketed, and utilized to benefit the citizens of the country (Eggert, 2002). However, their extraction requires destruction of large areas of natural vegetation and arable lands which hitherto served as a source of livelihood for the mining communities (Akabzaa and Darimani, 2001).

Available records underscoring the economic relevance of the mining industry to the growth of the Ghanaian economy demands its establishment. However, while some people see the mining sector as a blessing and therefore advocating for its existence, others see it as a bane. Those who see mining as a bane therefore, wonder if mining has profited the state and the communities within which they operate. Ross (2001) advocated that. developing nations should do away with the export-oriented extractive industries and rather concentrate on agricultural and manufacturing

sectors that provide a more sustainable and balanced benefits to the people. As a developing nation, should we exploit this gift of nature to better our lives or allow it to remain in the earth intact and remain in poverty? Nonetheless, the general desire has been to see the degraded environment back to its original ecosystem after exploitation of the minerals. As a rule, mining companies, are obliged to adhere to reclamation and environmental codes to ensure that the area mined is eventually transformed back into its original state (ICMM, 2008). However, a major challenge associated with mining in Ghana relates to postmining management after decommissioning. The large-scale mines often embark on systematic reclamation where various trees are grown to restore the site to its pre-mining status. The sequence of activities before and after mining operations affect the nature and properties of the soil during and after reclamation. The reclamation process involves earthworks, application of oxides materials/subsoil, addition of topsoil, and soil amendment, followed by establishing 60% seedlings of exotic and 40% indigenous plant species. It also involves monitoring, maintenance and measurement of success criteria (EPA-Ghana, 1994) to assess reclamation success.

In assessing reclamation success, most plantation managers attribute lush growth of trees and plants on reclaimed lands to the restoration of degraded lands and higher fertility status. However, Asensio et al. (2013) argue that the success of revegetation of a degraded mine site alone, cannot be used as a yardstick to predict soil quality to a point that can support most plants growth and development. Unlike natural forest soils and agricultural lands, the nature and properties of mined soils would have been drastically altered (Insam and Domsch, 1988; Mukhopadhyay et al. 2016). Therefore, development of the soil quality criteria should include soil physicochemical and biological indicators (Dick, 1994; Doran and Zeiss, 2000; Islam and Weil, 2000; Herrick et al. 2002; Aparicio and Costa, 2007; Mukhopadhyay et al. 2016). Many soil quality indices using

varied soil properties have been proposed (i.e., Karlen et al. 1997; Andrews et al. 2002; Amacher et al. 2007; Zornoza et al. 2007; Rossi et al. 2009; Sinha et al. 2009; Asensio et al. 2013; Mukhopadhyay et al. 2013; Zhao et al. 2013). However, plantation managers hardly pay attention to soil properties which are the critical indicators of soil quality (Mohd-Aizat et al. 2014; Malik et al. 2015) but rather base their judgement only on vegetation growth.

The research questions are:

1. What biogeochemical soil properties and soil quality indices could be used to evaluate the effectiveness of mine soil reclamation?
2. Does age of reclamation affect soil quality?

Soil is the most important medium for vegetation establishment on degraded mine lands, and it influences the trajectory of reclamation and post reclamation land use (Liu et al. 2017). However, its properties change steadily with time after reclamation. This change crucially affects the functions of the soil and is influenced by different land-use types, reclamation strategies, and other biotic and abiotic factors. For this reason, knowledge about the quality of soil after reclamation of degraded lands is equally crucial in guiding future reclamation programs (Dutta and Agrawal, 2003; Courtney et al. 2009). As the quest and knowledge of extracting gold from land increases and becomes more sophisticated, more agricultural lands are likely to be mined. Thus, a more robust approach to assess the soil quality of reclaimed gold mined soils becomes imperative.

1.3 Objectives of the study

The objectives of this study are to:

a. Assess the biodiversity of reclaimed gold mined sites.
b. Determine the characteristics of soil profiles at reclaimed sites.
c. Determine the quality of reclaimed gold mined soils.

CHAPTER TWO

LITERATURE REVIEW

2.1 Brief history of gold mining in Ghana

Gold mining in Ghana has a long-standing history with pre-colonial antecedent up until the present day (Aryee et al. 2003). From the pre-colonial period to modern-day Ghana, the country has produced a considerable amount of gold in the world (Hilson, 2002a). The local tribes of the ancient Ghana and Gold Coast, for centuries, traded with Arab merchants, before the invasion of the Europeans on the Coast. Quintessentially, the interest of the European powers in modern-day Ghana's Gold Coast was to battle for control of West Africa's gold monopoly. The Portuguese were the first European power to arrive in Amankwaakrom the present-day Elmina in 1471, followed by the Dutch and English. However, after centuries of power battles between various European powers, the British seized control of Ghana in the mid-1800s and established the Gold Coast colony in 1874. Shortly after the formation of the British Gold Coast colony in the nineteenth century, the British took control of the mining industry in Ghana (Hilson, 2002a) and imposed colonial rule. However, in 1957 the British colony of the Gold Coast became the independent nation of Ghana. Subsequently the then regime acquired all the assets of British mining companies which gave the government majority ownership (55%) in all mining operations.

However, successive governments after independence became more involved in the mining sector beginning with the establishment of the State Gold Mining Corporation in 1961 and the 1972 Mining Operations Decree (NRCD 132). From the 1970s and into the early 1980s, gold output declined steadily. However, after 1983, the PNDC government implemented a series of measures which sought to increase output in the mining sector as part of its Economic Recovery Programme (ERP). After the implementation of the Economic Recovery Programme, Ghana attracted foreign

investments and pushed towards privatization and state divestiture. After a protracted review from the early 2000s, the governing legislation for minerals and mining industry was updated in 2006 by enacting Act 703. The Minerals and Mining Act, 2006 (Act 703), categorizes mining operations in Ghana into large-scale and small-scale (including artisanal) mining. The large-scale industry is largely foreign-owned, but the Ghanaian government, as per the Minerals and Mining Act, holds a 10 percent free carried interest in all mining undertakings, and reserves the right to acquire an additional interest on terms to be agreed on by the government and the mining company. However, the small-scale mining industry is reserved for Ghanaians.

2.2 Moderation of mining activities in Ghana

Mining operations in Ghana are moderated under a legal framework covering core mining legislation and laws as well as policies on the environment, business, investment, spatial planning, local government, and other general legislation. The Ministry of Lands and Natural Resources and the Minerals Commission remain the two government bodies with the primary responsibilities of administering the mining industry in Ghana. However, the principal environmental laws applicable to the mining industry in Ghana are the Environmental Protection Agency Act 1994 (Act 490) and the Environmental Assessment Regulations 1999 (LI 1652). The Environmental Protection Agency is the regulatory body that administers these laws. Under these laws, the holder of a mineral right is required, to get the necessary approval and permits from the Forestry Commission and the EPA before undertaking an activity under the license. The agency requires that prior to the exploitation of the minerals, the inventory of the flora and fauna of the mined area should be taken and preserved until used. The topsoil which is the repository of organic matter, seeds of native plants and some fauna should be stripped and stockpiled in a protected environment. Guidelines on reclamation of degraded mine lands recommend that soil materials should be replaced in the

same sequence as they occur in nature. This is significant in ensuring the protection of the soil, terrestrial ecosystem and public health.

2.3 Surface mining

According to Sam (2013), surface mining includes: strip and open pit mining, placer and hydraulic mining in riverbeds, dredge, beaches and terraces. The strip-mining technique is used where the mineral body is relatively close to the surface and the deposit runs in a roughly horizontal direction (Sam, 2013) like coal, bauxite and to some extent, sand. The open-pit mining on the other hand, occurs at hard mineral mines, such as gold, copper or zinc. As put by Hartma (1987), globally, about 75 to 85% of mineral extraction is carried out following the open pit methods.

In Ghana, all large-scale gold mining companies, operate hard rock open pit mining. This type of mining is generally used when the deposit runs in a roughly vertical direction. The mine progresses downward through a series of benches which slope toward the bottom of the pit. Where non-mineral bearing rock herein referred to as waste rock, overlays the mineral, it must be excavated to provide access to the mineral. This waste material is placed in a separate waste rock dump which may be permanent or temporary. According to Akabzaa and Darimani (2001), open-pit mining has the most visible and dramatic destruction to the surrounding landscape, where large areas of land are stripped of their vegetation cover, leading to the disturbance and loss of forests and animal habitat while leaving huge open pits behind.

2.4 Mine wastelands and materials

Precious minerals such as gold are concealed below layer of unconsolidated materials or rocks that must be excavated in order to get to the ore deposits (Rankin 2011; Vela-Almeida et al. 2015). Surface mining of these minerals generally involves the removal of vegetation, topsoil, subsoil, overburden and then excavation of the minerals (Cooke and Johnson 2002; Gathuru 2011; Mensah

2015). The process generally involves removal of vegetation, topsoil, subsoil, overburden and then excavation of the minerals. The activities leading to the extraction of the ore deposits and other subsequent mining operations, generate enormous amounts of mine wastes including waste rocks, tailings, spoil and heap leach on land surfaces. Mining activities results in such wastelands as stripped areas, open-pits, over-burden stockpiles, waste rock dumps, tailings storage facilities and other degraded lands (Wong 2003; Li 2006; Sikaundi 2013; Venkateswarlu et al. 2016).

These wastes are matters of environmental concerns and must be managed or handled in a way that avoids negative environmental impact. According to Lèbre and Corder, (2015), cited by UNDP (2018), it is estimated that metal mining generates about 15 billion tons of waste per year, which is 10 times larger than global municipal waste. However, the amount of waste produced by mining activity is influenced by the type of mining technique used, mineral extracted and the ore grade (UNDP, 2018).

2.4.1 Tailings

After the removal of waste rocks, the ore is then excavated, crushed and refined using various enrichment methods that extract the desired mineral. Subsequent to the extraction and beneficiation process of the mineral, a high-volume of tailings is generated (Lottermoser, 2010) in fine-grained particles. The IFC/World Bank (2007), officially defines tailings as the residue of metallic ore which remains after it has been ground into fine particle sizes and the desired metals have been extracted with cyanide in case of gold or sulfuric acid for copper. This mill waste is transported in a form of slurry and disposed of into a wet impoundment facility called tailings storage facility (TSF) or tailings management facility (TMF). The impoundment of the tailings in the TSF or TMF is to prevent spillage and release of toxic constituents of the tailings into the environment.

The physicochemical characteristics of the tailings vary greatly from site to another and is dependent on the ore mineralogy and the method of extraction. Ritcey (1989) reported that tailings of the same type may have the same mineralogy and characteristics. A decommissioned mine tailings have characteristically low organic matter content (Cooke and Johnson 2002; Titshall et al. 2013), low pH (Chaturvedi et al. 2012) in the case of pyritic ores, high pH (McClure et al. 2009; Yassir et al. 2015; Guo et al. 2022) in carbonate bearing ores, poor water holding capacity, low microbial activity and high levels of heavy metals (Krzaklewski and Pietrzykowski 2002; IFC/World Bank 2007; O'Dell et al. 2007). Furthermore, the tailings have high bulk densities (Titshall et al. 2013), low infiltration rates (Wong 2003; Mensah 2015), deep and relatively uniform profiles. These mine wastes are characterized by their instability and lack of cohesion (Asensio et al. 2013) and typically range from sand to silt-clay in particle size.

2.5 Impact of mining on the ecosystem

The various phases in the life of a surface mine viz mineral exploration, mine development, mining operations and mine closure, pose numerous problems to the landscape on which they occur (Farell and Kratzing, 1996; Schueler et al. 2011). Kuma (2007) reported that public discussions and debates which have taken place over the years on mining, relate to the undesirable impact of mining on the ecosystem. The most severe impact of surface mining on the ecosystem relate to change in land form due to removal of vegetation, topsoil and deposition of large quantities of mine waste materials on the surface (Buchanan and Brenkley, 1994). These disruptions to the ecosystem cause significant damage to the soil, the native fauna, flora composition (Mummey et al. 2002) as well as surface and subsurface water (Chamber of Mines of South Africa, 2008). Surface mining can also have adverse effects on human health and the environment through soil, air, noise and water pollution (Sengupta, 1993; Younger, 2004; Bell and Donnelly, 2006).

Surface mining activities result in complete removal of vegetative cover of lands (Amegbey, 2001), depriving the site of plant nutrients essential for supporting a healthy ecosystem (Kundu and Ghose, 1997). Mining is also associated with loss of biodiversity and natural resources (Brooks, 1997; Yelpaala, 2004) and changes in hydrology and landscape stability (Danielson and Lagos, 2001). It also leads to alteration of land morphology, and the quality of air (Kavourides et al. 2002) and disruption of the aesthetic value of the landscape. The USEPA (1989) warns that the improper management of wastes materials produced from mining is detrimental to environment and public health. Sam (2013) also reveals that mining companies have been criticized severally over destruction of farmlands, polluted environment and regular drying up of water bodies.

The waste rocks on the mine may contain sulphide or potential acid forming materials and if not properly disposed of tend to produce AMD which dramatically reduces the pH of surface and ground water while damaging the downstream aquatic ecosystem. Along with acid mine drainage, the disposal of mine waste can also cause severe water pollution from toxic metals. The toxic metals such as As, Cu, Pb and Hg, commonly found in mine waste are harmful to the health of humans and wildlife if they are released into nearby streams.

The Commission on Sustainable Development at the United Nations has observed that the use of mercury, cyanide or the mixture of both for mineral extraction constitute a serious environmental threat (Spiegel and Hoeung, 2011). The Commission also credited mining as the single major source of mercury in the world. Periodic spillages of these chemicals from mining operations into the soil and downstream water bodies resulting in soil and water pollution and extermination of aquatic life are common features of mining communities. This also has detrimental effects on the health of the people living at or around the mining site. Research conducted by UNIDO (2001) showed higher concentration of mercury in all the environmental media in the village of Dumasi

in the Prestea Huni-Valley District in the Western Region of Ghana. The research found that fish caught in local streams contained mercury higher than the U.S-FDA threshold implying that they are not wholesome for consumption. They later revealed that inhabitants were over-exposed to mercury. In 2008, GNA reported an incidence of cyanide spillage into down streams in some communities within the concession of Newmont Ghana causing extermination of some aquatic life and other health related issues on people in those communities.

2.6 Reclamation of degraded mine land

To put it simply, to reclaim is to restore to a right condition. Reclamation within the context of mining, has been defined by Munshower (2000) as the process of returning a degraded mined site to conditions comparable to its natural state. Naeth et al. (2012) consider reclamation as the process of repairing land disturbed due to anthropogenic activities, back to its pre-disturbance state or a state of equivalent functionality. According to Bradshaw (1996), reclamation is a procedure of making a disturbed area suitable for cultivation which is aimed at enhancing fertility status of the soil to sustain plant life.

The Land Conservation and Reclamation Council (1992), also define reclamation as the procedure of reconverting damaged land to its original state or other profitable uses. The process of reclaiming a degraded mine sites involves physical stabilization of the terrain, landscaping, restoring topsoil and the revegetation. It also requires adequate quantities of soil of suitable quality for shaping relief and stabilizing the site (UNEP, 1983). Polster (2009) reported that dramatically disturbed soils are very difficult to reclaim due to the absence of rich topsoil and a substrate for plants growth. According to UNPD (2015), reclamation is best accomplished through the availability of fertile topsoil, soil amendments, and nitrogen fixing plants.

The overall goal of reclamation is to stabilize the soil, vegetation and water conditions of the reclaimed site, while returning affected areas as near as possible to their economic and ecological value (UNEP, 1983; Bradshaw, 1996; Yelpaala, 2004; Cao, 2007; Post-mining Alliance, 2015; Hayes, 2015). Meanwhile, it is not a process that should be considered only at, or just before, mine closure. Rather, it should be planned and considered across all stages of mine development and operation, from design to closure (Australian Government, 2006). Walde (1993), considers the reclamation program as an on-going activity. In most cases, successful reclamation can result in the timely establishment of a functional ecosystem. However, Bradshaw (1996) noted that rehabilitation of degraded mined site takes time and effort to accomplish any critical impact.

2.7 Reclamation process

According to ICMM (2008), the decision to commence reclamation is backed by planned programs consistent with the overall stated closure objectives, cost allocations, available equipment and technologies. Reclamation commences during the design stage and continues throughout the life of the mine. According to Strzałkowski and Ka´zmierczak (2019) reclamation works can be divided into three sequential phases: preliminary, technical, and revegetation. Additionally, the ICMM (2008) also included monitoring and measurement of success criteria as final and mandatory activity for relinquishment.

2.7.1 The preliminary phase

The preliminary phase aims at cleaning the terrain by removing extra vegetation, demolishing the remaining parts of buildings, clearing the rubble and debris from the demolished buildings. This phase also deals with the project documentation, project costing, project timing and duration, mobilization of materials and equipment.

2.7.2 Technical phase

The technical phase consists of earthworks, slope battering and restoration of the layer of fertile soil. This phase prepares the area for the introduction of vegetation.

2.7.2.1 Earthworks

The first critical step in reclamation process is the reconstruction of the degraded landscape, which is mainly guided by the natural landscape and the objective for post-mining land use. The process of earthworks involves grading, levelling, re-shaping, and recontouring to provide a minimum 3.0H:1.0V slope to blend with the nearby undisturbed land (Australian Government, 2006). Accordingly, earthworks should aim at recreating similar topographic and vegetation patterns to mimic natural landforms as much as possible (Australian Government, 2006) while synchronizing the hydrology of the disturbed land with the surrounding environment (Devito et al. 2012). The natural topographical slopes differ from constructed landscape slopes in many ways. Whereas the constructed landscape slopes are redesigned to a linear profile, the natural landscape are generally concave-shaped and tend to capture erosional sediment on the slope.

2.7.2.2 Spreading of oxide material on waste rocks

The essence of adding laterite or oxide material prior to addition of topsoil is fundamentally to bind the topsoil so as to enhance the stability of the reconstructed land surface (Thagesen, 1996). When reclaiming acid mine drainage (AMD) producing waste rock dump (WRD), the oxide material or laterite is used for encapsulation with the help of impact roller machine. The laterite is compacted to form a seal over the rock surface to prevent influx of air and water being the main ingredients of AMD. The laterite or oxides are soils formed under high temperature and heavy rainfall conditions which leads to leaching of soil, leaving great amounts of iron, aluminum and manganese oxides (Sposito, 1989; Thagesen, 1996). The presence of aluminum and iron oxides

stabilize clay minerals by decreasing critical coagulation concentration, clay dispersion, water uptake, and clay swelling and by increasing micro-aggregation (Thagesen, 1996; Schulze, 2005). Clay is an important component of lateritic soils and with its mineralogical composition of kaolinite, illite and montmorillonite, they are applicable to engineering practice. The fine content has a bearing on the engineering properties of soils used as sub-base material (Schulze, 2005).

2.7.2.3 Spreading of topsoil on constructed landscape

The topsoil is spread to provide a growth medium for plants during the post-mining land use. It is the most important element of a successful reclamation program which must be managed and used with great care. Subsequent to the application of laterite or oxide material, topsoil is spread at a depth of 150 mm, allowing a settlement differential of about 50-75 mm (Sangakkara et al. 2008). However, the volume of topsoil to be applied is influenced by the type of vegetation to be established, the quantity and quality of the topsoil and topsoil availability. The Chamber of Mines of South Africa/Coaltech November (2007), recommends that, soils should be replaced in original sequence in the catena (re-created slope) to their natural location. However, the Australian Government (2006) reveals that this is a theoretical ideal and no mining equipment would be able to do this without dramatically compacting the underlying soils as the next layer is placed. They further concluded that there are limited known cases of large-scale rehabilitation where a deliberate effort has been made to replace soil materials in the same sequence as they occur in nature. Nevertheless, this sequential soil replacement results in significant benefits in the re-establishment of the natural soil fertility recycling processes.

Aside from the nature of mining equipment used, the amount of water in the soil to a greater extent, influence the degree of soil compaction and structural breakdown during the spreading. Down and Stock (1973) advise that topsoil is best removed and spread when it is friable and at a time when

there are no rains to induce the effects of the equipment used. However, within the limit of these constraints, the combined use of a front-end loader, truck and bulldozer for the removal, transport and spreading of topsoil is the best combination to reduce compaction. According to Law (1984), the topsoil serves as the basis of the new ecosystem therefore its management is crucial for a successful reclamation of degraded mined soils.

2.7.3 Establishment of vegetation and field maintenance

Subsequent to the spreading of topsoil and allowing it to settle, new vegetation is re-established (Atyeo and Thackway, 2009) for specific purposes such as control of erosion, moisture content, or aesthetic values (MVLWB/AANDC, 2013). Grasses and cover crops; leguminous with nitrogen fixing ability such as *Centrosema pubescens, Puereria phaseoloides, Crotalaria juncea* are introduced first. The grasses and cover crops are established rapidly to control erosion (Polster 1991; Sangakkara et al. 2008), while the cover crops enhance the nitrogen content of the soil. The grasses are raised vegetatively (Polster 1991), while leguminous cover crops are established by broadcasting the seeds evenly over the entire reclamation site (Sangakkara et al. 2008). These are allowed a period of 3-6 months to establish before infill planting of seedlings or transplanting occurs (Withes, 1999).

According to EPA-Ghana, (1994), 40% native species and 60% exotic tree plants must be retained at a planting distance of 3 x 3 m to attain 1000 trees per ha. The native species, should include a combination of ground covers, shrubs and trees. Tree plants that are drought-resistant, fast-growing, have nitrogen fixing ability and can thrive well in degraded and nutrient-deficient soils are preferred. These plants, characteristically enhance organic matter buildup, nutrient cycling, promote microbial activities, while controlling erosion (Josa et al. 2012; Gao et al. 2017).

Literature replete that a disturbed area may be revegetated through natural recovery and or assisted revegetation. Bradshaw, (1996), reported that natural recovery requires enough time, but can be hastened if assisted. However, in most cases, assisted revegetation shows a higher biomass growth and better economic benefit as compared to the natural succession (Vacek et al. 2018). But Wampler et al. (1997), is of the view that combining the assisted and natural recovery is ideal. Furthermore, UNDP, (2018), specifies that species and varieties of plants should be selected to suit local soil conditions and the purposes of reclamation. The success of reclamation program is largely dependent on the fertility of the topsoil and management practices used during reclamation (Bradshaw, 1996). However, with time, natural succession is accelerated through the seeds stored in the topsoil.

When an area or the entire reclamation field experiences poor germination, revegetation is repeated to maintain continuous cover for the site. It also involves checking and controlling soil erosion, disease and pest infestation, weeds and invasive plant species.

2.7.4 Monitoring and measurement of success criteria

In mine site reclamation, success indicators are defined as the standards of performance that assist the regulatory agencies, communities and mining companies to evaluate the trajectory of the reclamation program (Elliot et al. 1996) needed for the closure of the site and the relinquishment (WA EPA 2006). EcOz Environmental Services, (2013), describes this phase as an important point in the reclamation processes that determines whether the rehabilitation program is achieving its goals and mine closure criteria. Asher and Bell (1999), reveal that routine monitoring of the flora and fauna communities on both reclaimed and immediate undisturbed areas, enables land practitioners to appreciate changes in biodiversity, which are underpinning indicators of successful ecosystem recovery. As noted by the EPA-Ghana (1996), frequent monitoring is required for

corrective mitigation measures. Mine site reclamation is considered successful and agreed upon for relinquishment when the site can be used for its intended purpose without posing any management threat (Laurence, 1999). According to Gravina et al. (2011), an ecosystem is deemed restored when it achieves at least 65 to 75% of reference site values when evaluated.

2.8 Measurement of biodiversity

Ecologists employ several methods to measure biological diversity of an ecosystem. These methods include canopy fogging, quadrat sampling, transect sampling, and netting (Baumgartner, 2003). However, the method used depends on the type of organisms sampling and the habitat. In fact, species diversity of ecosystem is often measured using the number of species present (species richness), relative abundance of individuals of each species (species evenness) and dissimilarities between species in a given habitat (Magurran, 1988; Baumgartner, 2003). Johnson and Tanner (2004) contend that biodiversity in reclaimed sites differ considerably from the original ecosystems. Largely, an ecosystem with many different types of species or more species is considered more diverse. As claimed by Groeneveld (2004), an ecosystem dominated by a few species is considered less diverse than an ecosystem where the abundance of all species is distributed evenly. Greater diversity of species indicates greater ecological quality (Singh et al. 2005, Singh and Nautiyal, 1990).

2.9 Biodiversity indices

The Simpson's index and Shannon-wiener index are the most frequently indices used by many Ecologists to assess the biodiversity of an ecosystem. However, the latter is a widely used biodiversity index and will be used for this study.

2.9.1 Shannon-Wiener index

The Shannon-Wiener index (*H'*) is widely used to determine biological diversity of an ecosystem. This index was derived by Shannon and Wiener from the application of information theory (Shannon and Weaver, 1949). The value of *H'* integrates both species richness and the species abundance (Gaston and Spicer, 1998; Tilman and Lehman, 2001). Pielou (1966) noted that, the Shannon-Wiener index assumes that: all species are represented in the sample, and that individuals are randomly sampled from large population. With Shannon-Wiener, even the rare species with one individual, are also included in the index. Therefore, if a given habitat has many rare species, their contributions would count. Shannon diversity index has a minus sign in the calculation, however, the index actually becomes positive.

The Shannon-Wiener diversity index **(***H'*) is often calculated as follows:

$$\mathbf{H' = -\Sigma\left[\left(\frac{n_1}{N}\right) \times \ln\left(\frac{n_1}{N}\right)\right]} \quad \textbf{..... Eqn. (2.1)}$$

Where ***H'*** is the Species diversity index, ***ni*** is the number of individuals or amount of each species (i'th species), ***N*** is total number of individual species (or amount) on the site, ***ln*** denotes natural logarithm of the number.

2.9.2 Species richness indices

Magurran (1988) defines species richness as the number of species that occur in a given habitat. It has been described as a single important metric that is valuable as the basic currency of biodiversity (Davis, 2003). However, Donovan and Welden (2001) maintains that richness alone cannot be used as a sole indicator of desired diversity but must be integrated with other metrics to fully capture biodiversity. Species richness is measured in the Margalef's index (Margalef, 1958).

$$\textbf{Margalef's index} = \frac{(S-1)}{\ln N} \quad \textbf{Eqn. (2.2)}$$

Where ***S*** is total number of species recorded, ***N*** is the total number of individuals in the sample and ***ln*** denotes natural logarithm.

2.9.3 Species evenness

This is the proportions of species present in a particular site. A site with high evenness indicates that more species dominate the site and the vice versa. Species evenness is calculated using Pielou's evenness (equitability) index (Pielou, 1966). The Pielou index J' is defined as:

$$\mathbf{J' = \frac{H'}{\ln(S)}} \quad \textbf{..... Eqn. (2.3)}$$

Where ***J'*** is Pielou's evenness index; ***S*** is the total species number in the sample; ***H'*** is Shannon-Wiener diversity index and ln denotes natural logarithm.

2.9.4 Similarity coefficient

It is a measure of similarity or degree of association in species composition between sites or habitats. Similarity of two habitats is subject to the presence or absence of species in the two habitats (Legendre and Legendre, 1984). The degree of similarity ranges from 0 to 1. Coefficients close to 1 show a more similarity among two populations while values close to 0 express low similarity among two populations. Similarity is calculated using the Czekanowski index below:

$$\mathbf{S_C = \frac{2\sum_{i=1}^{n} \min(X_i Y_i)}{\sum_{i=1}^{n} X_i + \sum_{i=1}^{n} Y_i}} \quad \textbf{..... Eqn. (2.4)}$$

Where S_C is similarity coefficient; X_i and Y_i are abundance of species *i;* $\sum_{i=1}^{n} \min(X_i Y_i)$ denotes sum of the lesser scores of species *i* where it occurs in both quadrats and ***n*** representing number of species.

2.9.5 Species density

Species density or population per m^2 is typically used to evaluate species richness. However, species richness increases with sample size. The smallest sample size may be 1 m^2 and the largest

may be the entire region or country. Species density is estimated as:

$$\textbf{Density} = \frac{\textbf{Number of individual species}}{\textbf{Area sampled}} \text{..... Eqn. (2.5)}$$

2.10 Soil quality and assessment

The term "soil quality" refers to the capacity of a particular type of soil to function within unmanaged or managed ecosystem boundaries to support plant productivity while preserving or improving water quality, promoting human health, and allowing for human habitation while minimizing soil degradation (Doran et al. 1994; Karlen et al. 1997; Karlen et al. 2003). The SSSA (1997) defines soil quality as "the ability of a certain kind of soil to function within ecosystem limits to sustain biological productivity, maintain environmental quality, and promote plant, animal, and human health". The above definitions suggests that soil quality is the special ability of a soil to perform its functions in the environment.

Many authors have interchangeably used the terms "soil quality" and "land quality" (Eswaran et al. 1997). In clear terms, soil quality is merely one aspect of land quality, which can be assessed by the cumulative effects of various soil attributes (Carter et al. 1997; Dumanski and Pieri, 2000). While land quality focuses primarily on the inherent soil properties that do not change readily and are frequently assessed beyond the plough layer, soil quality focuses primarily on the dynamic soil properties that are easily affected by management practices at the plough layer of the soil (Karlen et al. 2003). Hence, land quality has a more permanent character than soil quality (Bouma, 2002). Soil quality is a complex function that cannot be directly measured in the field or laboratory, but must instead be inferred from soil characteristics (Diack and Scott, 2001; Stockings, 2003).

Soil quality measurement is an assessment of soil properties that are sensitive to management effect and can be precisely measured within certain technical and economic constraints (Doran and

Parkin, 1996). However, soil quality is frequently influenced by a variety of factors such as soil type, management practices, environmental influences, and dynamic soil characteristics (Stott et al. 2013; Mukherjee and Lal 2014; Hammac et al. 2016). According to Zhao et al. (2013), soil quality development is critical in the soil reclamation process.

2.11 Indicator parameters for mine soil quality

The identification of soil quality indicators is critical for assessing the progress and success of degraded land reclamation. Indicators of soil quality are soil properties that are sensitive to changes in soil conditions (Doran and Jones, 1996; Herrick et al. 2002; Aparicio and Costa, 2007). An ideal soil quality indicator should be easy to measure, sustainable and possess specific characteristics that reflect the structure and function of ecological processes (Schloter et al. 2003) that could be used to quantify the quality of reclaimed mined soils. Soil quality indicators are commonly used to assess the anthropogenic and natural impacts on soils, and evaluate the effectiveness of sustainable land management practices (Karlen and Stott, 1994; Doran and Parkin, 1996) and to understand the progress of mine soil development. Soil reaction, bulk density, aggregate stability, SOM, and microbial activity are among the most commonly used soil properties to evaluate soil quality (Bastida et al. 2008). Arshad and Martin (2002) have reported on other indicators such as electrical conductivity, soil respiration, CEC, and heavy metals. A review on soil quality indicators by Bünemann et al. (2016) revealed the most frequently used biogeochemical indicator parameters for soil quality assessment (Figure 2.1). When evaluating the recovery of ecosystem and mining soil properties, integrated indices are regarded to be particularly well suited. However, soil indicators may also be utilized singly (Gil-Sotres et al. 2005).

2.11.1 Chemical parameters

2.11.1.1 Soil reaction

The pH of a soil is a numerical representation of its relative acidity or alkalinity. Soil pH is a crucial soil chemical characteristic that affects plant nutrient availability and crop development (Fageria and Baligar, 1998; Ryan et al. 2001; Rahman and Ranamukhaarachchi, 2003). Mengel and Kirkby, (1982) observed that soil pH determines the existence of soil-microorganisms and their interactions with the microenvironment. They noted that pH is a critical variable in organic matter decay and plant nutrient availability. Donelly et al. (1990) asserted that microbial biomass and lignin decomposition are significantly affected when the pH is less than 4.5. But Santa (2000) also found that microbial activity and nutrient turnover rate are hastened if the soil pH is greater than 4.5. In very acidic soil, calcium, phosphorus, and magnesium become less accessible to plants, whereas aluminum and manganese become more available and potentially more hazardous (Rowell, 1994). Soils that have pH at near neutral generally have the most readily available plant nutrients. Sheoran et al. (2010) attributed low soil pH to leaching due to excessive rainfall, plant uptake of basic cations, organic matter decomposition, plant root exudates and presence of acid forming substance in the parent material.

2.11.1.2 Soil organic matter

Soil organic matter is the part of soil that consists of remnants of plants and animals in different states of decomposition. It is by far the most influential, widely and reliable indicator that may be used alone to assess soil quality (Rajan et al. 2010; Bünemann et al. 2016). Bodláket al. (2012) noted that it is the most important soil property for assessing post-reclamation performance. Described as a crucial indicator parameter of soil quality (Doran and Parkin, 1996) soil organic

matter influences many fundamental biogeochemical processes in the soils (Mukhopadhyay et al. 2013) more than any other soil property.

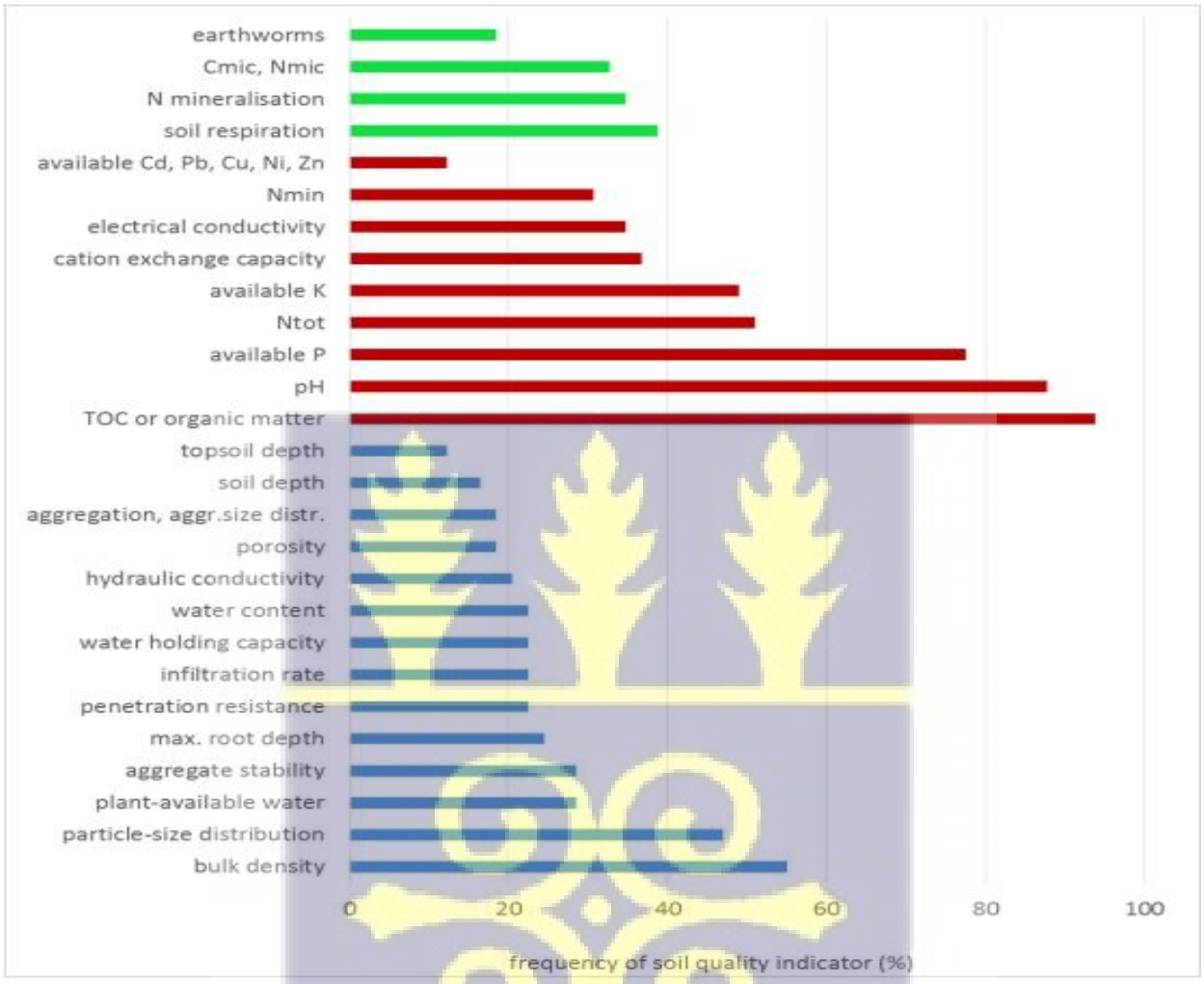


Figure 2.1. Frequency of different indicator parameters used for soil quality assessment (n = 49). Soil biological, chemical and physical indicators shown in green, red and blue, respectively. Source: Adapted from Concepts and indicators of soil quality – a review by Bünemann et al. (2016).

SOM provides many essential plant nutrients (Donahue et al. 1990) and has a high capacity to adsorb and exchange cations (Ryan et al. 2001). It serves as source of energy to micro-organisms in the soil which thence hasten its decomposition. Organic matter decomposes and releases

nitrogen, phosphorous, and sulfur to the soil matrix. Soil organic matter enhances aggregate formation, healthy tilth, and water infiltration while increasing the soil's ability to retain water and nutrients (USDA-NRCS 2004; Plaster, 2009). Soil organic matter positively correlate with such soil properties like cation exchange capacity, total N, available P, aggregate stability, water retention capacity and microbial biomass carbon but negatively correlate with pH and bulk density (Barber, 1995; Reynolds et al. 2009; Agus et al. 2014).

2.11.2 Physical parameters

2.11.2.1 Coarse fragments

USDA-NRCS (2008) describe coarse fraction as that fraction of soil that is greater than 2 mm size. It is a crucial indication of the quality of mined soil. Numerous researches have observed a greater proportion of coarse fraction in reclaimed mine soils and its effect on plant development (Andrews 1998). Mukhopadhyay et al. (2016) observed that in reclaimed soils, a large number of gravels inhibits the root development and overall performance of tree species. In his work, Rodrigue and Burge, (2004) classified coarse fraction as a critical mine soil properties which affect soil productivity and plant performance. According to USDA-NRCS (2008), coarse soils have poor physical properties that negatively impact water retention, soil microbial activity and nutrient cycling.

2.11.2.2 Bulk density

Bulk density is the most influential physical property of soils. It is a direct indicator of soil compaction, aeration, and water retention capacity (Hernanz et al. 2000). The Western Australian Environmental Protection Authority (2006), describes bulk density as an excellent measure of soil degradation which results from management practices such as poorly-timed tillage, trafficking by heavy equipment and stocking. Tsimba et al. (1999) also found that bulk density rises with soil

depth and ascribed it to low content of organic matter, decreased aggregation, root proliferation, and compaction resulting from the weight of overlying layers. Alexander (1980), also noted that soils with a high organic matter content tend to have a lower bulk density, most likely as a consequence of an increase in biological activity that leads to the formation of more soil pores. Bulk density <1.5 Mg m^{-3} is defined as desirable for optimum movement of air and water through the soil (Hunt and Gilkes, 1992). On the other hand, McKenzie et al. (2004) ranked bulk densities >1.6 Mg m^{-3} as critical value and tend to restrict root growth. The USDA-NRCS (2008) also indicated that bulk densities ranging from 1.6 to 1.7 Mg m^{-3}, will restrict water movement down the profile.

According to USDA-NRCS (2008) soils rich in organic matter, loose and porous generally have lower bulk density. Additionally, sandy soils have relatively higher bulk density than finer-textured soils, such as silt and clay loams which have good structure and greater volume of pore space compared to sandy soils. The works of Mukhopadhyay et al. (2013) also reported soil texture, poor soil aggregate, low organic matter input, and higher coarse fragments as crucial factors that contribute to higher bulk density. However, they concluded that the use of heavy earth moving machine during land preparation for reclamation constitute higher bulk density in reclaimed mined soils.

The usefulness of bulk density as soil quality indicator has been reported in several studies. Wander and Bollero (1999) suggested that bulk density should be included in the minimum data set of soil because it is a dynamic soil property. Bulk density, as reported by Shukla et al. (2004) was used as indicator parameter for assess reclaimed mine soils in southeastern Ohio.

The value of high and low bulk density can influence plant growth (Liu et al. 2014). As reported by Mukhopadhyay and Maiti (2011), higher bulk density would restrict root establishment, air movement and hamper water supply for the growing plants. On the other hand, soils with low bulk density and strength are open-textured and porous, susceptible to erosion, have poor water retention and oxidation of soil organic matter (Sparling et al. 2003).

2.11.2.3 Plant available water capacity

The concept of plant available water capacity (PAWC) of the soil was proposed by Veihmeyer and Hendrickson (1931, 1948). Plant available water capacity is the difference between field capacity (the maximum amount of water the soil can hold) and the wilting point (where the plant can no longer extract water from the soil) (Hunt and Gilkes, 1992).

Plant growth is largely governed by the available water capacity of the soil which is the difference between the water content at field capacity (about -0.3 bars water potential) and wilting (about -15 bars potential) (Grewal et al. 1990; Sharratt, 1990). This is often expressed as a volume fraction or percentage, or as a depth (in or cm) (USDA-NRCS, 2005). Generally, the AWC is used as an indicator to assess the capacity of soil to retain water and make it sufficiently available for plant. Soil moisture availability is significant factor for vegetation development (Huang et al. 2013) and survival of fauna population in an ecosystem.

The AWC is an indicator of duration and intensity of water deficit (Ritchie et al. 1972) which can be used to measure the water budget of soil. The amount of soil water available for plants to access is mainly governed by soil texture, soil structure and the depth of root zone (Ritchie et al. 1972; USDA-NRCS, 2005). According to USDA-NRCS (2005), soil organic matter and carbonate levels and stone content also affect moisture storage. Soil texture and organic matter are the key

components that determine soil water holding capacity (Wheeler and Ward, 1998; Morris, 2004; Asgarzadeh et al. 2010).

Poor structure, low organic matter and presence of stones all reduce the moisture storage capacity of a given texture class. Soils with smaller particle sizes, such as silt and clay have larger surface area can hold more water compared to sand which has large particle sizes which results in smaller surface area. As the level of organic matter increases in a soil, the water holding capacity also increases due to the affinity of organic matter for water.

A study by Morris (2004) and Asgarzadeh et al. (2010) revealed that a 1% increase in soil humus will cause a 4% increase in stored soil water. Similarly, Wheeler and Ward (1998) also reported that one part humus holds four parts of water. Therefore, humus correlates positively with water holding capacity of the soil. Water availability is an important soil quality indicator because plant growth and soil biological activity depend on water for hydration and delivery of nutrients in solution (USDA-NRCS, 2005).

2.11.2.4 Aggregate stability

Aggregate stability evaluates the resistance of the soil to disruptive factors such as tillage and rainfall. Kempler and Rosenau (1986) described erosion due to wind and water as key disruptive forces that affect aggregate stability. Celik (2005) on the other hand, emphasized that aggregate stability is degraded by physical disturbances. These disturbances of soil aggregates are the main causes of soil degradation (Groenevelt, 1991). But, Tisdall and Oades (1982) noted that soil organic matter and biological activities in the soil help to bind aggregates together and protect the aggregates from disturbance. Clay particles, silt, carbonates, gypsum and sesquioxides have been reported by Chepil (1955); Tisdall and Oades (1982) and Ciric et al. (2012) as cementing agents as they bind individual particles increasing soil aggregation. These large aggregates are more

sensitive to management effects, hence a better indicator of changes in soil quality. Soil aggregate stability is generally documented as an important indicator of soil structure and quality assessment (Karlen and Scott, 1994; Arshad et al. 1996). According to USDA-NRCS (1998), higher aggregate stability indicates better soil quality. The works of Amezketa (1999) indicated that high aggregate stability is important to a number of physical and biological properties, process and functions that occur in soils. Barthes and Roose (2002) found that high aggregate stability helps to reduce the amount of runoff and erosion. The stability of aggregates is important due to its effects on movement and storage of water, bulk density, aeration, erosion susceptibility (Cerda, 1996). Biological indicators, such as microbial biomass carbon and activity, nutrient mineralization and plant growth are also influenced by a decline in aggregate stability (Loveland and Webb, 2003; Zhang et al. 2014).

The mean weight diameter (MWD), geometric mean diameter (GMD), water-stable aggregation (WSA), normalized stability index (NSI), percentage of aggregates destruction (PAD) have been used as indices to evaluate soil aggregate stability (Nichols and Toro, 2011; Zhou et al. 2020). The most widely used indices is the MWD which measures the mean aggregate size of a soil to determine its susceptibility to wind and water erosion. Chenu et al. (2000), investigated the aggregate stability of two land use systems (soils) in France and found out that the soils from the forest had >2 mm size fraction dominating the distribution indicating greater aggregate stability. Soils from farmlands had lower aggregate stability with more size fractions < 2 mm. He attributed this trend to increasing trend of soil organic carbon in a varied amount of 52.6 g kg for the forest soil and 8.4 g kg for the soils from the farmlands. Sparling et al. (2003), also reported that a soil with mean weight diameter > 2 mm is ideal indicator for soil quality.

2.11.3 Biological parameters

2.11.3.1 Microbial biomass carbon (C_{mic})

The microbial biomass is said to be the most satisfactory estimates of the restoration of soil microbial populations (Izquierdo et al. 2005) and stability of a restored ecosystem (Mummey et al. 2002). It also provides a measure of organic matter dynamics and the quality of soil structure. Edgerton et al. (1995) found a positive correlation between soil aggregate stability and microbial biomass carbon. Carter et al. (1999) also reported that soil microbial biomass closely correlated with soil organic carbon. They later asserted that the ratio of soil microbial biomass carbon to soil organic carbon (C_{mic} : C_{org}) can be used as indicator of soil health. Additionally, Singh et al. (1989) noted that soil microbial biomass carbon acts as a source of plant nutrients in mine soils which increases progressively with age of reclamation. Soil microbial biomass, soil organic matter, and microbial activity has been reported as indicators of soil maturity.

2.11.3.2 Dehydrogenase activity

Dehydrogenase activity (DHA) have been used as a better and sensitive indicator of soil quality in the reclamation process (Caravaca et al. 2003). It responds rapidly to changes in soil management more than any other variables hence used as a measure of biological changes in the soil (Bandick and Dick, 1999). Sinha et al. (2009) described DHA as a direct measurement of microbial population and activity that is present and active in viable cells. Many authors have associated soil dehydrogenase activity with the success of reclamation of degraded mine land (Bandick and Dick, 1999) and restoration of soil system functionality (Harris et al. 1989).

2.11.3.3 Labile pool

Soil organic matter is made up of different pools with varied intrinsic properties and rate of decomposition (Lehmann and Kleber, 2015). The labile organic carbon is the most active fraction

of soil organic carbon with rapid turnover rates, and it changes substantially after disturbance and management (Harrison et al. 1993; Coleman and Crossley 1996). McGill et al. (1986) defined soil labile carbon as the fraction of soil organic carbon which turns over relatively rapidly within few days to years whereas the recalcitrant carbon has a turnover time of decades to centuries.

Generally, the labile organic carbon in soil is mainly influenced by the decomposition of plants and animals (Bolan et al. 2011) which serves as primary source of energy for soil organisms (Chantigny, 2003; Haynes, 2005) and is directly available for microbial metabolism (Liu et al. 2013; Li et al. 2018). According to Mukhopadhyay et al. (2016), accumulation of labile carbon during land reclamation results in increase in microbial biomass carbon and associated soil biological activity.

Labile carbon is involved in different soil functions such as soil aggregate stabilization, nutrient cycling and carbon sequestration (Tisdall and Oades, 1982; Deb et al. 2015). Additionally, the labile carbon has been used to assess the impact of agricultural management and land use change on soil quality (Mirsky et al. 2008; Ibrahim et al. 2013; Geraei et al. 2016; Awale et al. 2017) and has shown to be a more sensitive indicator of changes in soil quality and function than the total organic carbon (Quanying et al. 2014; Awale et al. 2017).

2.11.3.4 Earthworms

Earthworms are typical ecosystem engineers and are associated with an array of soil processes and functions linked to the productivity of an ecosystem (Lavelle et al. 1997; Blouin et al. 2013; Peigne et al. 2017). The higher the earthworm density and diversity, the better their living conditions and the more fertile the soil is likely to be (Peigne et al. 2017). Species richness and diversity of earthworms have always been observed to be higher in undisturbed lands than in disturbed lands

(Schmidt et al. 2003; Frazão et al. 2017). However, even under disturbed land conditions, species richness and diversity increase when soil environment becomes favorable (Frazão et al. 2017).

Earthworms constitute the largest part of macro-invertebrate biomass (over 50%) in most soil ecology (Edwards and Bohlen, 1996; Sims and Gerard, 1999; Sinha et al. 2009; Tondoh et al. 2007) and their role in plant litter decomposition determines the structure and function of an ecosystem (Fragoso and Lavelle, 1992). Earthworm populations differ significantly in terms of numbers, biomass in which they are present, and diversity across ecosystems in which they are found. They are generally sensitive to physico-chemical properties of the soils, land use, and management practices which in diverse ways influence their presence and population in an ecosystem (Singh et al. 2016; Peigne et al. 2017).

Pelosi et al. (2009), observed that earthworm distribution is often influenced by pH, moisture content, available organic carbon, ambient temperatures and soil type. Singh et al. (2016), noted that earthworms prefer soil with relatively high moisture, calcium, and pH levels. McCallum et al. (2016), also found that diversity and abundance of earthworms are very low in soils having pH near 4.5. Breure (2004) had previously noted that a low pH in the soil leads to a decrease in abundance of earthworms. These corroborate the early works of Brady (1996), who noted that few earthworm species are tolerant to low pH but most thrive well in soils not too acidic. Generally, most studies have reported that earthworms can tolerate a pH range of 5.0 to 8.0 and an abundance of earthworms increase as pH is shifted from acidic or basic to neutral. They prefer medium textured well-drained and well-aerated soils but moist habitats. High earthworm population is associated with increased exchangeable calcium, percent base saturation of the soil and forest floor turnover rate (Reich et al. 2005).

The presence of earthworms is a good indicator of biology in the soil and improved soil health (Blouin et al. 2013). They are the most frequently used indicator species because they are highly visible and easy to count and characterize, and abundant. They have large impact on soil structure and its properties as well as organic matter dynamics (Reich et al. 2005; Barrios 2007; Blouin et al. 2013; Singh et al. 2018). In the tropical climate, their role as ecosystem engineers contributes to litter decomposition (Singh et al. 2019), soil bioturbation, increased porosity, aeration, and drainage of soils (Prusienkiewicz and Bigos, 1978; Lee, 1985; Blouin et al. 2013). Earthworms accelerate nitrogen mineralization from organic matter (Lee, 1985; Scheu, 1987), but the extent depends on the species and their interaction with soil characteristics, organic matter location, and soil biota (Butenschoen et al. 2009). Depending on the season, species, and stage of life cycle, earthworms are present at different soil depths (Bouche´ and Gardner 1984). Bouche´ (1977) classified earthworms into three categories based on their ecological distribution in soil. The categories are:

Epigeic species (surface/leaf litter dwellers). They live in the litter, and decompose fresh organic matter close to the soil surface. They are small and mostly dark red. Their preferred habitat is mostly grassland.

Endogeic species (shallow burrowers/mineral soil-dwelling). They live in vertical burrows and connected with the soil surface. They decompose organic substances in the soil. They are pale, not pigmented.

Anecic species (dwelling in deep vertical soil burrows). They make horizontal or randomly oriented burrows in the mineral soil, pull plant parts from the soil surface into their vertical stable burrows where they partly decompose organic materials on which they feed (Lee 1959; Bouche´, 1977; Lee, 1985; Lee, and Forster 1991).

According to Bouche´ (1969), earthworm sampling methods are classified into two categories namely: (a) passive and (b) behavioral. By the passive method, earthworms are physically (hand) sorted from the soil, litter, and other habitats. Using the behavioral method, earthworms are forced to the surface by drenching the soil with an irritant chemical or by stimulation with other expellant such as electrical shock or a combination of both (Darwin, 1883). The chemical extraction method does not require physical disruption of the soil system as earthworms are collected as they emerge at the surface (Pelosi et al. 2009). The chemical extraction method produces results faster than the hand sorting techniques.

The commonly used chemical expellant include formaldehyde (Raw, 1959; Satchel, 1969; Callaham and Hendrix, 1997; Rombke et al. 2006), commercial hot mustard (Gronstol et al. 2000; Lawrence and Bowers, 2002), allyl isothiocyanate (AITC) (Zabroski, 2003), and most recently, onion (Steffen et al. 2013). Electric shock extraction or mechanical vibration methods (Gunn, 1992; Thielemann, 1986) and other chemical repellant such as potassium permanganate (Evans and Guild, 1947) and household detergents (East and Knight, 1998) have also been used.

2.12 Effects of vegetation and litter on earthworm community structure and dynamics

Available information in literature indicate that the earthworm community is usually assumed to be stable and driven by vegetation. Plant species, patterns of plant litter disposal and their chemistry, determine the structure of the earthworm community (Gill, 1969; Choudhury, 1988; Polunin, 1984). In addition to soil properties, litter quality and quantity (Warren and Zou 2002; Aubert et al. 2003; Negrete-Yankelevich et al. 2008) have strong influence on the abundance and composition of earthworm communities (Tsukamoto and Sabang, 2005; Singh et al. 2016). It is recorded that plant litter affects the earthworm's habitat (Gill, 1969; Stanton, 1979; Richards, 1987) by suppressing their recruitment through shade (Berendse, 1998), microclimate effects

(Troumbis et al. 2002), creation of a physical barrier and biochemical influences to soil (Facelli and Pickett, 1991). High earthworm density is found in grasslands than areas with high litter accumulation (Knapp and Seastedt, 1986; Stinner et al. 1984). Higher growth of earthworms is associated with lower C: N litter types, but their abundance and density dwindle in polyphenolics and worsen by litter lignin content (Shipitalo et al. 1988; Kasurinen et al. 2007).

Litter quality and palatability influence feeding behavior and litter decomposition rates (Brown et al. 2000; Suarez et al. 2006). This was corroborated by Tresch et al. (2019) who noted that richness of plant species indirectly affects litter decomposition as a result of the number of fauna species and the activity of soil microbes. In the works of Warren and Zou (2002) in Puerto Rico, they concluded that litter quality has more influence on soil macro-invertebrates than litter quantity.

Depending on the contents of structural and secondary compounds in the tissues, Homer et al. (1988) classified litter into short-lived and long-lived organs. They postulated that short-lived organs decompose faster, while litter from long-lived organs is more resistant to decomposition. While Berendse et al. (1987); Choudhury (1988) attributed the faster decomposition rate of short-lived organs to presence of less lignin content and secondary chemicals. Feeny (1970) asserted that, long-lived organs usually have more lignin, less protein and secondary compounds, and the litter produced takes longer time to decompose.

2.13 Soil quality index

Soil quality index (SQI) is a way to integrate multiple points of information into one tool that can be used for decision making (Karlen and Scott, 1994). Practically, the integrated quality index has been commonly used and considered as a good method for estimating suitable soil quality index (Doran and Parkin, 1994; Andrews et al. 2002; Li et al. 2013). Wienhold et al. (2004) reported that

measuring these factors together and producing an index will help improve the sustainability of the land. A soil quality index can be used as a potential tool to measure the environmental conservation (Masto et al. 2007) and to select appropriate tree species suitable for reclamation (Sinha et al. 2009; Mukhopadhyay et al. 2013). Soil quality measurements need to be easily performed, and made widely available to land managers (Shukla et al. 2006) to enable them evaluate the effectiveness of land restoration strategies (Mukhopadhyay et al. 2014; Costantini et al. 2016).

Unlike air quality, water quality and chemical pollutants (heavy metals), no well-established standards exist for soil quality assessment. This makes SQI values meaningless. However, its usefulness largely depends on comparing the obtained values with values from the soils of undisturbed ecosystems (Arshad and Martin, 2002), where optimum values can be obtained and soil functioning is at its maximum potential (Masto et al. 2008). Additionally, De la Rosa and Sobral (2008) also noted that comparing SQI of disturbed soils with natural and undisturbed soils, where there is ideal balance between their physical, chemical, and biological properties appears to be an appropriate procedure to assess soil quality.

2.14 Soil quality indexing methods

Three methods thus, simple additive SQI, weighted additive SQI and statistically modeled SQI based on principal component analysis (PCA) have been used to estimate the SQI. These methods are conceptually different from each other. The simple additive SQI and weighted additive SQI rely mainly on subjective expert opinion on unscreened total data set and literature review and have high level of bias. The principal component analysis (PCA) is a statistics-based model which has been widely used to select the most appropriate soil indicator(s) to estimate soil quality index (e.g., Andrews et al. 2002; Sharma et al. 2005; Mukhopadhyay et al. 2016). The PCA is a reduction

tool which is more objective and could avoid any disciplinary bias and data redundancy to create a minimum data set (MDS) to reduce the indicator load using mathematical formulae (Doran and Parkin, 1996; Andrews et al. 2002; Navas et al. 2011).

The primary function of PCA is to reduce the dimensionality of the total or untransformed data set involving a large number of interrelated variables, while retaining as much as possible most of the variations present in all the original data set (Dunteman, 1989; Jolliffe, 2005). Many studies on soil quality assessment have used PCA as a multivariate technique to establish appropriate MDS. Andrews et al. (2002) compared index methods composed of different indicator selection methods (expert opinion and PCA) with scoring functions (linear and non-linear) for vegetable production systems. Sharma et al. (2005) used PCA to establish MDS for assessing soil quality affected by different management treatments. Mukhopadhyay et al. (2016) also used PCA to evaluate Soil quality index of reclaimed coal mine spoil. Tesfahunegn (2014) reported that a PCA based SQI appears more sensitive to disturbances and management practices than the expert opinion. Tesfahunegn (2014) described statistically modeled SQI based on PCA as the best and easiest model due to its objectivity and relatively higher success to predict crop yield with MDS. It is regarded as a relatively less expensive procedure over time compared to simple additive SQI and weighted additive SQI.

2.15 Establishing the minimum dataset

Larson and Pierce, (1991) were first to suggest that a minimum data set (MDS) should be used when measuring the quality of soils. They also suggested that a standard set of methodologies should be instituted to determine the MDS. Andrews et al. (2004) advised that in choosing parameters for the minimum data set, the reason soil quality is being measured should be a guiding principle. However, Acton and Gregorich (1995) noted that parameters chosen for the minimum

dataset, should provide numerical data that show the ability of the soil to fulfil its expected functions. The use of minimum dataset is the most practical way to broadly assess trends in soil quality for sustainable land management practices (Doran and Parkin, 1996) and to establish a national soil monitoring network (Arshad and Martin, 2002). There is no cap on the number and which indicator parameters to be used for soil quality evaluation (Schipper and Sparling, 2000) since soil quality is a complex concept and is measured on site specific bases and according to the purpose of land use (De Lima et al. 2008).

Several attributes have been chosen as key indicators in many previous works. According to Andrews et al. (2002), some main indicator consists of SOC, EC, pH and available P as a MDS for various kind of soil system. Ranjbar et al. (2016) also used sand, relative field capacity, Zn, sodium adsorption ratio, Ca, $CaCO_3$, Fe and BD as the refined MDS to evaluate soil quality for saffron fields. On the other hand, Mukhopadhyay et al. (2016) used pH, Coarse fragment, SOC, DHA, EC, Ca, P, and S as the final indicators selected by PCA to evaluate soil quality index of reclaimed coal mine spoil. Noviyanto et al. (2017) used PCA to determine the MDS and they consisted of pH, base saturation, BD, EC, CEC, available P, total N and organic carbon. Mukhopadhyay and Maiti, (2018) used soil CO_2 flux, organic carbon, dehydrogenase, coarse fraction, moisture, base saturation.

CHAPTER THREE

MATERIALS AND METHODS

3.1 Site characteristics

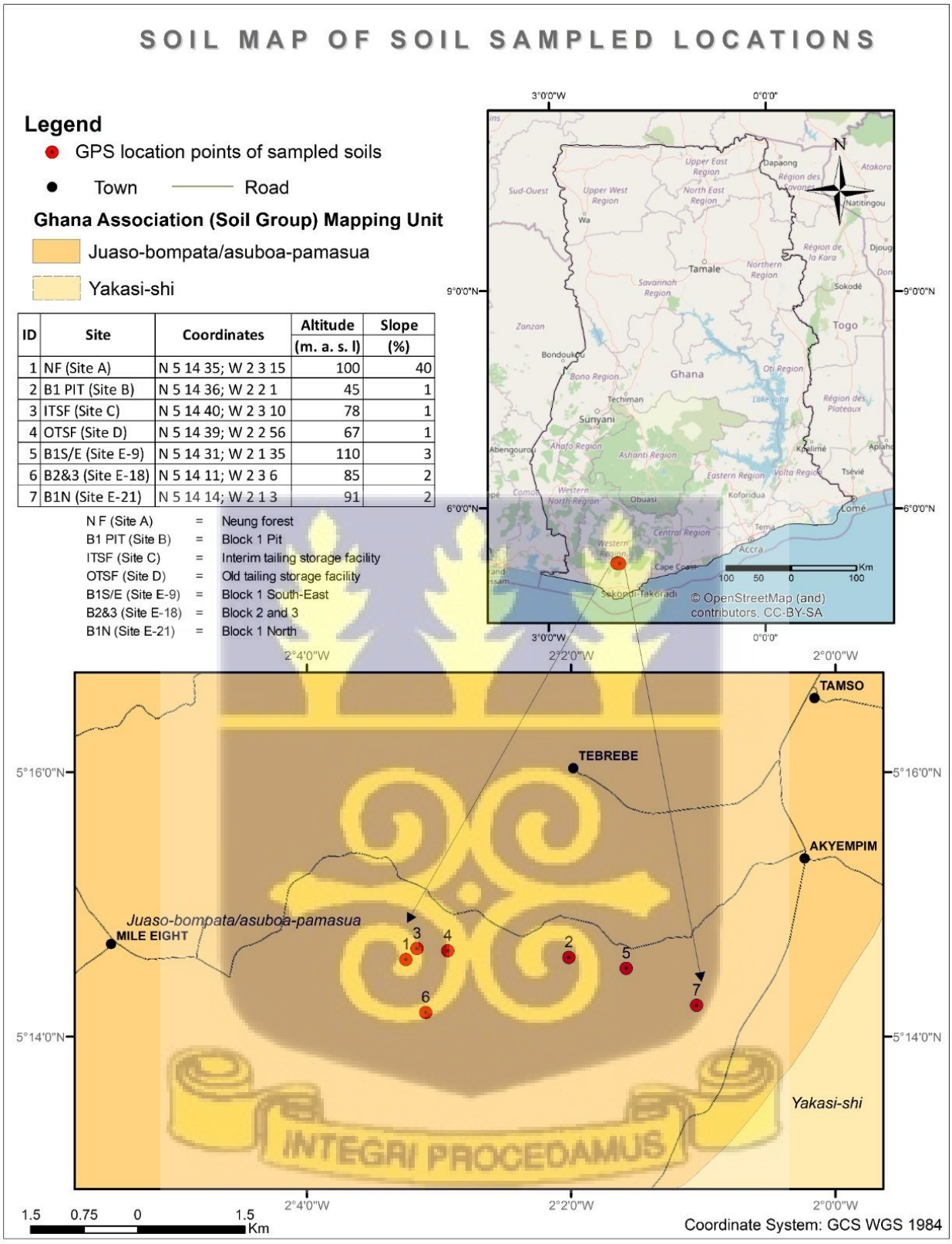
3.1.1 Location and geology

The study area is located at the AngloGold Ashanti Iduapriem concession area in the Tarkwa Nsuaem Municipality in the Evergreen Rainforest zone of Ghana (Figure. 3.1). Geologically, the Iduapriem mine is situated at the southernmost extremity of the Tarkwa basin. The Tarkwa orebodies are located within the Tarkwaian System, which constitutes a large component of the Ashanti Belt's strata in southwest Ghana (Kesse, 1985; Leube and Hirdes, 1986).

According to Kesse (1985), the Ashanti Belt lies in a broad syncline structure trending along the north-easterly direction. The belt comprises of sediments and igneous rocks from the Lower Proterozoic and is underlain by metamorphosed igneous rocks and sediments of the Birimian System (Leube and Hirdes, 1986). The Tarkwaian system unconformably overlies the Birimian system and is characterized by metamorphism of lesser intensity and the prevalence of coarse-grained, juvenile sedimentary strata. The topography of the area is comprised of a sequence of ridges and hills within the Tarkwaian geological formation, with moderately raised areas, lowlands, and valleys interspersed between them.

3.1.2 Climate

The climate of the study sites is transitional between the high rain forest (extremely humid) zone and the semi-deciduous rain forest (humid). The climate is tropical, with considerable rainfall in two main wet seasons and consistently high temperatures. From March to October, the climate is temperate and moist but gets hot and dry from November to February (GSS, 2014). The Tarkwa region is known for high rainfall. The average annual rainfall is between 1750 and 2000 mm. The



ID	Site	Coordinates	Altitude (m. a. s. l)	Slope (%)
1	NF (Site A)	N 5 14 35; W 2 3 15	100	40
2	B1 PIT (Site B)	N 5 14 36; W 2 2 1	45	1
3	ITSF (Site C)	N 5 14 40; W 2 3 10	78	1
4	OTSF (Site D)	N 5 14 39; W 2 2 56	67	1
5	B1S/E (Site E-9)	N 5 14 31; W 2 1 35	110	3
6	B2&3 (Site E-18)	N 5 14 11; W 2 3 6	85	2
7	B1N (Site E-21)	N 5 14 14; W 2 1 3	91	2

Figure 3.1. Map of the study site.

primary wet season spans the period March to July with a peak in June and a minor wet season from September to October. The main dry season lasts from November to February, whereas the minor dry season lasts only from August to September. June and January have the greatest and lowest average monthly rainfalls of 412 mm and 317 mm, respectively (GSS, 2014).

3.1.3 Soils and land use

The study area has two major land use systems namely, the undisturbed natural land with indigenous natural forest vegetation and the rehabilitated mined area with planted vegetation. The undisturbed soils (Mawso series or Haplic Acrisols) are located on the summit to upper slope of the landscape and are shallow, red and brown in colour, well to moderately-drained sandy clay and sandy clay loams containing ironstone concretions at varying depths below the surface. The rehabilitated land comprises three broad forms. The morphological characteristics of the undisturbed soil (unmined land) and rehabilitated soils (mined land) are described in section 3.2.

The natural vegetation is the evergreen forest type and falls within the rain forest belt of South-Western Ghana. The area is characterised by both primary and secondary forests and some swamps occurring in the valleys. The largest block of primary forest occurs within the Neung Forest Reserve, with the Northern Hills and the Iduapriem ridge retaining good vegetation cover. A large area of forest block also exists within the Ajopa hills. While the primary forest, is characterised by three layered emergent trees, the secondary forest is characterized by colonizing species at various stages of development. Some of the common trees found in the area are *Macaranga bartei, Baphia nitida, Alfzelia africana, Hildegardia barteri* and *Lophia alata.*

3.2 Site selection

The natural forest site and six mined sites under different modes and varied ages of reclamation were selected based on their availability, age and mode of reclamation. The reclaimed sites were categorized into four, which are: (i) an 8-year-old rehabilitated site which followed the conventional protocol on reclamation of mined soils (EPA, 1994); (ii) a 7-year-old soil reclaimed with tailing storage mixed with waste rocks (See Appendix 1) and other overburden materials; (iii) a 20-year-old soil reclaimed with tailing storage; and (iv) three soils of varying ages (9-, 18- and 21-year-old) with the same mode of rehabilitation (profiles packed with stored subsoil materials only).

For clarity, the following designations will be used for the soils studied:

Pedon A – the natural forest site;

Pedon B – an 8-year-old rehabilitated soil;

Pedon C - 7-year-old soil reclaimed with tailing storage mixed with waste rocks and other overburden materials;

Pedon D - 20-year-old soil reclaimed with tailing storage

Pedon E – three soils of varying ages (9-, 18- and 21-year-old) with the same mode of rehabilitation (profiles packed with stored subsoil materials only).

Characteristics of the soils at the five sites are described in the following subsections.

3.2.1 Pedon A - The Neung forest site

The soil from the undisturbed forest site was used as the control. The dominant vegetation in the Nueng forest consisted of *Macaranga bartei, Baphia nitida, Alfzelia africana, Hildegardia barteri* and *Lophia alata.* A cross section of the profile revealed four horizons (A, BA, Bt and Ct) with a very thick floor litter.

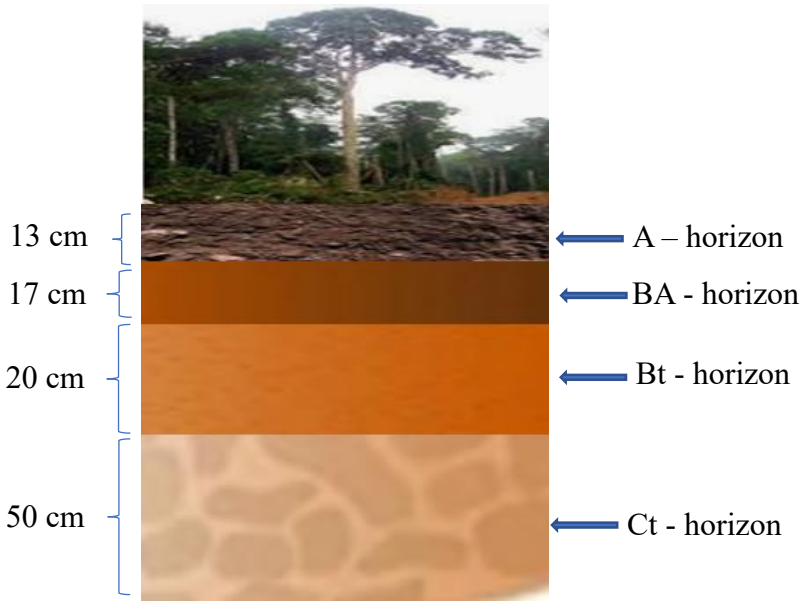


Figure 3.2. Schematic profile of Pedon A (natural forest) soil.

3.2.2 Pedon B - 8-year rehabilitated site.

The Pedon B soil is an 8-year-old rehabilitated profile formed on a large flat surface over an altitude of 45 m above sea level. The area is a decommissioned gold mine pit, backfilled with the excavated materials of weathered rocks and laterite, sub-soil, and topsoil. The planted vegetation was dominated by *Funtumia elastica, Gongronema latifolium, Chromolaena odorata, Vernonia conferta* and *Acacia mangium*. A cross section of the 100 cm profile showed a three-packing arrangement in order of topsoil, subsoil and a mixture of weathered rocks and other overburden materials from top to bottom. Cleared vegetation was incorporated in the soil materials during the reclamation process.

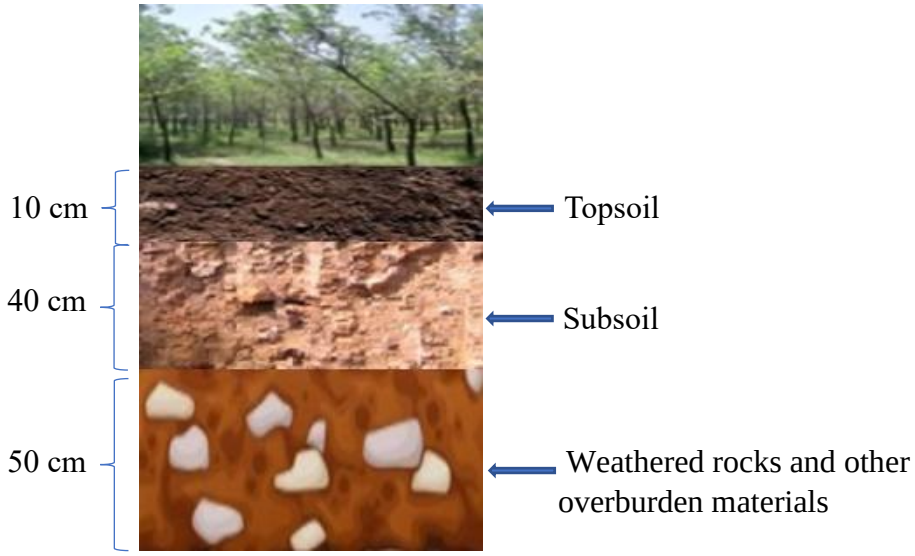


Figure 3.3. Schematic profile of Pedon B soil.

3.2.3 Pedon C - 7-year rehabilitated site

This is a 7-year-old reclaimed soil from a mixture of tailing storage, waste rock and other overburden materials, which occurred on a large flat surface at an altitude of 78 m above sea level. The planted vegetation was made up of grasses and herbaceous plants, *Cynodon dactylon*, and *Athyrium filx-femina*. Within the 100 cm depth, the soil material was uniformly packed with a mixture of tailings, waste rocks (See Appendix 1) and other overburden materials.

Figure 3.4. Schematic profile of Pedon C soil.

3.2.4 Pedon D -20-year rehabilitated site

The Pedon D soil is a 20-year-old tailing storage facility reclaimed over a very large flat surface on an elevation of 67 m above sea level. The dominant planted species were *Xylopia staudii, Funtumia elastica, Gongronema latifolium, Athyrium filx-femina,* and *Acacia kamerunensis.* A 100 cm vertical cross section of the soil profile, showed a uniformly packed with tailings material.

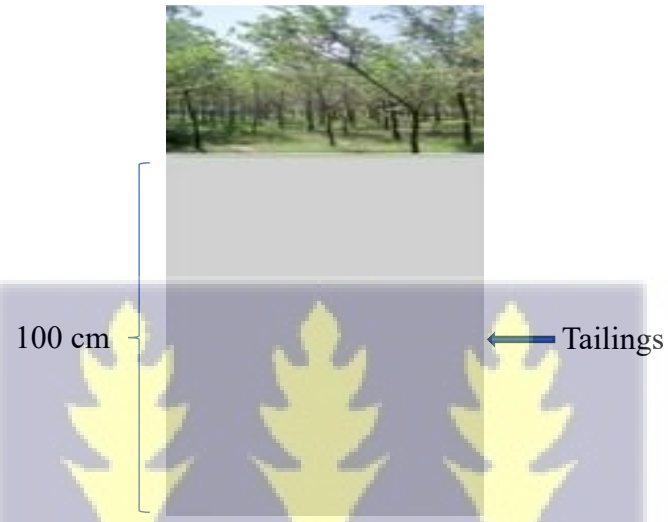


Figure 3.5. Schematic profile of Pedon D soil.

3.2.5 Pedon E - The 9-, 18- and 21- year rehabilitated sites

3.2.5.1 Nine-year-old reclaimed site (E-9)

The E-9 site is a 9-year-old reclaimed soil which occurs on a gently undulating landform at an elevation 110 m above sea level. The area is predominantly occupied by *Athyrium filx-femina, Funtumia elastica,* and *Gongronema latifolium* which were established for rehabilitation. The profile showed a uniformly packed 80 cm of subsoil material below the surface.

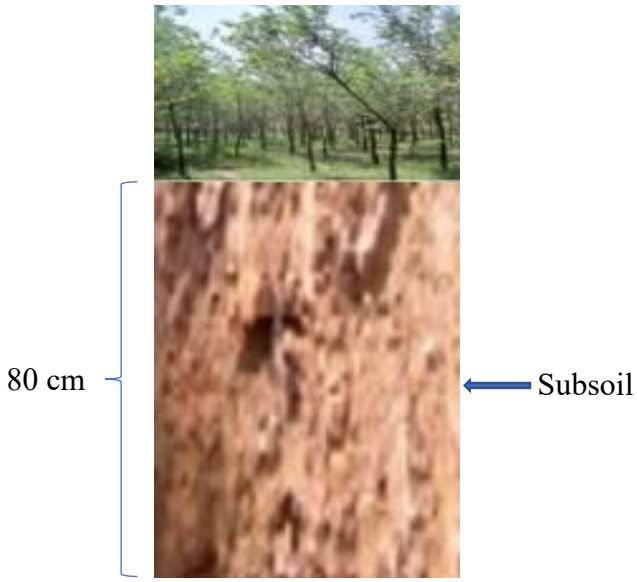


Figure 3.6. Schematic profile of the E-9 soil.

3.2.5.2 The 18-year-old reclaimed site (E-18)

The profile at the E-18 pedon is an 18-year reclaimed soil occurs on a gently undulating landform at an elevation of 85 m above sea level. The dominant planted vegetation at this site consists of *Xylopia staudii*, *Funtumia elastica*, *Gongronema latifolium*, *Athyrium filx-femina*, and *Acacia kamerunensis.* A vertical cross section of the profile showed a uniformly packed 90 cm of subsoil material below the surface.

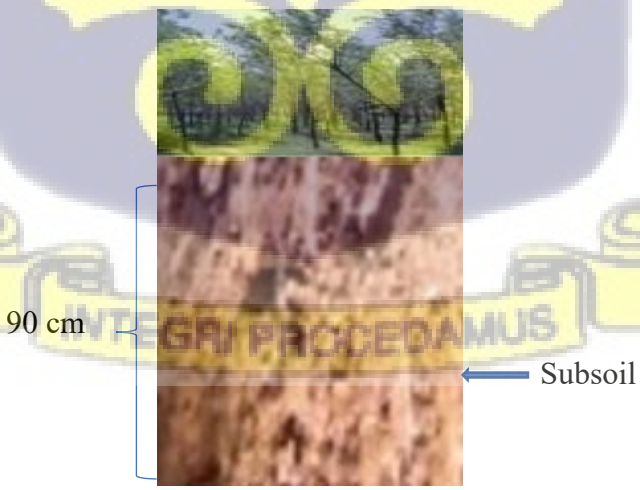


Figure 3.7. Schematic profile of E-18 soil.

3.2.5.3 The 21-year-old reclaimed site (E-21)

The 21-year-old reclaimed soil occurs on a gently undulating landscape at an altitude of 91 m above sea level. The area is predominantly occupied by planted trees, which consisted of *Xylopia staudii*, *Funtumia elastica* and *Khaya anthotheca*. The soil profile showed a uniformly packed 100 cm of subsoil material below the surface.

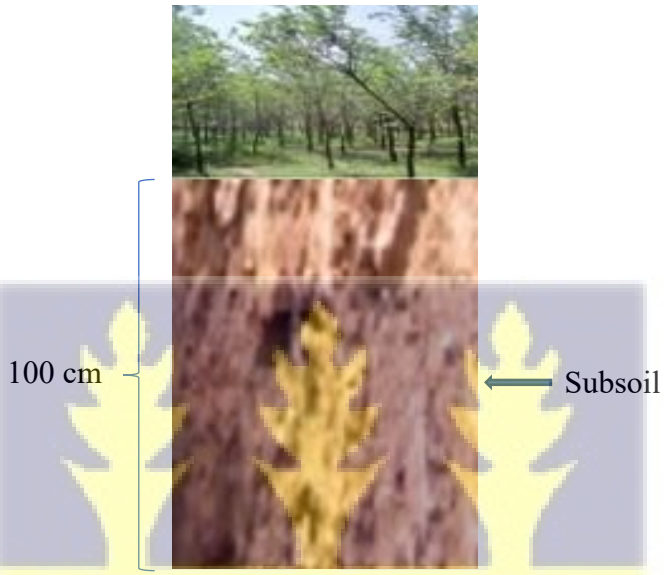


Figure 3.8. Schematic profile of the E-21 soil.

3.3 Soil sampling

On each site (A to E), a grid system covering an area of 10 m x 10 m (100 m^2) was laid down and from the grid, soil samples were collected from 0-20 cm depth using an auger at 1 m intervals along the midpoint section of each grid line. The collected samples were then bulked, mixed thoroughly and reduced by conning-quartering method to yield one composite sample. A profile pit (100 cm deep) was dug at the centre of the grid system and samples were collected at 10 cm intervals down the profile. Core samples were also taken within 10 cm intervals starting from the bottom to the top of the pits.

3.3 Biodiversity studies

3.3.1 Inventory of plant species

Information on plant species was collected between the months of February and March 2020. A 200 m transect was cut through each site and a 5 m x 5 m quadrat was placed at a 10 m interval along the transect. In each 5 m x 5 m quadrat, rooted tree species, shrubs, and climbers were identified, counted, and recorded. The floristic composition of the herb layer was also assessed within 1 m x 1 m quadrats randomly distributed in the 5 m x 5 m quadrat. The identification of plant species was done in-situ with the help of a plant taxonomist from the Forestry Commission of Ghana, Tarkwa. Photographs were taken of those plants that could not be identified. Some of them were preserved and taken to the forestry commission for proper identification. Families and species were identified using the guide by Hawthorne and Jongkind (2006). The floristic composition in the various sites and the natural forest was estimated using species richness, diversity, evenness, relative abundance and similarity coefficient indices.

3.3.2 Earthworm sampling

Sampling earthworms from the study sites was carried out in the early hours of the day in April and May 2020, during the rainy season using the formalin extraction method. At random on each study site, an area of 100 m² (10 m x 10 m) was delineated for sampling of earthworms. Then five sampling spots measuring 25 cm x 25 cm were marked at the selected sampling area. Four of the spots were at the vertices of the 100 m² area while the fifth was at the centre (See figure 3.9). Overgrown grasses and litter were removed from the sampling spots. Then a plastic quadrat measuring 25 cm x 25 cm was used to delineate the sampling spots. A 2% formalin solution was prepared and used for the extraction. For each sampling spot, 750 mL of diluted formalin was poured at a time. The earthworms that emerged at the surface after 20 minutes were collected.

Thereafter, the sampling spot was dug up to 10 cm depth and the soil put into a plastic bowl. All the remaining earthworms in this soil volume were searched and collected. The sampling process was repeated three times in each 25 cm x 25 cm quadrat.

The total number of earthworms collected were then separated into Epigeic, Endogeic, Anecic and counted. At the end of the sampling process, the excavated soils were returned to their original places. The sampling procedure was carried out in replicates of 3 per site.

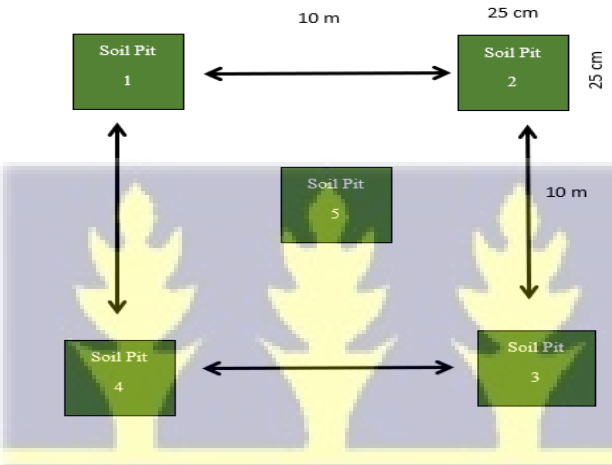


Figure 3. 9. A 10 m ×10 m earthworm sampling pattern.

3.3.3 Estimation of biological diversity estimating plant composition

The biological diversity of vegetation and earthworms of the study area were estimated using the following indices:

3.3.3.1 Shannon-Wiener index (H)

The Shannon-Wiener diversity index was calculated from the given expression:

$$H' = -\Sigma\left[\left(\frac{n_1}{N}\right) \times ln\left(\frac{n_1}{N}\right)\right] \quad \text{.....Eqn. [3.1]}$$

Where ***H*** is Species diversity index, ***ni*** is the number of individuals or amount of each species (i'th

species), ***N*** is total number of individuals (or amount) for the site and ***ln*** is the natural log of the number.

3.3.3.2 Species richness

Species richness was estimated using the Margalef (1958) index.

$$\boldsymbol{Margalef's\ index} = \frac{(S-1)}{ln\ N} \text{..... Eqn. [3.2]}$$

Where ***S*** is total number of species recorded, ***N*** is total number of individuals in the sample and ***ln*** is natural logarithm.

3.3.3.3 Species evenness (equitability)

Species evenness was calculated using Pielou (1966) evenness index. The Pielou index J' is defined as:

$$\mathbf{J'} = \frac{\mathbf{H'}}{\mathbf{ln(S)}} \text{..... Eqn. [3.3]}$$

Where ***J'*** is Pielou's evenness index, ***S*** is the total species number in the sample, ***H'*** is Shannon-Wiener diversity index and ***ln*** is natural logarithm.

3.3.3.4 Similarity coefficient

The similarity was calculated using the Czekanowski index formulae below:

$$\boldsymbol{S_C} = \frac{2\sum_{i=1}^{n} \boldsymbol{min}(X_i Y_i)}{\sum_{i=1}^{n} X_i + \sum_{i=1}^{n} Y_i} \text{..... Eqn. [3.4]}$$

Where $\boldsymbol{X_i}$ and $\boldsymbol{Y_i}$ are abundance of species *i*, $\sum_{i=1}^{n} \boldsymbol{min}(X_i Y_i)$ is sum of the lesser scores of species *i* where it occurs in both quadrats and ***n*** is number of species.

3.3.3.5 Density

The density was estimated as:

$$\mathbf{Density} = \frac{\mathbf{Number\ of\ individual\ species}}{\mathbf{Area\ sampled}} \text{..... Eqn. [3.5]}$$

3.4 Soil morphological characteristics

The profiles and horizons were described according to the guidelines for soil profile description (FAO, 2006, Schoeneberger et al. 2012). The descriptions include horizon thickness and sequence, colour, structure, consistence, roots distribution and coarse fragments.

3.5 Laboratory analysis

3.5.1 Bulk density

Metal cylindrical sampling cores with sharp cutting edges, measuring 5 cm in diameter and 5 cm in height, were driven into the ground. The sampling cores were dug out and extra dirt was removed from the cylinder's top and bottom sides using a flat knife. The soil core samples were then packed in labelled plastic bags and delivered to a laboratory for bulk density analysis. Soil samples were placed into a moisture container and oven-dried at 105 ºC for 48 hours in the laboratory. The bulk density of the soil was determined by dividing the weight of the oven-dried soil by the core sampler's volume.

$$\rho b = \frac{\text{weight of oven dry soil}}{\text{volume of soil}} \quad \text{Eqn. [3.6]}$$

3.5.2 Particle size distribution

Using Day's (1965) modified Bouyoucos hydrometer technique, the different size fractions of the soil were analysed. A 40 g of air-dried, 2.0 mm soil was weighed into a beaker and 50 mL of 30% H_2O_2 was then added to destroy the soil organic matter. The suspension was then mixed with 100 mL of 5% sodium hexametaphosphate solution and agitated in a mechanical shaker for two hours after being let to stand for ten minutes. After that, the suspension was poured into a graduated cylinder and filled to the 1000 mL mark with distilled water. The suspension was let to stand so that it could adjust to ambient temperature. Using a plunger, the contents of the cylinder were

extensively stirred. The hydrometer and temperature were then measured after 5 minutes and 5 hours. After 5 minutes and 5 hours, a blank hydrometer measurement of 5% sodium hexametaphosphate solution with 1000 mL added was taken. After reading the hydrometer for five hours, the suspension was poured over a 50 µm sieve and the effluent was discarded. The particles remaining on the 50 µm sieve were completely cleaned with distilled water, transferred to a moisture container, oven-dried at 105 ºC for 24 hours and cooled in a vacuum desiccator. The weight of the dried samples was used to indicate the sand component. Based on the difference between the 5 minute and 5-hour data, the clay and silt fractions were calculated. The percentage clay and silt were estimated using the relation below.

$$\%(\mathbf{Clay} + \mathbf{Silt}) = \frac{\mathbf{Hydrometer\ reading\ at\ 5mins}}{\mathbf{weight\ of\ soil\ (g)}} \times \mathbf{100} \quad \mathbf{Eqn.\ [3.7]}$$

$$\mathbf{Clay}\ (\%) = \frac{\mathbf{Hydrometer\ reading\ at\ 5\ hours}}{\mathbf{weight\ of\ soil\ (g)}} \times \mathbf{100} \quad \mathbf{Eqn.\ [3.8]}$$

$$\mathbf{Silt}\ (\%) = [\%\mathbf{Clay} + \mathbf{Silt}] - \mathbf{Clay}\ (\%) \quad \mathbf{Eqn.\ [3.9]}$$

$$\mathbf{Sand}\ (\%) = \frac{\mathbf{Weight\ of\ oven\ dry\ sand\ retained\ on\ the\ 50\ \mu m\ seive}}{\mathbf{weight\ of\ soil\ (g)}} \times \mathbf{100} \quad \mathbf{Eqn.\ [3.10]}$$

The textural class was based on the USDA-NRCS textural triangle (Appendix 2).

3.5.3 Aggregate stability

The dry sieving technique was used to measure the aggregate stability of the reclaimed sites and the natural forest (Kemper and Rosenau, 1986). Triplicate random samples of the topsoil were taken from the 10 m^2 grid sampling area in each site. At each sampling area, undisturbed soil samples were taken from the 20 cm topsoil with a spade to avoid compression and disturbance of the samples. Only the part of the sample not touched by the spade were used. The collected samples

were then placed in rigid large sampling box, labeled appropriately and transported to the laboratory. Prior to analysis, plant roots and stones were extracted from the samples stored in a controlled environment. Then, 100 g of each sample was carefully weighed and placed on top of the nest sieve (4 mm) in the rotary equipment for sieving (oscillating sieving analyzer, JH-200, Beijing, China) and subjected to two minutes of continuous shaking at 270 rpm. The rotational sieving equipment had five sieves with aperture dimensions of 4, 2, 1, 0.5, 0.25, and 0.25 mm. The distributed soil aggregates were collected separately in each sieve and weighed on an electronic balance to calculate the distribution of the (> 4, 4–2, 2–1, 1–0.5, and 0.5–0.25 mm aggregate size classes. The residual soil sample in each sieve was recorded in order to compute the aggregate stability and mean weight diameter of dry soil using the following calculations. Each sieve nest's dry-stable aggregate was calculated as follows:

$$\mathbf{SAS} = \mathbf{100} \times \left(\frac{\mathbf{SA}}{\mathbf{SA} + \mathbf{NA}}\right) \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \mathbf{Eqn.\ [3.11]}$$

Where ***SAS*** is soil aggregate stability (%), ***SA*** is weight of stable aggregate (g) and ***NA*** is weight of unstable aggregate (g). Using the weights of the above aggregate size classes, the mean weight diameter MWD (mm) was calculated using the following equation (Hillel, 2004):

$$\mathbf{MWD} = \sum_{\mathbf{i=1}}^{\mathbf{n}} \mathbf{Xi\ Wi} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \mathbf{Eqn.\ [3.12]}$$

Where $\boldsymbol{X_i}$ is the mean diameter of each aggregate size classes (mm) and $\boldsymbol{W_i}$ is the weight percentage of each aggregate size classes with respect to the total sample.

3.5.4 Moisture characteristics

By following the pressure plate extraction technique, the soils' moisture content was determined (Klute, 1986). Soil samples that had been disturbed were placed in 2-centimeter-tall rings,

saturated with water using a sand table, and then dried to field capacity (1/3 bar or 33 kPa) and wilting point (15 bar or 1500 kPa). The difference in moisture content of the soil at wilting point and at field capacity was then calculated as a measure of available moisture content of the soil.

3.5.5 Soil pH

Using an electrode MV88 Praitronic pH meter and a ratio of 1:2 (soil to solution), the pH of the soil was evaluated in both distilled water and 0.01 M $CaCl_2$. In a 50 mL beaker, a 10 g soil sample was weighed, and 20 mL of solution was added. Allowing the suspension to settle and equilibrate at room temperature, the liquid was agitated with a glass rod for 30 minutes and allowed to settle for 1 hour. The pH meter was calibrated using pH 4.0 and pH 7.0 standard buffer solutions (Sigma Chemicals, USA). On the electronic pH meter, the pH of the supernatant was measured. For each soil sample, triplicate analyses were performed on distinct subsamples, and the mean results were calculated.

3.5.6 Exchangeable bases

For each soil, 10 g of air-dried samples were weighed into an extraction bottle, and 100 mL of a pH 7.0 buffered, 1.0 M ammonium acetate solution was added. The bottle's contents were put in the mechanical shaker and agitated for one hour. The content was filtered into a clean, empty plastic bottle using No. 42 Whatman filter paper. An aliquot of the extract was analysed using a Perkin Elmer atomic absorption spectrophotometer for Ca, Mg, K, and Na.

3.5.7 Cation exchange capacity and base saturation

For each site, 10 g of air-dried soil samples were weighed into an extraction vial, and 100 mL of a pH 7.0 buffered, 1.0 M ammonium acetate solution was added. The bottle's contents were put in the mechanical shaker and agitated for one hour. The content was filtered into a clean, empty

plastic bottle using No. 42 Whatman filter paper. The sample was then leached with four 25 mL volumes of methanol to remove excess ammonium from empty plastic bottles. The soil was then leached with four 25 mL volumes of 1 M acidified potassium chloride in clean, empty plastic bottles. A 5.0 mL aliquot of the digest was pipetted into a Markham distillation apparatus, followed by the addition of a 5.0 mL aliquot of a 40% NaOH solution. Then, 5.0 mL of 2% boric acid and a few drops of mixed indicator were added to a conical flask and positioned under the condenser's delivery tube so that the tip was below the liquid's surface. The flask was taken from the still, the condenser tip was washed, and the distillate was titrated with 0.01 M HCl until the color changed from green to purple. From the number of moles of HCl used in the back titration procedure, the cation exchange capacity in $cmol_c\ kg^{-1}$ soil was then estimated. The CEC calculation was as follows:

$$\mathbf{CEC\ (cmol\ kg^{-1})} = \frac{(\mathbf{a-b}) \times \mathbf{0.1} \times \mathbf{v} \times \mathbf{MCF} \times \mathbf{100}}{\mathbf{w} \times \mathbf{al}} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \mathbf{Eqn.\ [3.13]}$$

Where ***a*** is titre volume of 0.1 N HCl for the sample (ml), ***b*** is titre volume for the blank (ml), ***v*** is volume of the extracting solution (ml), ***MCF*** is moisture correction factor $\left(\frac{100+\%\ \text{Moisture}}{100}\right)$, ***w*** is weight of the sample (g) and ***al*** is sample aliquot (ml). Total base saturation was calculated by summing together the levels of calcium, magnesium, potassium, and sodium found in the soil; then expressed this sum as a percentage of the CEC value.

3.5.8 Exchangeable acidity

The method of Thomas (1982) was used to extract aluminum and hydrogen (Al^{3+} and H^{+}) ions from the samples using 1 N KC1. A 100 mL plastic extraction container was filled with ten grams of the fine earth. Twenty-five milliliters of 1 N KC1 were added, then the contents were well mixed. The suspension was allowed to stand for 30 minutes before being transferred to a buchner funnel

equipped with filter paper and set on a 250 mL vacuum flask. The sample was then leached with an additional 125 ml of 1 N KC1 in increments of 25 mL. Using phenolphthalein as an indicator, the exchange acidity was then evaluated by titrating the extracts with 0.1 M NaOH.

3.5.9 Total carbon and nitrogen

The total carbon and total nitrogen contents of the soil samples were determined by dry combustion method (Dumas's method) using a CNS elemental analyzer (Leco Trumac version 1.3). A 0.5 g of 2 mm sieved soil sample was weighed into platinum crucibles and combined with 500 mg of comcat in each. For the purpose of ensuring the accuracy of measurements, standard samples with known C and N concentrations were produced. The autosampler/furnace of the analyzer was loaded with both the soil samples and the reference samples. Six minutes after combustion, the C and N concentrations of the samples were measured. The following formula was used to quantify the organic matter content of soil samples:

$$\textbf{Organic matter}\ (\%) = \textbf{Total carbon}\ (\%) * \mathbf{1.724} \quad \textbf{Eqn. [3.14]}$$

3.5.10 Determination of total phosphorus

For each soil, a 2.0 g sample screened through 0.5 mm sieve was weighed into a 250 mL Erlenmeyer flask. Then, 10 mL of concentrated HNO_3 and 15 mL of 60% $HClO_4$ were added, and the combination was digested until the digest became colourless and the thick white vapours of $HClO_4$ disappeared. The digest was cooled before being diluted with distilled water. The digest was filtered using No. 42 Whatman filter paper into a 100 mL volumetric flask, which was then filled to capacity. In the next step, a 5.0 mL aliquot of the digest was pipetted into a 50 mL volumetric flask. A few drops of P-nitrophenol indicator and a few drops of 4 M NH_4OH were added till the sample solution became yellow to correct the pH. Afterwards, 8.0 mL of a solution

comprising concentrated sulphuric acid, ammonium molybdate, potassium antimony tartrate, and ascorbic acid was added, and the volume was brought to the correct level with distilled water. The solution was completely combined and allowed to stand until a blue hue appeared. At a wavelength of 712 nm, both standard and sample absorbance were measured. Total phosphorus content of the soil was calculated as:

$$\mathbf{P}\left(\frac{\mathbf{mg}}{\mathbf{kg}}\right) = \frac{(\mathbf{spectrometer\ reading} - \mathbf{blank\ reading}) \times \mathbf{volume\ of\ extract}}{\mathbf{Volume\ of\ aliquot} \times \mathbf{sample\ weight}} \ldots\ldots\ldots \mathbf{Eqn}\ [\mathbf{3.15}]$$

3.5.11 Available phosphorus

The Bray and Kurtz (1945) approach was used to measure the available P content of acidic soils. To 5.0 g of air-dried soil sample that was weighed in an extraction bottle, a 50 mL of the extractant (0.03 M NH_4F in 0.025 M HCl) was added to make a soil-to-solution ratio of 1:10 (soil: extractant). The suspension was violently shaken for precisely two minutes using a mechanical shaker. Following the filtration of the sample through Whatman No. 42 filter paper, a 5.0 mL aliquot was pipetted for colour development. At a wavelength of 712 nm, the absorbance of standards and samples was determined. Available phosphorus content of the soil was calculated as follows:

$$\mathbf{P}\left(\frac{\mathbf{mg}}{\mathbf{kg}}\right) = \frac{(\mathbf{spectrometer\ reading} - \mathbf{blank\ reading}) \times \mathbf{volume\ of\ extract}}{\mathbf{Volume\ of\ aliquot} \times \mathbf{sample\ weight}} \ldots\ldots\ldots\ldots\ldots \mathbf{Eqn}\ [\mathbf{3.16}]$$

3.5.12 Soil microbial biomass carbon and nitrogen

The soil microbial biomass carbon and nitrogen was estimated following the chloroform fumigation incubation method of Brookes et al. (1985). Field moist soil samples were sieved through a 2 mm mesh sieve to remove stones, coarse roots, and all visible litters. Ten grams of each sieved sample was weighed into a 50 mL beaker. For each soil sample, three replicates were made. The beakers with the soils were placed in a vacuum desiccator. A shallow dish containing

30 ml of alcohol-free chloroform and boiling chips was placed in the middle of the desiccator. The lid of the desiccator was closed and sealed. Thereafter, the desiccator was evacuated with a vacuum pump until the chloroform boiled vigorously for 5 minutes. Thereafter, the valve on the desiccator was closed and the desiccator kept in the dark for 5 days at 25° C. Next, another 10 g of same soil samples were weighed in three replicates into a 50 mL beaker for unfumigated extraction and then kept in a separate desiccator and incubated for 5 days.

After 5 days, the soil samples were then transferred into 100 mL bottle each. Fifty millilitres (50 mL) each of 0.5 M K_2SO_4 solution was added to the soil samples, capped, placed and shaken on a horizontal shaker at 200 revolution per minutes for 2 hours (Ghani et al. 2003). The soil suspension was allowed to settle and the supernatant filtered the through filter paper (Whatman No. 42) to get the extracts for both the fumigated and unfumigated soil samples.

To determine microbial biomass carbon, an aliquot of five millilitres (5 mL) each of the extracts were pipetted into Erlenmeyer flasks. Ten millilitres (10 mL) of 0.17 M of $K_2Cr_2O_7$ solution serving as an oxidizing agent was dispensed into the flask. Twenty millilitres (20 mL) of concentrated H_2SO_4 were added to the solution as a source of heat to speed up the rate of reaction and swirled for a few seconds. The solutions were left to stand for about 30 min to cool. After cooling, 200 mL of deionized water was added to each solution. Five millilitres (5 mL) of orthophosphoric acid and 3 drops of barium diphenylamine sulphanate indicator were added to the solutions and titrated against 0.2 M $Fe(NH_4)_2(SO_4)_2.6H_2O$ solution to a green endpoint. Three replicates will be maintained in each case.

To determine microbial biomass nitrogen, an aliquot of 5 mL of the extracts was pipetted each into a Kjeldahl flask. Five millilitres (5 mL) of 40% sodium hydroxide was pipetted into the flask and

swirled for a few seconds, the ammonia (NH_3) was distilled into 5 mL of 2% boric acid (containing a methylene blue and methyl red indicator mixture) in a 150 mL conical flask to trap the NH_3. This changes the colour of the boric acid to green. Fifty millilitres of the distillate were collected. After distillation, the distillate was back titrated against 0.01 M HCl to a purple endpoint. Three replicates were maintained in each case. Microbial biomass C (BC) was calculated following Wu et al. (1990):

$$\boldsymbol{BC = \frac{EC}{KEC}} \text{..... Eqn. [3.17]}$$

Where ***EC*** is extractable C in fumigated soil extracts – extractable C in non-fumigated soil extracts and ***KEC*** is 0.45 (extractable part of microbial C after fumigation). Microbial biomass N was calculated according to Jenkinson (1988):

$$\mathbf{BN = \frac{EN}{KEN}} \text{ ... ... Eqn. [3.18]}$$

Where ***EN*** is total N extracted from fumigated soil - total N extracted from unfumigated soil, ***KEN*** is 0.45 (extractable part of microbial N after fumigation).

3.5.13 Dehydrogenase determinations

The dehydrogenase activity (DHA) was estimated using the method described by Casida et al. (1964). A 20 g of fresh homogenized air-dried soil sample (< 2 mm) and 0.2 g of $CaCO_3$ in ratio was weighed into a test tube (18×150 mm) and mixed thoroughly. After $CaCO_3$ addition, the soil was dispensed in 6 g portions into each of three test tubes (18 by 150 mm). Thereafter, 1 ml of 3% aqueous solution of 2, 3, 5-triphenyltetrazolium chloride and 2.5 ml of distilled water was added to each test tube. A blank sample was similarly prepared with 1ml distilled water. The content of each test tube was mixed thoroughly with a sterile glass rod; then rubber stoppers were inserted,

and the tubes were incubated in the dark at 37 °C for 24 hours. After incubation, 10 ml of methanol was added to each sample and shaken for 1 minute. The content was allowed to stand in dark for six hours. The suspension was filtered into a glass funnel plugged with absorbent cotton placed on a 100 ml volumetric. The soil in each test tube was transferred with methanol into the funnel. Additional methanol (in 10 ml portions) was added to the funnel until the reddish color disappeared from the cotton plug. The filtrate was diluted to a 100 ml volume with methanol. The obtained supernatant liquid (red methanolic solutions of the formazan) was poured into a clean tube, and the absorbance of the solution were read at 485 nm against the extract from the non-2, 3, 5-triphenyltetrazolium chloride soil blank by using a Spectronic 20 colorimeter (Bausch and Lomb, Rochester, N.Y). Triphenylformazan (TPF) concentration was calculated using a calibration curve (prepared according to the standard method).

3.5.14 Permanganate oxidizable carbon

The procedure for the determination of permanganate oxidizable carbon is detailed by Vieira et al. (2007). One gram of a 2 mm soil sample was weighed into an extraction bottle, followed by the addition of 25 ml of 0.001332 M $KMnO_4$. The container was closed tightly and the contentment was shaken for 30 minutes in a mechanical shaker. Thereafter, the content was filtered through Whatman No. 42 filter paper. The filtrate was further diluted 250 times, and its absorbance at 565 nm was measured using a spectrophotometer. The absorbance of five additional concentration standards, 0.000270 M, 0.00285 M, 0.000300 M, 0.000315 M, and 0.000330 M, was also measured, and a standard curve was constructed. The spectrophotometer was read at the same wavelength as before.

The following is how the Permanganate Oxidizable Carbon (labile carbon) was determined:

$$\mathbf{POXC\ (mg/kg\ soil)} = \left[\mathbf{1.332} \times \mathbf{10^{-3}} \boldsymbol{mol\ L^{-1}} - (\mathbf{a} + \mathbf{b} \times \mathbf{Abs})\right] \times \left(\mathbf{9000}\ \boldsymbol{mg\ C\ mol^{-1}}\right)(\mathbf{0.333\ L\ solution/weight}) \ldots\ldots\ldots \mathbf{Eqn.}\ [\mathbf{3.19}]$$

3.5.15 Soil quality determination

Estimation of SQI of the natural forest and the six reclaimed mined soils was carried out following three basic procedures of Andrews (1998) (Figure 3.10):

(1) selecting appropriate minimum data set (MDS);

(2) transforming indicator scores; and

(3) integrating the indicator scores into soil quality index.

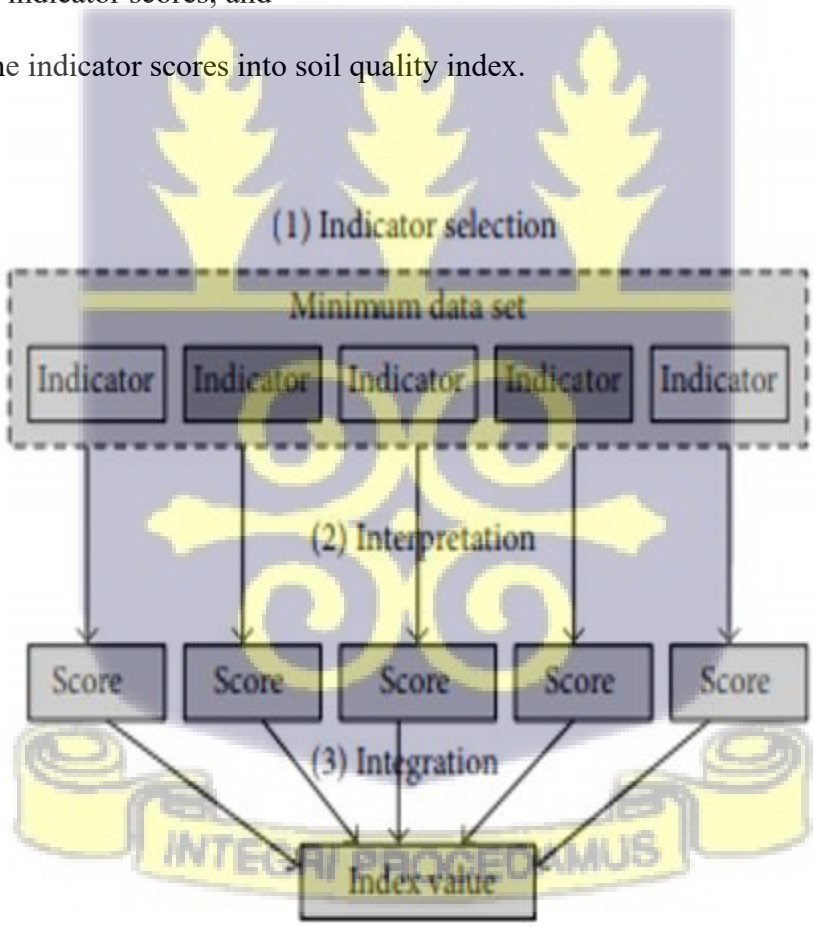


Figure 3.10. A conceptual model for computing soil quality indices. (Adopted from Andrews, 1998).

3.5.15.1 Indicator selection

The whole data set of the soil properties (untransformed data) from all six reclaimed sites and the natural forest were transposed into the PCA model using SPSS, version 21.0 to select the individual PCs and the MDS. The principal components (PCs) with eigenvalues > 1 (Kaiser, 1960) were selected and subjected to varimax rotation. Only variables with higher factor loadings were maintained for indexing under a given principal component. High factor loadings were defined as absolute values within 10 percent of the highest factor loading (Andrews et al. 2002). When more than one variable was maintained under a single PC, their correlations were assessed to see if the variable was redundant and, thus, could be deleted from the MDS (Masto et al. 2008). If the highly loaded variables were not associated, each was deemed significant and kept in the SQI. For the SQI, the variable with the greatest factor loading (absolute value) was selected from among the variables having strong correlations. Each PC explained a specific degree of variance (percent) in the whole data set; this percentage determined the weight of variables selected under a certain PC. All of the highlighted and underlined soil characteristics in were chosen as the final MDS.

3.5.15.2 Indicator transformation/scoring function

After selecting the MDS using PCA, each real value of the soil properties was converted into scores (S) using an equation that defines a sigmoidal type (Sinha et al. 2009; Mukhopadhyay et al. 2014).

$$x = \frac{a}{\left[1+\left(\frac{x}{x_o}\right)^{b}\right]} \quad \text{.....Eqn. [3.20]}$$

Where x is the soil property value (i. e. value of pH, EC, and SOC), a is the maximum score (=1.00) of the soil property, x_o is the mean value of each soil property and b is the value of the slope of the equation. The slope was -2.5 for the 'more is better curve' and 2.5 for the 'less is better curve' to

obtain a sigmoidal curve tending to 1 for all the proposed properties. After calculating the S values for all the indicator parameters, each property was then weighted using the PCA results. The final normalized weights were calculated as per the equation below.

$$\textbf{Normalized weight} = \frac{\mathbf{PCA\sigma_i}^2}{(\sum \mathbf{W_i} * \mathbf{100})} \ldots\ldots\ldots\ldots\ldots\ldots \textbf{Eqn. [3.21]}$$

Where $PCA\sigma^2$ is the variance explained by the individual PCs, W_i is the weight of the soil properties selected by PCA.

3.5.15.3 Integration into index value

Having determined the weight of each selected indicator property (W) and their respective scores (S), the final PCA based SQI was calculated using this equation:

$$\textbf{SQI} = \sum_{i=1}^{N} \mathbf{W_i S_i} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \textbf{Eqn. [3.22]}$$

Where W is the weight of the soil properties selected by PCA, S is their respective score and N is total number of individuals for the site.

CHAPTER FOUR

BIODIVERSITY STUDIES

4.1 Introduction

Global climate change has emerged as an accelerator of biodiversity loss in recent times. This biodiversity loss is projected to become a progressively more significant threat in the coming decades (Convention on Biological Diversity, 2010) since anthropogenic influence on the environment has become pervasive.

Biological diversity, often shortened to biodiversity (Wilson and Peter, 1989), refers to the existence of many different species of plants and animals in a quantified area (Noss and Harris, 1986). It is the variety of life on earth, in all its forms and interactions. Biodiversity is defined according to the Convention on Biological Diversity of the United Nations (1992) as the variation among living organisms from all sources, including terrestrial, marine, and other aquatic ecosystems and the ecological complexes of which they are a part, as well as diversity within species, between species, and among ecosystems. Odum (1975) also defines biodiversity as the total number and variety of organisms within a specific region.

Biological diversity measures the degree of heterogeneity or stability of an ecosystem (Trichon, 1997) which is often considered as an index of success in reclamation activities (Martin et al. 2005). Biodiversity can be estimated using mathematical formulae to index different attributes of diversity in a given habitat (Odum, 1975). Conventionally, biodiversity is measured by taking into account both the number of species and the number of individuals of each species (Noss and Harris, 1986). High biodiversity is perceived as synonymous with ecosystem health. In general, diverse ecosystems are believed to have increased stability, increased productivity, and resistance to

invasion and other disturbances (OTA, 1987).

Practically, all of earth's ecosystems have been dramatically transformed through anthropogenic activities which have resulted in increased species extinction rate (Odum, 1975). Characteristically, mining activities over the years, have rendered vast arable lands expandable resulting in substantial destruction of flora and faunal communities (Sheoran et al. 2010). These changes in important components of biological diversity are more rapid and devastating in open pit or strip-mining system (Asensio et al. 2013) which represent loss of biodiversity (Millennium Ecosystem Assessment – MEA, 2005). Mining companies undertake a series of activities to restore the productivity of degraded mined land after mining. This involves establishing new vegetation on degraded mined site with a mixture of different plant species to provide varied forms of potential habitats for faunal community (Parrotta et al. 1997; Davis et al. 2012) that is usually similar to what had existed prior to mining. The productivity of the re-established vegetation facilitates the rate of re-establishment of ecosystem processes, such as soil nutrient cycling and faunal retention. Also, the works of Macdonald et al. (2015) reveals that establishing and maintaining an ecosystem with plant species of different successional status can result in increased forest productivity and diversity of flora and fauna.

The rehabilitated site is subsequently monitored and evaluated periodically to establish the status of restoration and the extent of biodiversity recovery. But, evaluating the state of biodiversity of an ecosystem requires the use of reliable indicators that would capture changes that occur therein. According to the European Commission (2010), the selection of indicator organisms should be based on their availability, significance, measurability and costs. Vegetation and earthworms are the most reported indicator organisms used to assess ecosystem recovery. These two indicators are the most abundant in a given habitat, easy to assess at less cost and their role in ecosystem service

provision is paramount. However, the assessment of biodiversity of an ecosystem in the AngloGold Ashanti Iduapriem mines has been mainly based on arable lands and natural forests. Few studies conducted on reclaimed mine soils focused mainly on the use of vegetation as an indicator parameter. These assessments seldom consider the combined use of vegetation and earthworms as indicators for measuring ecosystem recovery. Yet, both are crucial components of a healthy and productive ecosystem. Data collected on the type of plant and earthworm species and their population from different habitats may provide useful information on the quality and strength of a rehabilitated ecosystem.

The objectives of this study are to:

a. Evaluate the floristic composition and species diversity of the study sites;

b. Determine earthworm densities per functional category of the study sites; and

c. Determine the distribution and diversity of earthworms in the study sites.

4.2. Results

The results of the biodiversity of the vegetation and earthworms in the rehabilitated area and the natural forest are presented in the following sections. The selected areas (A to E) are referred to as sites.

4.2.1 Floristic composition

A total of 2225 plants made up of 74 species and 36 families were identified at the study sites. Six forms comprising, 831 trees (tall, with a single main trunk), 103 shrubs (medium-sized woody plants with multiple stems), 797 herbs (small soft plants, non-woody stems), 295 grasses (flowering plants belonging to the family Poaceae), 195 climbers (weak stems that grow upwards with support) and 4 lianas (a long-stemmed, woody vine that is rooted in the soil and climbs or

twines around other plants) were observed (Table 4.2). In terms of family, Sterculiaceae was dominant with 5 species, followed by Annonaceae, Asteraceae, Caesalpiniaceae and Rubiaceae with 4 species each. With regards to the number of trees per family, *Funtumia elastica* was the dominant species in the rehabilitated sites with 314 trees, followed by *Xylopia staudii, Khaya anthotheca*, and *Antrocaryon micraster* with 110, 33, and 20 trees respectively. Site A (natural forest) was dominated by *Macaranga bartei* with 47 trees, followed by *Baphia nitida* with 18 trees while *Alfzelia africana* and *Lophia alata* recorded 10 trees each. Out of the 103 shrubs recorded from the study sites, Sites B recorded 46, C had 45, E-9 had 10 while D had 2. With a total of 797 herbs observed from the study area, *Athyrium filx-femina* dominated with 793 species followed 4 individuals of *Thaumatococcus daniellii*. While all the 4 individuals of *Thaumatococcus daniellii* were found on Site A, the 793 *Athyrium filx-femina* were observed on the rehabilitated sites (Table 4.1).

Two species of grasses were identified from the study area. The dominant grass species was the *Cynodon dactylon* with 286 individuals was found on Site C followed by *Mapnia baldwinii* with 9 individuals on Site B. Climbers were the 4th highest in terms of form from the study area with 195 individuals. Whereas sites E-18 and A recorded 60 and 46 individual climbers respectively, D, B, E-9, C and E-21 recorded 37, 17, 15, 12 and 8 individual climbers.

Table 4. 1. Plant species observed at the study area.

Family Name	Species Name	Habit/Form	No. of plants	Site B	Site C	Site D	Site E			Site A
							E-9	E-18	E-21	control
Anacardiaceae	Antrocaryon micraster	Tree	20	x	x	x	**2**	**4**	**10**	**4**
Annonaceae	Cleistopholis potens	Tree	2	x	x	x	x	x	x	**2**
	Monodora myristica	Tree	3	x	x	x	x	x	**2**	**1**
	Enantia polycarpa	Tree	4	x	x	x	x	**1**	**1**	**2**
	Xylopia staudii	Tree	110	**4**	x	**28**	**3**	**22**	**47**	**6**
Apocynaceae	Funtumia elastica	Tree	314	**38**	**1**	**53**	**44**	**73**	**100**	**5**
	Astonia boonei	Tree	2	**2**	x	x	x	x	x	x
Arecaceae	Sclerosperma mannii	Tree	1	x	x	1	x	x	x	x
Burseraceae	Canarium schweinfurthii	Tree	8	x	x	**6**	x	x	**2**	x
	Dacryodes klaineana	Tree	2	x	x	x	x	x	x	**2**
Caesalpiniaceae	Amphimas pterocarpoides	Tree	1	x	x	x	x	x	x	**1**
	Daniellia ogea	Tree	14	x	x	x	x	**5**	x	**9**
	Distemonanthus benthamianus	Tree	1	x	x	x	x	x	x	**1**
	Dialium aubrevillei	Tree	3	x	x	x	**1**	x	x	**2**
Cannabaceae	Trema orientalis	Tree	1	x	x	x	x	x	x	**1**
	Parinari excelsa	Tree	2	x	x	x	x	x	x	**2**
Combretaceae	Strephonema pseudocola	Tree	7	x	x	x	x	x	x	**7**
Ebenaceae	Diospyros sanza-munika	Tree	5	x	x	**2**	x	x	**1**	**2**
Euphorbiaceae	Drypetes floribunda	Tree	8	x	x	x	**2**	**1**	x	**5**
	Macaranga bartei	Tree	55	**3**	x	x	**5**	x	x	**47**
	Margaritaria discoidea	Tree	2	x	x	x	**2**	x	x	x
Fabaceae	Leucaena leucocephala	Tree	15	**5**	**5**	x	**5**	x	x	x
	Acacia mangium	Tree	17	**12**	**1**	x	x	x	**4**	x

Table 4.1. Continued.

Family Name	Species Name	Habit/Form	No. of plants	Site B	Site C	Site D	Site E			Site A
							E-9	E-18	E-21	control
Fabaceae	Acacia mearnsii	Tree	11	x	**5**	**6**	x	x	x	x
	Senna siamea	Tree	21	x	**2**	**7**	**6**	x	**6**	x
	Alfzelia Africana	Tree	10	x	x	x	x	x	x	**10**
Guttiferae	Allanblackia parvifloria	Tree	12	**1**	x	**2**	x	x	**1**	**8**
	Omphalocarpum ahia	Tree	3	x	x	x	x	**1**	x	**2**
Irvingiaceae	Irvingiaceae gabonensis	Tree	2	x	x	x	x	x	x	**2**
Melastomataceae	Memecylon latifolium	Tree	6	x	x		x	**1**	x	**5**
Meliaceae	Khaya anthotheca	Tree	33	x	x	x	x	x	**33**	x
	Carapa procera	Tree	4	x	x	x	x	x	x	**4**
Mimosaceae	Parkia bicolour	Tree	1	x	x	x	x	x	x	**1**
	Albizia zygia	Tree	3	x	x	x	**1**	**2**	x	x
Moraceae	Musanga cercropioides	Tree	1	x	x	x	**1**	x	x	x
	Myrianthus arboreus	Tree	4	x	x	x	**1**	x	x	**3**
Ochnaceae	Lophia alata	Tree	10	x	x	x	x	x	x	**10**
Olacaceae	Strombosia glaucescens	Tree	8	x	x	**2**	**1**	x	x	**5**
Palmaceae	Raphia hookeri	Tree	1	x	x	x	x	**1**	x	x
	Elaeis guineensis	Tree	11	x	x	**1**	**1**	**4**	**5**	x
Papilionaceae	Baphia nitida	Tree	25	x	x	x	**2**	**4**	**1**	**18**
Rubiaceae	Hellea stipulosa	Tree	5	**5**	x	x	x	x	x	x
	Zanthoxylum leprieurii	Tree	1	x	x	x	x	x	x	1
	Nauclea diderrichii	Tree	7	**7**	x	x	x	x	x	x
	Psydrax subcordata	Tree	12	**6**	x	**2**	**1**	**3**	x	x
Sapindaceae	Blighia sapida	Tree	1	x	x	x	**1**	x	x	x
Sapotaceae	Chrysophyllum albidum	Tree	1	x	x	x	x	x	x	**1**
	Tieghemella heckelii	Tree	1	x	x	x	x	x	x	**1**
Simouraceae	Hannoa Klaineana	Tree	5	x	x	x	x	x	x	**5**

Table 4.1. Continued.

Family Name	Species Name	Habit/Form	No. of plants	Site B	Site C	Site D	Site E			Site A
							E-9	E-18	E-21	control
Sterculiaceae	Heritiera utilis	Tree	3	**1**	x	**1**	**1**	x	x	x
	Cola gigantea	Tree	1	x	x	x	x	x	x	**1**
	Cola chlamydantha	Tree	4	x	x	4	x	x	x	x
	Majiodea fosteri	Tree	1	x	x	x	x	x	x	**1**
	Hildegardia barteri	Tree	14	x	x	x	x	x	x	**14**
Lecythidaceae	Napoleonaceae vogelii	Tree	9	x	x	x	x	x	x	**9**
Leguminoceae	Piptadeniastrum africanum	Tree	1	x		x	x	x	x	**1**
Loganiaceae	Strychnos aculeata	Tree	1	x	x	x	x	x	x	1
	Anthocleista nobilis	Tree	1	x	x	x	x	x	x	**1**
Fabaceae	Delbergia saxatilis	Liana	4	x	x	x	x	x	x	**4**
Fabaceae	Centrosema pubescens	Climber	21	**5**	**12**	**4**	x	x	x	x
	Acacia kamerunensis	Climber	95	x	x	**19**	**5**	**50**	x	**21**
Apocynaceae	Gongronema latifolium	Climber	74	**12**	x	**14**	**10**	**10**	**8**	**20**
Arecaceae	Laccosperma opacum	Climber	3	x	x	x	x	x	x	**3**
	Eremospatha macrocarpo	Climber	2	x	x	x	x	x	x	**2**
Verbenaceae	Sida acuta	Shrub	9	**4**	**5**	x	x	x	x	x
Compositaceae	Vernonia conferta	Shrub	33	**23**	**8**	x	**2**	x	x	x
Asteraceae	Alchornea cordifolia	Shrub	14	**5**	**1**	x	**8**	x	x	
	Lantana camara	Shrub	2	x	x	2	x	x	x	x
	Mimosa pudica	Shrub	4	**2**	**2**	x	x	x	x	x
	Chromolaena odorata	Shrub	41	**12**	**29**	x	x	x	x	x
Poaceae	Cynodon dactylon	Grass	286	x	**286**	x	x	x	x	x
Cyperaceae	Mapania baldwinii	Grass	9	x	x	x	x	x	x	**9**
Marantaceae	Thaumatococcus daniellii	Herb	4	x	x	x	x	x	x	**4**
Athyriaceae	Athyrium filx-femina	Herb	793	**1**	**236**	**196**	**270**	**83**	**7**	x

Table 4. 2. Floristic composition of the study area.

Site	Habit/form						No. of plants
	Trees	Shrubs	Herbs	Grasses	Climbers	Liana	
B	84	46	1	9	17	0	157
C	14	45	236	286	12	0	593
D	115	2	196	0	37	0	350
Site E							
E-9	80	10	270	0	15	0	375
E-18	122	0	83	0	60	0	265
E-21	213	0	7	0	8	0	228
A	203	0	4	0	46	4	257
Total	**831**	**103**	**797**	**295**	**195**	**4**	**2225**

4.2.2 Diversity of plant species

The relative abundance of individual species observed on each site was estimated (Figures 4.1a, 4.1b, 4.1c, 4.1d, 4.1e, 4.1f and 4.1g). Whereas 53.50% of the total number of plants observed at Site B was made up of trees, shrub constituted 29.30%, grasses and herbs made up of 5.73% and 0.64% respectively. Site C (ITSF) was dominated by lower forms of plants. Grass species from the Poaceae family dominated the site at 48.23% followed by herb with 39.80% abundance. Meanwhile, shrubs, trees and climbers constituted 7.59%, 2.36% and 2.02%, respectively. Although no grasses and lianas were observed at Site D, the site was dominated by herbs with 56% followed by 32.86% of trees and 10.57% climbers. Shrub was 0.57%.

Site E-9 was dominated by herbs with 72% followed by trees with 21.33%; shrubs and climbers constituted 2.67% and 4%, respectively but no grasses and lianas were observed at this site. At Site E-18, trees dominated with 46.04% followed by herbs with 31.32% and climbers with 22.64% with no shrubs, grasses and lianas. The entire E-21 site was made up of trees, climbers and herbs without grasses, shrub and lianas. A total of 93.42% of the observed plants were trees whereas

3.51% and 3.07% were climbers and herbs respectively. Site A (NF) was made up of 78.99% trees, 17.89% climbers while herbs and lianas had 1.56% each. Neither shrub nor grasses was found on the forest.

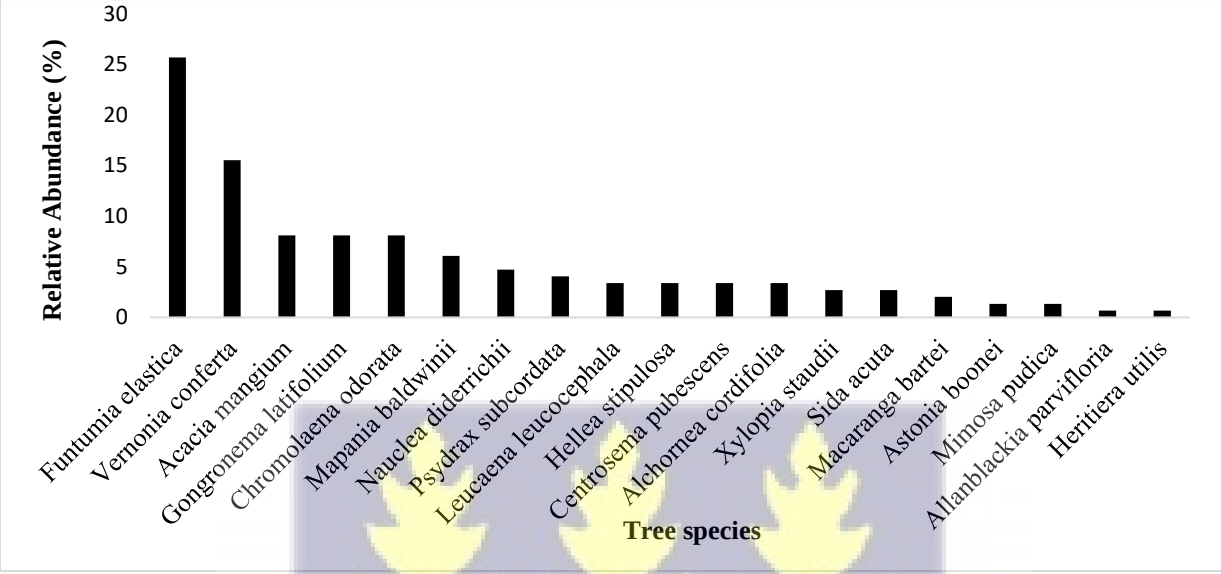


Figure 4.1a. Relative abundance of vegetation at Site B.

Figure 4. 1b. Relative abundance of vegetation at Site C.

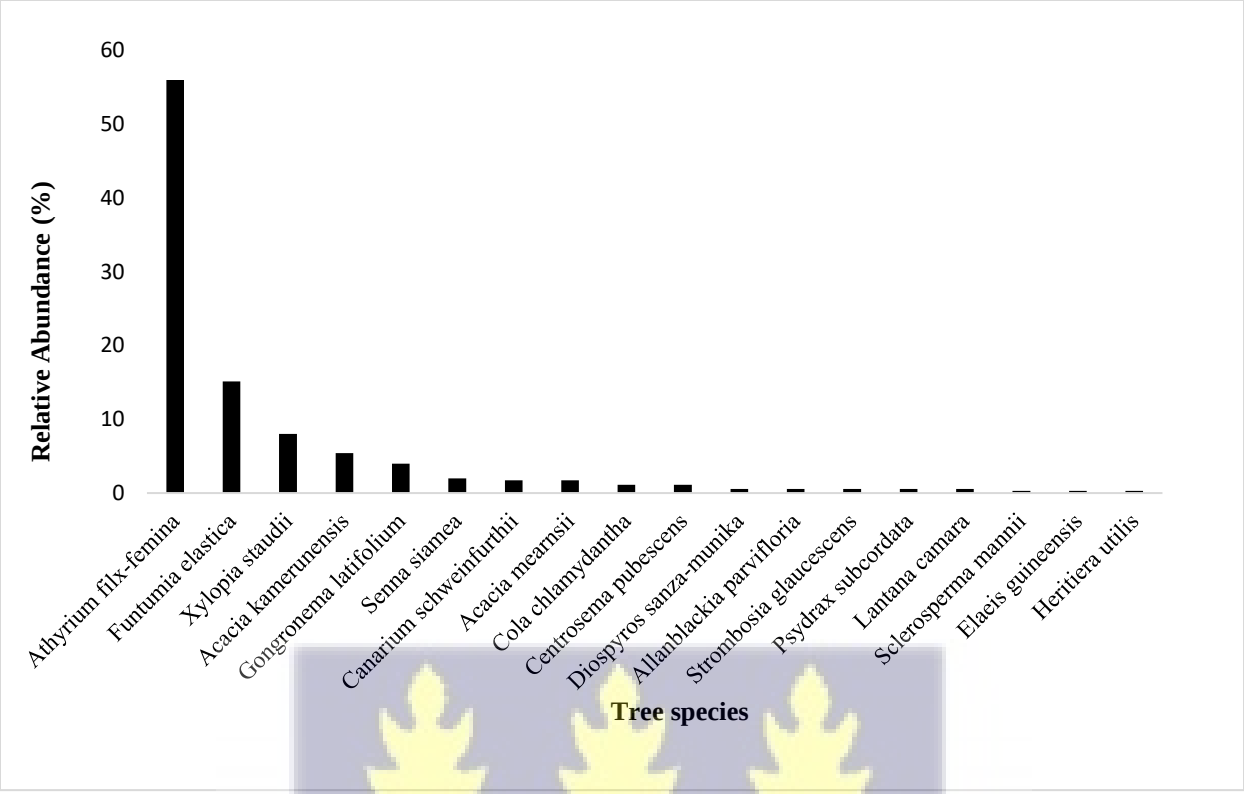


Figure 4.1c. Relative abundance of vegetation at Site D.

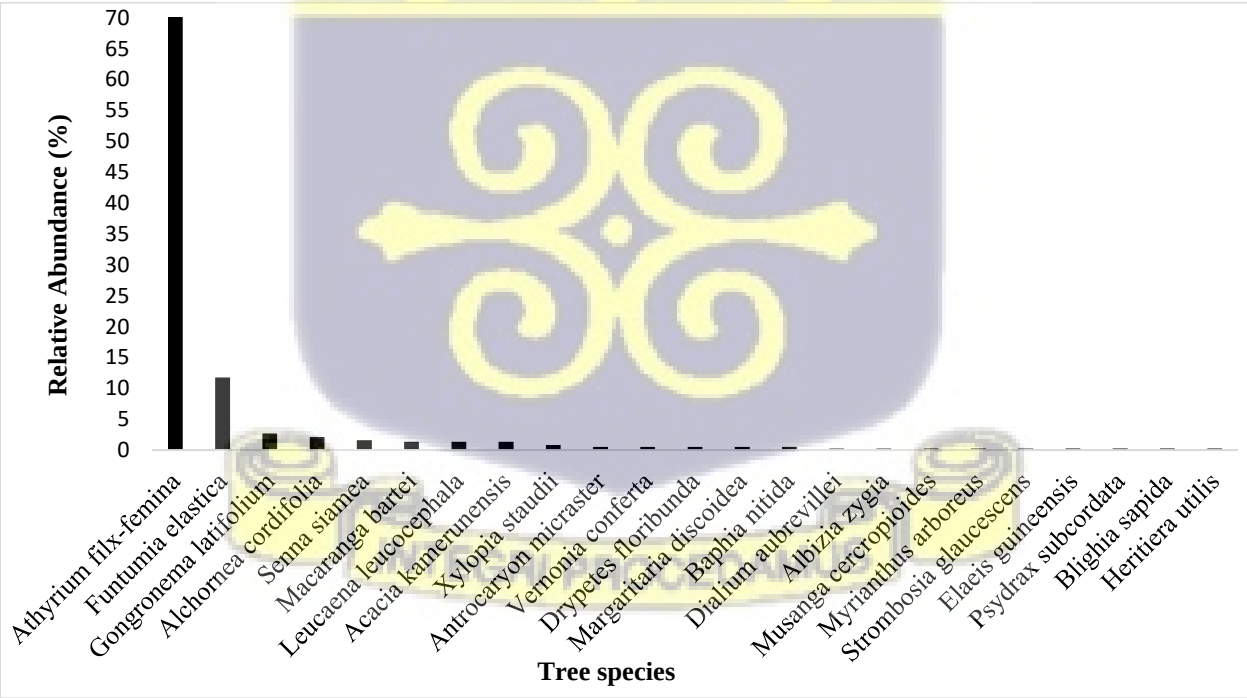


Figure 4.1d. Relative abundance of vegetation at site E-9.

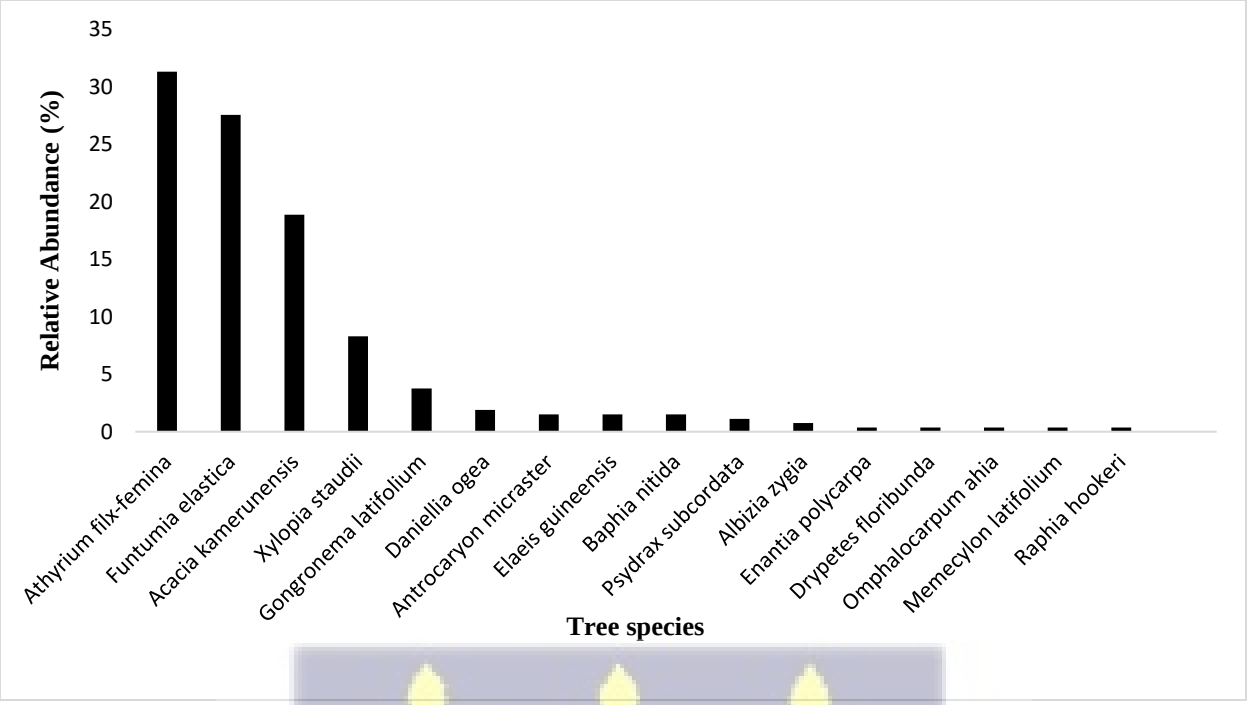


Figure 4.1e. Relative abundance of vegetation at Site E-18.

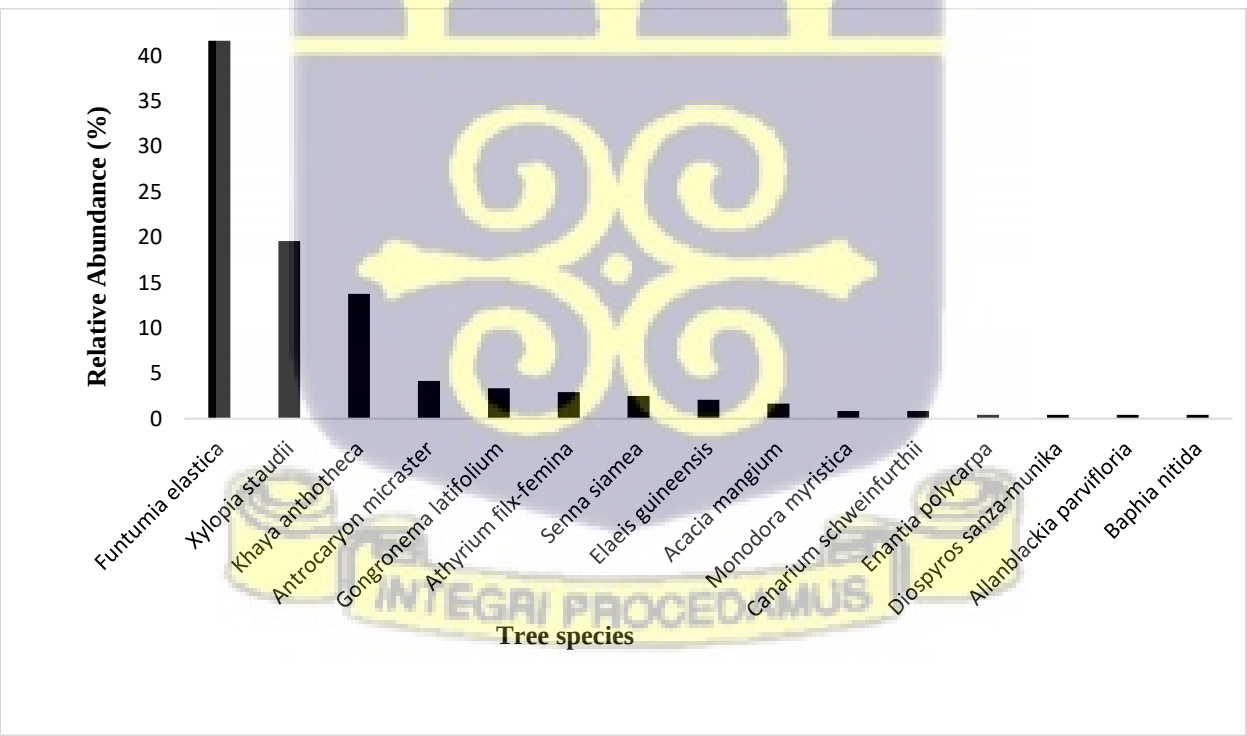


Figure 4.1f. Relative abundance of vegetation at Site E-21.

Figure 4.1g. Relative abundance of vegetation at Site A (NF).

The Margalef's species richness as observed in each site (Figure 4.2) shows that greater number of species occurred in Site A, while the least occurred in Site C. Although Site C recorded 593 individual plants out of the total number of 2225 plants, it was least in richness. Out of the 593 plants reported from Site C, 88.03% was dominated by only *Cynodon dactylon*, and *Athyrium filx-femina*. Site C was a two-plant dominated community.

The Pielous' equitability index was calculated for each site. The value of equitability varied from 0 to 1. It is equal to 1 when all the species have same abundance and tends towards 0 when the near total of flora is concentrated on only one species. The values of this index varied from 0.39 to 0.85 in Sites E-9 and A, respectively (Figure 4.3). Site A had the highest index (0.85) followed by Sites B (0.82), E-18 (0.66), E-21 (0.65), D (0.56), C (0.46) and E-9 (0.39).

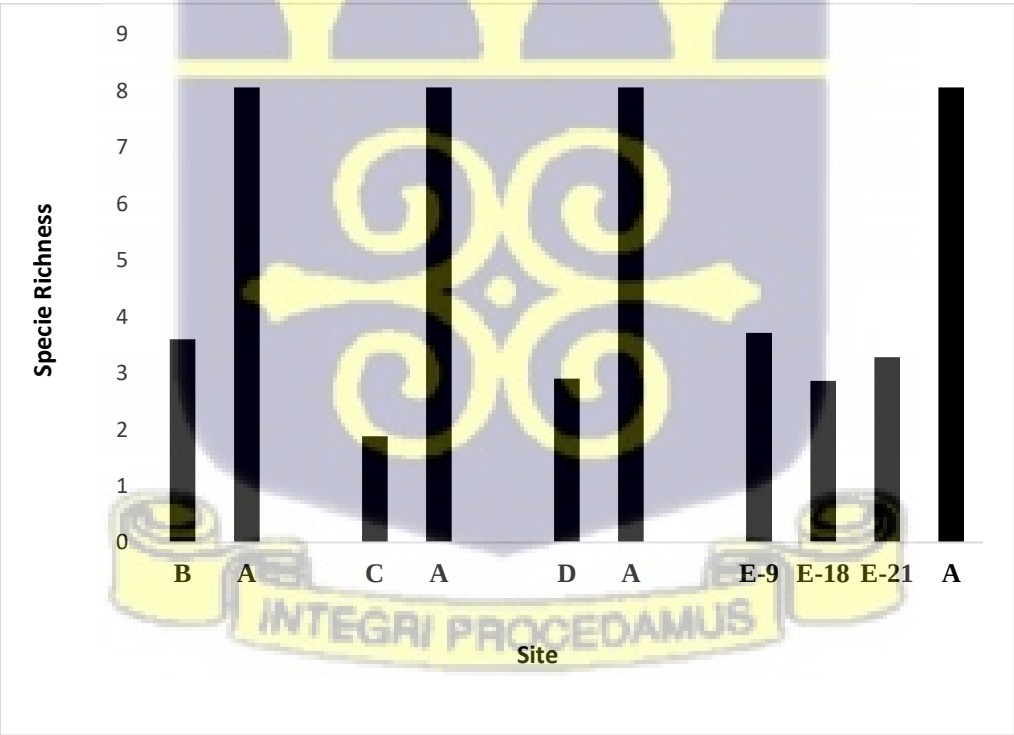


Figure 4.2. Margelef's species richness at the study sites.

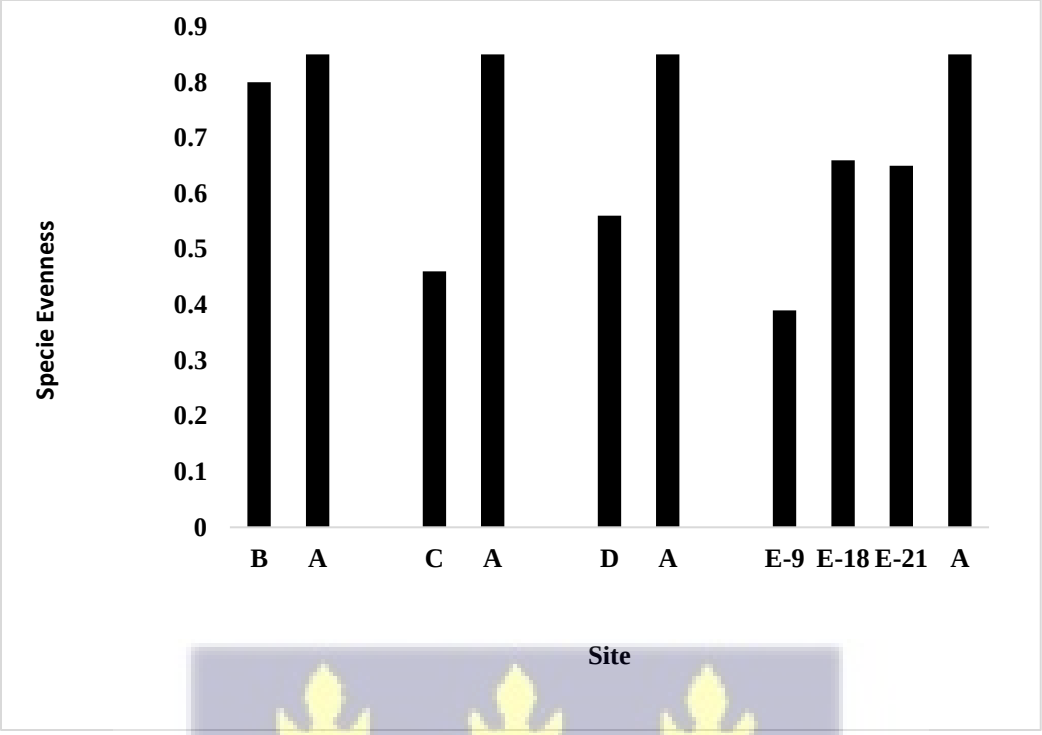


Figure 4.3. Pielous' equitability index (Evenness).

The Shannon diversity index values obtained from this study varied greatly from the natural forest to the rehabilitated sites since the tree diversity of the study sites consisted of several tree species of native and exotic origins and of various life history traits. Within the study area, a low Shannon diversity index value was obtained for Site C (1.19) whereas the highest value was obtained for site A (3.13). Aside from Site B with an index value of 2.64 which came close to Site A (natural forest), the index values for the other rehabilitated sites did not vary greatly from each other. The Shannon-Wiener index of the study area was in the order Sites A (3.13) > B (2.64) > E-18 (1.82) > E-21 (1.76) > D (1.62) > E-9 (1.23) > D (1.19) as shown in (Figure 4.4).

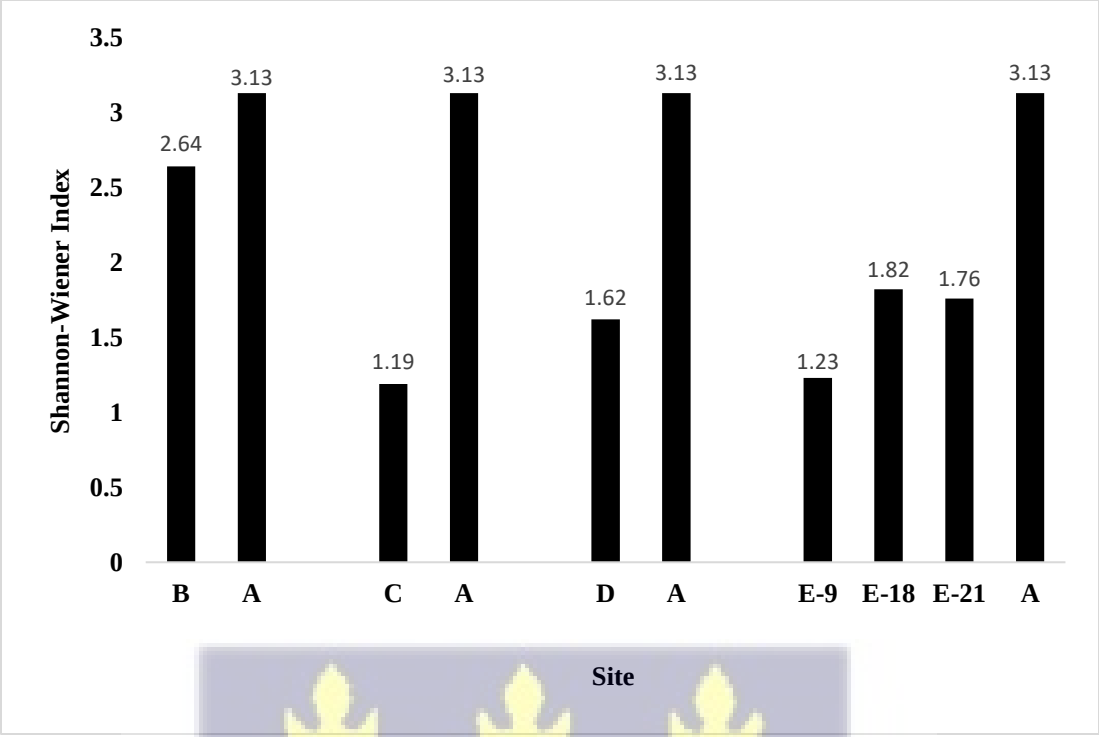


Figure 4.4. Shannon-Wiener diversity index.

4.2.3 Similarity

The Czekanowski similarity coefficient (Csc) gives a very good idea of the presence or absence of species in two different sites. The range of this coefficient is between 0 and 1. Interpretation of the Czekanowski similarity coefficient values is as follows: 1- both sites had only common species; 0 - both sites had only a single species; 0.5 - the two sites had as many common species as the sum of singular species at each site. From the study area the similarity coefficient from 21 combinations (Table 4.3), ranged from 0.002 (0.2%) to 0.739 (73.9%). However, among the rehabilitated sites, the combinations of Site D - Site E-9 showed a relative higher similarity coefficient 0.739 (73.9%) followed by Site D - Site E-19 with an index of 0.619 (61.9%) while Site C – E-9 recorded an index of 0.510 (51.0%).

Table 4.3. Czekanowski similarity coefficient.

Site	B	C	D	E-9	E-18	E-21	A
B	1						
C	0.108	1					
D	0.253	0.441	1				
E-9	0.264	0.510	0.739	1			
E-18	0.134	0.195	0.619	0.474	1		
E-21	0.298	0.027	0.370	0.239	0.482	1	
A	0.121	0.002	0.162	0.115	0.220	0.113	1

4.2.4 Diversity of earthworms

The earthworm population in the study area was generally low. However, there were variations among the seven selected sites where several factors could be used to justify the performance of each site. In all, a total number of 954 earthworms were collected from the study area (Table 4.4). Out of this, 540 earthworms were collected from Site B, 201 from Site E-18 and 129 from site D. The natural forest (Site A) had 30 earthworms which comes after Site C with 43 individual earthworms. Sites E-21 and E-9 produced 5 and 6 earthworms respectively. In terms of functional categories, 475 earthworms were identified as epigeic, 395 were endogeic while 84 were anecic. Whereas Sites B and E-18 had earthworms in all three categories, C, D and A had two categories of earthworms while E-9 and E-21 had only one category.

Table 4. 4. Types and total number of earthworms collected from the study sites.

Earthworm category	Site B	Site C	Site D	Site E			Site A
				E-9	E-18	E-21	Control
Number of epigeic	194	35	96	6	117	5	22
Number of endogeic	304	8	Nil	Nil	75	Nil	8
Number of anecic	42	Nil	33	Nil	9	Nil	Nil
Total no. of earthworms	540	43	129	6	201	5	30

The species richness index as shown in (Table 4.5) has no limit value and shows variation depending upon the number of species. Specie richness of earthworms was highest at Site B with an index value of 2.84 followed by Site E-18 with a close index of 2.81. Sites A, C and D were in the range of 1.71 to 1.79. However, both Sites E-21 and E-9 had 0 index each. The Shannon-Wiener earthworm diversity index of the study area was in order of Site B (0.89) > Site E-18 (0.82) > Site A (0.58) > Site D (0.57) > Site C (0.48). However, the index values for Sites E-21 and E-18 was 0 for each site where only one species of earthworm (epigeic) as shown in (Table 4.5 and 4.4) was found.

Table 4. 5. Diversity indices of earthworms from study sites.

Index	**Site B**	**Site C**	**Site D**	**Site E**			**Site A**
				E-9	**E-18**	**E-21**	**Control**
Density/m^2	115.2	9.17	27.52	1.28	42.88	1.07	6.4
SR*	2.84	1.73	1.79	0	2.81	0	1.71
EQ**	0.3	0.69	0.82	0	0.27	0	0.19
SWI***	0.89	0.48	0.57	0	0.82	0	0.58

SR* = Species richness; EQ** = Equitability; SWI*** = Shannon-Wiener Index

4.3 Discussion

4.3.1 Diversity of plant species

The Shannon diversity index values obtained from this study vary greatly from the natural forest to the rehabilitated sites since the tree diversity of the study sites consists of several tree species of native and exotic origins and of various life history traits. Within the study area, the highest Shannon diversity index value was obtained at Site A (natural forest). However, Site B had an index value of 2.64 which comes close to Site A. The higher Shannon diversity index value of Site

B relative to other rehabilitated sites is attributable to the fact that Site B followed the conventional protocol of reclaiming degraded mined soils. Again, the fertile layer of topsoil which had stored seeds of native plants became the final surface material at Site B during reclamation. This corroborates the works of Bradshaw (1996) who concluded that the success of reclamation program is largely dependent on the fertility of the topsoil and management practices used during reclamation. From the study, Site B, had relatively more organic carbon, stable aggregates, good texture, and more earthworms population compared to the other reclaimed sites. Site C had an index value of 1.19 against 3.13 index value obtained by Site A. The low diversity for Site C was due to the young nature of the site and the fact that the reclamation materials were packed in a haphazard way. Site C was a mixture of tailings, waste rocks and other overburden materials which was highly compacted during the reclamation process. The impoverished nature and higher bulk density of site C, contributed to its low diversity index. The Site D, which is basically sand, had an index value of 1.62 compared to the 3.13 index value of the Site A. The low diversity for Site D is due to the sandy and the impoverished nature of the soil. Among the same mode of reclamation, E-9, E-18, and E-21 had an index value of 1.23, 1.82 and 1.76, respectively. However, the index values for E-18 and E-21 did not vary greatly from each other. The low diversity among these sites, stem from the impoverished nature emanating from the haphazard way of packing the reclamation materials.

This study reveals variations in tree species richness and diversity among the seven selected sites. This could be explained by several reasons; 1) the type of land use or technique of reclamation; 2) degree of disturbances; 3) age of reclamation; and 4) altitude. Nevertheless, the high species richness, evenness and diversity in Site A followed by Site B is a manifestation of their

multifunctional and structural complexities (Agbogidi and Adolor, 2013) underpinned by various biotic and abiotic factors.

4.3.2 Similarity of plant species among the sites

There is a very weak similarity indices between Site A (natural forest) and the six rehabilitated sites (Sites B, C, D, E-9, E-18 and E-21). However, despite close resemblance of species richness among the rehabilitated sites, species composition showed a very low similarity coefficient among sites. Out of 21 combinations, 18 combinations had values of similarity indices lower than 50%. The natural forest (Site A) dominantly consisted of native species with no exotic species, culminating in a very largely dissimilarity coefficient with the rehabilitated sites. This is a subtle indication that the rehabilitated sites were inherently high in tree diversity prior to being severely degraded and that the current species assemblage is partly shaped by its geodiversity. Again, high dissimilarity between the natural forest and the rehabilitated sites may be attributable to the degree of disturbances, reclamation procedures used and post planting maintenance. These factors reflect the different forces influencing species establishment and disappearance in the rehabilitated sites and hence create a heterogeneous landscape among the individual rehabilitated sites and natural forest.

4.3.3 Diversity of earthworms

Except for Site D, earthworm population according to functional category decreased with depth. This exception is due to the fact Site D is basically sandy. Sandy soil heats up and dries out fast at the surface, so earthworms often move deeper into the soil profile. Or seek out spots with more moisture and decaying organic material, like under leaf litter or mulch or near plant roots where organic matter is more available. This accounted for more epigeic and anecic as seen Table 4.4.

The decrease in population with depth is due to the fact that plant litter and organic matter which are their primary source of food are abundant on the soil surface and the first few centimeters in the soil. Again, increasing trend of bulk density with depth as depicted in Table 5.2 restrict their movement down the profile. Bulk density increasing with depth is eminent in reclaimed mine soils since land preparation involves the use of soils poor in organic matter and heavy traction.

High species richness and evenness was observed in Site B (2.84) and Site D (0.82), respectively, which may be due to the vegetation type and pH levels in the sites. Site B had a diverse vegetation which offered multiple niches, food sources, microhabitats and promoted coexistence of species. Blanchart and Julka (1997) also found that flora in a particular area determined the relative abundance of earthworm species. The highest number of earthworm population, species richness and diversity index recorded by the Site B may be highly attributed to the type of reclamation technique used (Back filled approach), the slope (1%), the lower altitude 52 m. a. s. l and low bulk density value and floral composition of the site. These favourable biotic and abiotic variables of the site accounted for high earthworm population found in all three functional categories. Site D on the other hand, had a neutral pH which offered a conducive environment for a wider range of plants and microbial species.

Site E-18 came close to Site B in terms of earthworm population, density, species richness and diversity. It was, however, poor in term of its equitability index. Although these two sites are highly disturbed lands, yet their performance outweighs the undisturbed natural forest. These results are in agreement with the findings of Frazão et al. (2017) who revealed that even under disturbed land conditions, species richness and diversity can be higher than undisturbed land when soil environment is favorable. The natural forest (control site/undisturbed) recorded a significantly low earthworm population, species richness and diversity index. This was in variance to Frazão et

al. (2017) and Schmidt et al. (2003) who reported higher species richness and diversity of earthworms in undisturbed lands than in disturbed lands. Low earthworm density and associated richness and diversity recorded in Site A may be attributed to the extremely acid (3.60) nature and the consequent low microbial activity leading to a pile up of litter on the forest floor (Appendix 3). McCallum et al. (2016) also found that diversity and abundance of earthworms reduce drastically at soil pH less than 4.5.

High litter density impacted negatively on the earthworm density and species richness in Site A (natural forest). This also manifested in the works of Knapp and Seastedt (1986) and Stinner et al. (1984) who found high earthworm density in grasslands than the forest areas with high litter accumulation. Litter from natural forest are long-lived organs with high polyphenolics, high lignin content and low decomposition rate (Feeny, 1970; Shipitalo et al. 1988; Kasurinen et al. 2007) which tend to suppress their recruitment through shade (Berendse, 1998) and creating physical barrier and biochemical influences to soil (Facelli and Pickett, 1991).

Nevertheless, the higher values of earthworm population, density, species richness, diversity indices in Site B, attest to the greater functional biodiversity and the higher potential resilience of the site which are necessary elements in sustaining the ecosystems and in the provisioning of specific ecosystem services (Moonen and Bàrberi, 2008). The extremely low performance of Sites E-9 and E-21 is attributed to their extremely acid nature. Even though they have relatively high soil organic matter and moisture content, their performance was low. Pelosi et al. (2009) in his work, observed that earthworm distribution is influenced by soil pH, moisture content, available organic carbon and soil type.

4.4 Conclusion

Mining and its allied activities led to a considerable reduction in tree species diversity and a change in their composition within the Iduapriem concession. There is high dissimilarity between the natural forest and the rehabilitated sites. Whereas the natural forest was typically of native or local species of trees, the rehabilitated sites were more of exotic species. Great disparity of native species between the rehabilitated sites and the natural forest suggests that the 40% native species and 60% exotic species recommended by EPA-Ghana was not strictly followed or no proper post planting maintenance was carried out. The better botanic performance of Site B relative to other reclaimed sites (C, D and the Es), is clear evidence that Site B, followed the conventional protocol of reclamation. The Site B, performed better than the natural forest and other reclaimed sites in terms of earthworm density and diversity, but on the other hand, came close to Site A (natural forest) in terms of species richness and diversity of vegetation of the study area.

CHAPTER FIVE

PEDOLOGICAL CHARACTERISTICS OF THE REHABILITATED SOILS

5.1 Introduction

Mining activities substantially destroy landforms altering soil horizons, structure, porosity, microflora, and various nutrient cycles which are crucial for maintaining a healthy and productive ecosystem (Sheoran et al. 2010). Mining activities have over the years, degraded arable lands. This will continue, or accelerate, in the future since the demand for mineral resources is increasing due to population growth, and the advancement in science and technology.

Surface gold mining under large-scale system follows this sequence: first, the topsoil, the most organic enriched fraction of the soil profile, is removed and stored until replaced. Subsequently, subsoil horizons are removed and stored. Finally, the bedrock is broken and removed to expose the ore or the gold-bearing rock which then is mined and taken to the mill for processing. The activities leading to the extraction of gold, result in excavation of greater depths of soil, overburden materials, non-economic rocks (waste rocks) and the ore body resulting in creation of huge void (pit) and pile of waste rock dumps on the mine. After the beneficiation of the ore, the milled rocks, herein referred to as tailings is stored in an embankment called tailing storage facility (TSF). The open-pit cut generates variety of wastelands and materials differing in mineralogical, chemical and physical composition which must be reclaimed.

Reclamation of the degraded mined soil involves re-creating the existing landscape, pre-mining soil profile and revegetation of different plant community to suit the new soil. According to the Australian Government (2006), these reconstruction activities are guided by the natural landform and the objective for post-mining land use. Additionally, the nature of mining operation and the

nature and properties of the reclamation materials and their sequence of packing are important determinant of the final landscape and soil profile that is reconstructed.

The dumping sequence that restores the original materials in the original sequence ensures that the layer of fertile topsoil becomes the final surface (Chamber of Mines of South Africa/Coaltech, 2007), which is the conventional protocol prescribed by EPA-Ghana (1994). Conversely, the haphazard way of packing reclamation materials significantly affects the restoration of the physical, chemical and the biological fertility of the soil.

Several authors have reported extensively on the influence of reclamation activities on degraded mine soils (Juwarkar et al. 2010; Mukhopadhyay et al. 2013). However, in AngloGold Ashanti Iduapriem mines, few, if any, research on reclaimed soils have tracked changes in soil properties in the profile (0-100 cm depth), particularly across different modes of reclamation and reclamation duration.

The research question is do reclamation methods affect mine soil properties? Mine soil properties change over time and is influenced by the climatic condition, organisms, parent material and topography. The combined effects of these factors result in horizonation of reconstructed landform. However, anthropogenic activities have profound effects on the horizonation, morphology, physical and chemical attributes of mined soils. The objective of the study is to assess the differences in soil properties between reclaimed gold mined soils relative to the undisturbed soil.

5.2 Results

The results of the field studies including analytical data on selected physical and chemical properties of the rehabilitated soils and the undisturbed soil are presented in the following sections. The profile at each study site (A to E) is referred to as Pedon.

5.2.1 Morphological characteristics

5.2.1.1 Pedon A

The observed depth of Pedon A at the natural forest site is 100 cm. The surface soil (0-10 cm) was dark brown (10YR 3/4) but was bright yellowish brown (10YR 6/8) within 10-50 cm depth and yellow orange (7.5YR 7/8) at 50-100 cm depth (Table 5.1). The soil structure of the surface layer was granular. The subsurface horizons had sub-angular blocky within 10-50 cm and sub-angular blocky with some variations in the size of peds within the lower 50 cm of the profile. The consistency in profile within (0-10 cm) depth was slightly soft, friable and slightly sticky when (dry, moist and wet). Within the depth of (10-30 cm) it was slightly hard, friable and sticky when (dry, moist and wet) and plastic but however, turned hard, friable and slightly sticky when (dry, moist and wet) within (30-50 cm) and very hard, friable and slightly sticky when (dry, moist and wet) in (50-100 cm) depth. The root distribution of the profile was many medium, many fine and many very fine in the surface layer (0-10 cm) to many medium and many very fine in the (10-30 cm), very few fine in (30-50cm) and no root detected within the lower half of the profile. The coarse fragments increased with depth. Coarse fragment at the surface layer (0-10 cm) was less than 2%. In the subsurface horizons it was less than 5% in (10-20 cm), 8% in within 30-50 cm and rose to 15% in 50-80 cm and 25% in in the bottom.

Table 5. 1. Morphological properties of the soil samples from the profiles at the study sites.

Pedon	Depth (cm)	Colour Moist	Structure*	Consistence¶ Dry	Moist	Wet	Roots†	CF‡ (%)
A	0-10	10YR 3/4	gr	ss	fr	ss	mm, mf, mvf	None
(Control)	10-20	10YR 6/8	gr-sbk	sh	fr	s	mm, mvf	< 5
	20-30	10YR 6/8	gr-sbk	sh	fr	s	mm, mvf	< 5
	30-40	10YR 6/8	gr-sbk	h	fi	ss	vff	8
	40-50	10YR 6/8	gr-sbk	h	fi	ss	vff	8
	50-60	7.5YR 7/8	sbk	vh	fi	ss	nd	15
	60-70	7.5YR 7/8	sbk	vh	fi	ss	nd	15
	70-80	7.5YR 7/8	sbk	vh	fi	ss	nd	15
	80-90	7.5YR 7/8	sbk	vh	fi	ss	nd	25
	90-100	7.5YR 7/8	sbk	vh	fi	ss	nd	25
B	0-10	7.5YR 5/6	gr-sbk	sh	fr	ss	mm, mf, mvf	< 5
	10-20	10YR 5/8	gr-sbk	sh	fr	ss	mm, mf	< 5
	20-30	7.5YR 8/1	gr-sbk	sh	fr	ss	mf	8
	30-40	7.5YR 7/8	gr-sbk	sh	fr	ss	nd	8
	40-50	7.5YR 7/8	gr-sbk	sh	fr	ss	nd	8
	50-60	10YR 4/4	sbk	h	fi	ss	nd	8
	60-70	10YR 4/4	sbk	h	fi	ss	nd	8
	70-80	10YR 4/4	sbk	h	fi	ss	nd	15
	80-90	10YR 4/4	sbk	h	fi	ss	nd	15
	90-100	10YR 4/4	sbk	h	fi	ss	nd	15
C	0-10	7.5YR 6/3	sg	l	l	ss	vf	< 45
	10-20	7.5YR 6/3	sg	l	l	ss	vff	< 45
	20-30	7.5YR 6/3	sg	l	l	ss	nd	< 45
	30-40	7.5YR 6/3	sg	l	l	ss	nd	< 45
	40-50	7.5YR 6/3	sg	l	l	ss	nd	< 45
	50-60	7.5YR 6/3	sg	l	l	ss	nd	< 45
	60-70	7.5YR 6/3	sg	l	l	ss	nd	< 45
	70-80	7.5YR 6/3	sg	l	l	ss	nd	< 45
	80-90	7.5YR 6/3	sg	l	l	ss	nd	< 45
	90-100	7.5YR 6/3	sg	l	l	ss	nd	< 45
D	0-10	10YR 5/2	sg	l	l	ss	mm, mf, mvf	None
	10-20	10YR 5/2	sg	l	l	ss	vff	None
	20-30	10YR 5/2	sg	l	l	ss	vff	None
	30-40	10YR 5/2	sg	l	l	ss	nd	None
	40-50	10YR 5/2	sg	l	l	ss	nd	None
	50-60	10YR 5/2	sg	l	l	ss	nd	None
	60-70	10YR 5/2	sg	l	l	ss	nd	None
	70-80	10YR 5/2	sg	l	l	ss	nd	None
	80-90	10YR 5/2	sg	l	l	ss	nd	None
	90-100	10YR 5/2	sg	l	l	ss	nd	None

5.2.1.2 Pedon B

The profile thickness of Pedon B was 100 cm (Table 5.1). Whereas the colour (moist) was brown (7.5YR 5/6) in the (0-10 cm) depth, it turned yellowish brown (10YR 5/8) and light gray (7.5YR 8/1) in 10-20 cm and 20-30 cm respectively. Within 30-50 cm depth, the soil colour was yellow orange (7.5YR7/8) and became brown (10YR 4/4) in the remaining part of the profile. Granular subangular blocky structure was observed within the first 50 cm, however it changed to subangular blocky within the remaining 50 cm depth below. The consistency in profile from the surface layer to the bottom (0-100 cm) was slightly hard, friable and slightly soft when (dry, moist and wet). Many medium, many fine and many very fine roots were found in the (0-10 cm). Whereas it was many medium and many fine in the (10-20 cm) and many fine roots in (20-30cm), no root was detected within the last (30-100 cm). Coarse fragment was 8% within (0-70 cm) and increased to 15% in 70-100 cm depth.

5.2.1.3 Pedon C

The profile thickness of Pedon C was 100 cm. Soil color (moist) was dull brown (7.5YR 6/3) throughout the profile (0-100 cm) (Table 5.1). The structure of the soil within the profile was generally single grained and the consistency remained loose, loose, slightly sticky when (dry, moist and wet) throughout the profile. The roots in the profile were very fine and very few fine in the 0-10 cm and 10-20 cm depth respectively. However, no root was found below 20 cm depth of profile. The coarse fragments content of the soil remained greater than 45% throughout the profile which according to Schoeneberger et al. (2012), is rated as abundant coarse fragments.

5.2.1.4 Pedon D

The profile thickness of Pedon D was 100 cm. Soil color (moist) was grayish yellow brown (10YR5/2) with a single grained structure throughout the profile (0-100 cm) (Table 5.1). The

consistency of the soil remained loose, loose, slightly sticky when (dry, moist and wet) throughout the profile. However, the distribution of roots in the profile were mainly many medium, many fine and many very fine in the 0-10 cm and very few fine within 10-30 cm depth. However, no root was found below 30 cm depth of the profile. No coarse fragments were observed in the profile.

5.2.1.5 Pedon E

5.2.1.5.1 Pedon E-9

The profile thickness of Pedon E-9 was 80 cm. Soil color (moist) was bright reddish brown (5YR5/8) from top to the bottom of the profile (Table 5.1). Whereas the structure was granular in 0-10 cm depth, it turned into subangular blocky structure in the remaining 10-100 cm depth of the profile. In the 0-10 cm depth, the consistency of the soil was slightly hard, friable, and sticky when (dry, moist and wet). It however, turned into very hard, friable and slightly sticky when (dry, moist and wet) within 10-80 cm depth. With regard to distribution of roots in the profile, many medium, many fine roots were observed in 0-10 cm depth while few fine roots were found in 10-20 cm depth. However, no root was found below 20 cm depth of the profile. The coarse fragments observed in the profile were less than 2% in all depth.

5.2.1.5.2 Pedon E-18

The profile thickness of Pedon E-18 was 90 cm. The color (moist) of the soil was consistently dull reddish brown (2.5YR4/4) throughout the profile (Table 5.1). The structure was granular in all depth of the profile. The consistency was soft, very friable, and slightly sticky 0-10 cm when (dry, moist and wet). Within the depth of 0-10 cm depth many medium, very fine roots were observed. Whereas few fine and very fine roots were found in 10-20 cm and 20-30 cm depth respectively,

Table 5. 1. Continued.

Pedon	Depth (cm)	Colour Moist	Structure*	Consistence¶ Dry	Moist	Wet	Roots†	CF‡ (%)
E-9	0-10	5YR 5/8	gr	sh	fr	s	mm, mf	< 2
	10-20	5YR 5/8	sbk	vh	fr	s	ff	< 2
	20-30	5YR 5/8	sbk	vh	fr	ss	nd	< 2
	30-40	5YR 5/8	sbk	vh	fr	ss	nd	< 2
	40-50	5YR 5/8	sbk	vh	fr	ss	nd	< 2
	50-60	5YR 5/8	sbk	vh	fr	ss	nd	< 2
	60-70	5YR 5/8	sbk	vh	fr	ss	nd	< 2
	70-80	10YR 5/2	sbk	vh	fr	ss	nd	< 2
E-18	0-10	2.5YR 4/4	gr	s	vfr	ss	mm, vf	5
	10-20	2.5YR 4/4	gr	s	vfr	ss	ff	5
	20-30	2.5YR 4/4	gr	s	vfr	ss	vf	5
	30-40	2.5YR 4/4	gr	s	vfr	ss	nd	5
	40-50	2.5YR 4/4	gr	s	vfr	ss	nd	5
	50-60	2.5YR 4/4	gr	s	vfr	ss	nd	5
	60-70	2.5YR 4/4	gr	s	vfr	ss	nd	5
	70-80	2.5YR 4/4	gr	s	vfr	ss	nd	5
	80-90	2.5YR 4/4	gr	s	vfr	ss	nd	5
E-21	0-10	2.5YR 6/8	sbk	h	fr	s	ff	2
	10-20	2.5YR 6/8	sbk	h	fr	s	vff	2
	20-30	2.5YR 6/8	sbk	h	fr	s	nd	2
	30-40	2.5YR 6/8	sbk	h	fr	s	nd	2
	40-50	2.5YR 6/8	sbk	h	fr	s	nd	2
	50-60	2.5YR 6/8	sbk	h	fr	s	nd	2
	60-70	2.5YR 6/8	sbk	h	fr	s	nd	2
	70-80	2.5YR 6/8	sbk	h	fr	s	nd	2
	80-90	2.5YR 6/8	sbk	h	fr	s	nd	2
	90-100	2.5YR 6/8	sbk	h	fr	s	nd	2

***** gr = granular; sbk = subangular blocky; sg = single grained.
¶ s = soft; ss = slightly soft; h = hard; sh = slightly hard; vh = very hard; l=loose; fr = friable; vfr = very friable; fi = firm; s = sticky; ss = slightly sticky.
† mm = many medium, mf = many fine, ff = few fine, vff = very few fine; vf =very fine; mvf = many very fine; nd = not detected.
‡ cf = coarse fragments; 0% = none; < 2% = very few coarse fragments; 2% = few coarse fragments; < 5% = few coarse fragments; 5% = common coarse fragments; 8% = common coarse fragments; 15% = many coarse fragments; 25% = many coarse fragments; < 45% = abundant coarse fragments.

no root was found below 20 cm depth of the profile. The coarse fragments observed in the profile were 5% consistently in all depth of the profile.

5.2.1.5.3 Pedon E-21

The profile thickness of Pedon E-21 was 100 cm. Soil color (moist) was bright brown (2.5YR6/8) in the entire profile (Table 5.1). Soil structure was sub-angular blocky in all depth of the profile with some variations in size of peds. The consistency of the soil was hard, friable, and sticky when (dry, moist and wet) in the entire profile. Except for 0-10 cm and 10-20 cm depth that had few fine roots and very few fine roots respectively, the remaining depth (20-100 cm) had no roots in the profile. The observed coarse fragments in the profile were 2% in all depth.

5.2.2 Physical properties

5.2.2.1 Pedon A

Results on the particle-size distribution of the Pedon shows that the sand content increased from 43% in the 0-10 cm to 44.5% in the 10-20 cm and then steadily declined along the profile (Table 5.2). The clay content slightly decreased from the surface layer (0-10 cm) to (10-20 cm) and then increased consistently with depth. The texture of the soil was dominated by clay loam in top soil but turned into clay in the horizons below 20 cm depth. The silt/clay ratio decreased with depth of profile. The bulk density of the studied profile, ranged from (1.20 Mg m^{-3}) in the surface layer (0-10 cm) to (1.98 Mg m^{-3}) in the bottom layer (90-100 cm) of the profile. Bulk density increased consistently with depth. The aggregate stability at the plough layer reduced from 4.16 mm in the 0-10 cm to 3.68 mm in the 10-20 cm depth. The moisture content at FC (0.33 bar) and at PWP (15 bars) had shown a consistent increase with increasing depth. Within the profile, the highest FC (43.2%) and PWP (31.3%) were recorded at the bottom part of the profile that contained higher

clay content, regardless of its high bulk density. However, the available water capacity generally same within the 100 cm depth.

5.2.2.2 Pedon B

The particle size distribution was more of the sand fractions while the textural classes were mainly sandy loam in (0-30 cm) and sandy clay loam in the 30-100 cm depth (Table 5.2). Whereas the sand content decreased with depth, the clay content was the reverse. The silt/clay ratio increased from 0.14 in 0-10 cm to 0.71 in 20-30 cm and afterward declined along the profile. The bulk density ranged from 1.45 Mg m^{-3} in the surface layer (0-10 cm) to 2.26 Mg m^{-3} in the bottom layer (90-100 cm) of the profile. The aggregate stability was 2.57 mm in the 0-10 cm and decreased to 2.19 mm in the 10-20 cm at the plough layer. The moisture content at FC (0.33 bar) and at PWP (15 bars) increased with increasing depth. The moisture content at FC (0.33 bar) had increased from 19.2% in 0-10 cm to 28.3% in the 90-100 cm. That at PWP (15 bars) increased from 12.6% at surface layer to 19.5% at the bottom of the profile. Correspondingly, the available water capacity increased from 6.6 % in the surface layer to 8.8% in bottom layer of the profile.

5.2.2.3 Pedon C

The sand content of the profile ranged from 67.5 to 87.5% with the highest found at the surface. The clay content was low, relative to silt. However, the lowest silt content (2%) was observed at the surface (Table 5.2). Except for the surface layer (0-10 cm) that was loamy sand textured, the remaining portion of the profile was entirely sandy loam. The silt/clay ratio was very low within the upper 10 cm but increased inconsistently with depth. Soil bulk density was greater than 2 Mg m^{-3} in the entire profile. Aggregate stability of the soil observed at the plough layer was very low (0.67 mm). The moisture content at FC (0.33 bar) and at PWP (15 bars) and the available water capacity inconsistently decreased with increasing depth.

Table 5. 2. Selected physical properties of the soil profiles.

Pedon	Depth	P S D‡ (%)			Texture‖	Silt/Clay	BD‡	MWD‡	Moisture content‡ (%)		
	(cm)	Sand	Silt	Clay			Mgm^{-3}	(mm)	1/3 bar	15 bars	AWC
A	**0-10**	43.0	22.0	35.0	CL	0.63	1.20	4.16	34.3	22.0	12.3
(control)	**10-20**	44.5	25.0	30.1	CL	0.83	1.33	3.68	31.6	19.2	12.4
	20-30	23.3	31.0	45.5	C	0.68	1.51	nd	40.6	27.6	13.0
	30-40	22.5	30.0	47.5	C	0.63	1.51	nd	41.3	28.8	12.5
	40-50	21.8	27.9	50.1	C	0.56	1.64	nd	42.1	29.7	12.4
	50-60	21.3	27.7	51.7	C	0.54	1.72	nd	42.5	30.2	12.3
	60-70	21.2	27.7	51.0	C	0.54	1.70	nd	42.6	30.2	12.4
	70-80	20.5	27.2	52.1	C	0.52	1.76	nd	42.9	30.8	12.1
	80-90	20.0	27.5	52.5	C	0.52	1.82	nd	43.2	31.3	11.9
	90-100	20.0	27.5	52.5	C	0.52	1.98	nd	43.2	31.3	11.9
	Mean	**25.8**	**27.4**	**46.8**	**C**	**0.60**	**1.6**	**3.92**	**40.4**	**28.1**	**12.3**
B	**0-10**	80.0	2.5	17.5	SL	0.14	1.45	2.57	19.2	12.6	6.6
	10-20	75.0	7.5	17.5	SL	0.43	1.80	2.19	20.1	12.6	7.5
	20-30	70.0	12.5	17.5	SL	0.71	2.06	nd	21.1	12.6	8.5
	30-40	70.0	9.0	21.0	SCL	0.43	2.01	nd	22.7	14.3	8.4
	40-50	70.0	9.0	21.0	SCL	0.43	2.16	nd	22.7	14.3	8.4
	50-60	67.5	2.5	30.0	SCL	0.08	2.24	nd	28.3	19.5	8.8
	60-70	65.0	2.5	32.5	SCL	0.08	2.14	nd	30.5	21.2	9.3
	70-80	67.5	2.5	30.0	SCL	0.08	2.18	nd	28.3	19.5	8.8
	80-90	67.5	2.5	30.0	SCL	0.08	2.08	nd	28.3	19.5	8.8
	90-100	67.5	2.5	30.0	SCL	0.08	2.26	nd	28.3	19.5	8.8
	Mean	**70.0**	**5.3**	**24.7**	**SCL**	**0.26**	**2.0**	**2.38**	**25.0**	**16.6**	**8.4**
C	**0-10**	87.5	2.5	10.0	LS	0.25	2.00	0.67	13.0	8.0	5.0
	10-20	75.0	10.0	15.0	SL	0.67	2.18	0.67	18.4	10.9	7.5
	20-30	67.5	20.0	12.5	SL	1.60	2.19	nd	18.8	9.8	9.0
	30-40	72.5	17.5	10.0	SL	1.75	2.15	nd	16.1	8.0	8.1
	40-50	75.0	15.0	10.0	SL	1.50	1.98	nd	15.7	8.0	7.7
	50-60	72.5	15.0	12.5	SL	1.20	2.05	nd	17.7	9.8	7.9
	60-70	75.0	15.0	10.0	SL	1.50	2.29	nd	15.7	8.0	7.7
	70-80	70.0	15.0	15.0	SL	1.00	2.01	nd	16.8	8.1	8.7
	80-90	70.0	15.0	15.0	SL	1.00	2.13	nd	16.8	8.1	8.7
	90-100	75.0	12.5	12.5	SL	1.00	2.05	nd	17.3	9.8	7.5
	Mean	**74.0**	**13.8**	**12.2**	**SL**	**1.15**	**2.10**	**0.67**	**16.6**	**8.9**	**7.8**
D	**0-10**	82.5	2.5	15.0	SL	0.17	1.48	0.44	16.8	10.9	5.9
	10-20	77.5	7.5	15.0	SL	0.50	1.52	0.42	17.8	10.9	6.9
	20-30	82.0	5.0	13.0	SL	0.38	1.57	nd	17.0	10.9	6.1
	30-40	85.0	5.0	10.0	LS	0.50	1.60	nd	13.6	8.0	5.6
	40-50	80.0	7.5	12.5	SL	0.60	1.64	nd	14.6	8.0	6.6
	50-60	80.0	7.5	12.5	SL	0.60	1.66	nd	14.6	8.0	6.6
	60-70	80.0	7.5	12.5	SL	0.60	1.76	nd	14.6	8.0	6.6
	70-80	82.5	7.5	10.0	LS	0.75	1.78	nd	14.0	8.0	6.0
	80-90	82.5	5.0	12.5	LS	0.40	1.81	nd	15.7	9.7	6.0
	90-100	82.5	7.5	10.0	LS	0.75	1.89	nd	14.0	8.0	6.0
	Mean	**81.5**	**6.3**	**12.2**	**SL**	**0.53**	**1.7**	**0.43**	**15.3**	**9.0**	**6.2**

5.2.2.4 Pedon D

The soil contained large quantity of sand, ranging from 77.5-85.0% followed by 10-15% clay content with silt showing the lowest content with the range 2.5-7.5 within the profile (Table 5.2). The texture was and intermixed of sandy loam and loamy sand within the profile. The silt/clay ratio was low and fell within the range of 0.17-0.75. Bulk density was high and increased with depth from 1.48 Mg m^{-3} in the surface layer to 1.89 Mg m^{-3} in the bottom of the profile. The aggregate stability was determined within the plough layer, and it was 0.44 mm at 0-10 cm but marginally dropped to 0.42 mm in 10-20 cm depth. Results on moisture content at FC (0.33 bar) showed an inconsistent decrease with depth while a general decrease in moisture at PWP (15 bars) with depth was recorded within the profile. However, the available water capacity at various depth increased inconsistently with depth.

5.2.2.5 Pedon E

5.2.2.5.1 Pedon E-9

The texture of Pedon E-9 was dominated by high clay content followed by silt and then sand (Table 5.2). At the surface, the clay content was 37.5% but generally increased with depth. Conversely, sand and silt contents decreased with increasing depth. The texture was clay loam at the surface but sharply changed to clay throughout the depth below 10 cm in the profile. The silt/clay ratio was 0.87 at the surface but generally decreased down the profile. Whereas the bulk density ranged between 1.23 to 1.59 Mg m^{-3}, it increased with depth up to 60 cm beyond which it reduced. Within the plough layer, the aggregate stability was 2.42 within 0-10 cm depth but sharply increased to 2.94 in 10-20 cm depth. Whereas the moisture at FC (43.2%) and PWP (31.3%) increased with depth, the available water capacity decreased with depth.

Table 5. 2. Continued.

Pedon	Depth	P S D‡ (%)			Texture⁋	Silt/Clay	BD‡	MWD‡	Moisture content‡(%)		
	(cm)	Sand	Silt	Clay			Mgm^{-3}	(mm)	1/3 bar	15 bars	AWC
E-9	**0-10**	30.0	32.5	37.5	CL	0.87	1.23	2.42	35.0	20.7	14.3
	10-20	17.5	32.5	50.0	C	0.65	1.42	2.94	42.2	29.6	12.6
	20-30	15.0	27.5	57.5	C	0.48	1.43	nd	44.8	33.8	11.0
	30-40	22.5	25.0	52.5	C	0.48	1.47	nd	43.2	31.3	11.9
	40-50	20.0	25.0	55.0	C	0.45	1.59	nd	43.9	32.3	11.6
	50-60	17.5	22.5	60.0	C	0.38	1.59	nd	45.6	34.9	10.7
	60-70	20.0	17.5	62.5	C	0.28	1.38	nd	45.7	35.0	10.7
	70-80	17.5	22.5	60.0	C	0.38	1.40	nd	45.6	34.9	10.7
	Mean	**20.0**	**25.7**	**54.3**	**C**	**0.50**	**1.4**	**2.68**	**43.3**	**31.6**	**11.7**
E-18	**0-10**	40.0	25.0	35.0	CL	0.71	1.52	2.45	33.4	20.7	12.7
	10-20	32.0	29.0	39.0	CL	0.74	1.78	2.45	36.5	23.2	13.3
	20-30	20.0	37.0	43.0	C	0.86	1.78	nd	39.2	25.5	13.7
	30-40	22.0	35.5	42.5	C	0.84	1.79	nd	38.0	25.0	13.0
	40-50	22.5	35.0	42.5	C	0.82	1.80	nd	38.2	24.0	14.2
	50-60	23.0	35.5	41.5	C	0.86	1.83	nd	38.6	24.0	14.6
	60-70	22.5	34.5	43.0	C	0.80	1.88	nd	38.7	25.0	13.7
	70-80	21.5	36.0	42.5	C	0.85	1.78	nd	39.0	24.8	14.2
	80-90	22.5	36.0	41.5	C	0.87	1.75	nd	38.7	25.2	13.5
	Mean	**25.1**	**33.7**	**41.2**	**C**	**0.82**	**1.8**	**2.45**	**37.8**	**24.2**	**13.7**
E-21	**0-10**	32.5	37.5	30.0	CL	1.25	1.30	3.45	33.4	19.1	14.3
	10-20	25.0	37.5	37.5	CL	1.00	1.37	3.96	37.5	23.3	14.2
	20-30	27.5	37.5	35.0	CL	1.07	1.37	nd	36.0	21.8	14.2
	30-40	27.5	37.5	35.0	CL	1.07	1.43	nd	36.0	21.8	14.2
	40-50	22.5	35.0	42.5	C	0.82	1.46	nd	39.5	26.0	13.5
	50-60	22.5	35.0	42.5	C	0.82	1.47	nd	39.5	26.0	13.5
	60-70	22.5	33.5	43.5	C	0.77	1.49	nd	35.9	20.6	15.3
	70-80	17.0	42.0	41.0	SiC	1.02	1.58	nd	39.3	24.8	14.5
	80-90	17.0	42.0	41.0	SiC	1.02	1.53	nd	39.3	24.8	14.5
	90-100	18.0	42.0	40.0	SiC	1.05	1.44	nd	36.3	24.8	11.5
	Mean	**23.2**	**38.0**	**38.8**	**CL**	**0.99**	**1.4**	**3.705**	**37.3**	**23.3**	**14.0**

‡ PSD = Particle Size Distribution; BD = Bulk density; AWC = Available water capacity; MWD = Mean weight diameter; nd = not determined.
⁋ CL = Clay Loam; C = Clay; SL = Sandy Loam; SCL = Sandy Clay Loam; L = Loam; SL = Sandy Loam; LS = Loamy Sand; SiC = Silty Clay.

5.2.2.5.2 Pedon E-18

The mean particle size distribution in the profile was 25.1%, 33.7% and 41.2% for sand, silt and clay, respectively (Table 5.2). However, sand was dominant (40.0%) in the surface but decreased consistently to 22.5% at the bottom. Silt was 25% at the surface but increased down the profile. The clay content, on the other hand, was 35% at the surface but increased inconsistently down the profile. The upper 20 cm was clay loam textured, while the remaining depth below were occupied by a clay textured soil.

The silt/clay ratio generally increased with depth. Bulk density was high and ranged from 1.52 to 1.88 Mg m^{-3} in the profile and increased with depth. Aggregate stability at plough layer increased from 2.45 mm within the plough layer. Whereas the moisture content at FC (0.33 bar) decreased inconsistently with depth, moisture at PWP (15 bars) and available water capacity generally increased with depth of profile.

5.2.2.5.3 Pedon E-21

Results on the particle size distribution show the following order clay >silt >sand with respective mean values of 38.8%, 38.0% and 23.2% (Table 5.2). Whereas the silt and clay content in the profile increased with depth, sand however, decreased with depth. The texture of soil in the profile was clay loam in the upper portion and clay in the middle portion followed by silty clay in the bottom portion.

The silt/clay ratio generally decreased along the profile depth. Generally, bulk density in the profile was high and increased with profile depth. The aggregate stability of the soil at the plough layer was high and ranged from 3.45-3.96 mm. The moisture content at FC (0.33 bar) and at PWP (15 bars) and the available water capacity showed irregular trends within the profile.

5.2.3 Chemical properties

The analytical data of the selected chemical properties of various pedons are presented in the following sections.

5.2.3.1. Pedon A

Soil pH is the most important chemical characteristic of soil solution. The pH (H_2O) in the profile ranged between 3.5 and 3.9 in the upper 40 cm and increased to 4.1 in 40-100 cm depth. Within the profile, pH in $CaCl_2$ ranged from 3.2 to 3.6 (Table 5.3). Soil pH in the profile increased with depth. The organic carbon (OC) and total nitrogen (TN) contents of the forest soil were high in the topsoil but decreased consistently with depth. Whereas the organic carbon content ranged from 0.30 to 4.12%, the total nitrogen ranged from 0.09 to 0.44% with mean values of 1.32% and 0.19%, respectively. The C/N ratios of the profile ranged between 9.4 in the surface horizon to 3.8 at the bottom. The concentration of available P was very low (4.11 mg kg^{-1}) in the surface layer (0-10 cm) and generally decreased down the profile to 2.94 mg kg^{-1}. The total phosphorus on the other hand, showed irregular pattern within the profile but was generally low.

The mean values for the exchangeable bases extracted from the forest soil occurred in the following order: Ca > Mg > K > Na in the profile (Table 5.4). All the observed values were relatively high in the surface soil but decreased continuously with depth. Exchangeable Ca levels decreased consistently with depth and ranged between 0.82 and 0.97 $cmol_c\ kg^{-1}$ in the profile. According to the ratings by FAO (2006), exchangeable Ca levels in the profile were very low. Exchangeable Mg levels ranged between 0.39 and 0.75 $cmol_c\ kg^{-1}$ and were low as suggested by FAO (2006), indicating Mg insufficient for the natural forest soil. Exchangeable sodium levels ranged between 0.22 and 0.40 $cmol_c\ kg^{-1}$ in the profile and were rated as low to medium (FAO, 2006). The values of Na in the studied profile were less than the critical limit (1 $cmol_c\ kg^{-1}$) for

sodium to be detrimental to plants. Exchangeable K ranged between 0.25 and 0.60 $cmol_c\ kg^{-1}$ in the profile. According to the ratings by FAO (2006), K levels were rated as very low (< 0.2 $cmol_c\ kg^{-1}$), low (0.2 to 0.3 $cmol_c\ kg^{-1}$), medium (0.3 to 0.6 $cmol_c\ kg^{-1}$), high (0.6 to 1.2 $cmol_c\ kg^{-1}$) and very high (>1.2 $cmol_c\ kg^{-1}$). As a result, the forest soil had medium K in the upper 70 cm, while the lower 30 cm depth had low levels of K. The topsoil of the studied profile had relatively high values of exchangeable bases indicating accumulation of the nutrient in the topsoil. Exchangeable acidity was high throughout the profile and decreased consistently down the profile. The exchangeable acidity within the profile ranged between 3.0-4.5 $cmol_c\ kg^{-1}$. The cation exchange capacity in the topsoil was 14.6 $cmol_c\ kg^{-1}$ and decreased to 10.1 $cmol_c\ kg^{-1}$ in the profile. FAO, (2006), categorized CEC < 6 as very low, 6-12 $cmol_c\ kg^{-1}$ as low, 12 - 25 $cmol_c\ kg^{-1}$ as medium, 25-40 $cmol_c\ kg^{-1}$ as high and > 40 $cmol_c\ kg^{-1}$ as very high. These values suggest that CEC in the profile was low to medium. Moreover, the CEC of soil profile followed the trend exhibited by the exchangeable basic cations, reflecting that fewer of those sites are occupied by useful nutrients, and more are likely occupied by exchangeable acidity (H^+ and Al^{3+}).

Base saturation varied within the profile with no clear pattern of values occurring in a range between 14-20%. Landon (1991) had classified BS < 20% as low, 20-60 as medium, and above 60% as high and considered as fertile soil. Hence the soil from the natural forest had low percent base saturation.

5.2.3.2 Pedon B

The pH (H_2O) of Pedon B was generally low (4.5 to 5.3) with mean value of 4.9 and consistently increased down the profile. That of $CaCl_2$ was also low ranging from 3.8 to 4.6 with mean value of 4.2 (Table 5.3). The total nitrogen content was 0.18 at the surface (0-10 cm) but dropped to 0.07 in the middle of the profile and further increased to 0.12% in the bottom of the profile.

Table 5. 3. Soil reaction, organic carbon and total N contents of the profile.

Pedon	Depth	pH*		Total N†	C_{org}†	C/N†	Av. P†	Total P†
	(cm)	H_2O	$CaCl_2$	(%)	(%)		mg kg^{-1}	
A	**0-10**	3.8	3.2	0.44	4.12	9.4	4.11	320.08
(control)	**10-20**	3.6	3.4	0.34	2.94	8.8	3.45	267.54
	20-30	3.5	3.3	0.28	1.97	7.0	3.52	262.34
	30-40	3.9	3.6	0.15	0.92	6.1	3.65	250.12
	40-50	4.1	3.5	0.15	0.82	5.6	3.55	252.43
	50-60	4.1	3.6	0.13	0.80	5.9	3.09	272.56
	60-70	4.1	3.6	0.13	0.69	5.1	3.00	285.00
	70-80	4.1	3.5	0.09	0.31	3.4	3.09	285.64
	80-90	4.1	3.6	0.09	0.30	3.3	3.12	279.94
	90-100	4.1	3.6	0.09	0.30	3.3	2.94	294.98
	Mean	**4.0**	**3.5**	**0.19**	**1.32**	**5.8**	**2.85**	**277.06**
B	**0-10**	5.1	4.6	0.18	2.41	7.8	5.50	198.24
	10-20	4.6	4.0	0.17	1.25	7.4	3.98	148.68
	20-30	4.8	4.4	0.10	0.70	7.0	3.71	153.76
	30-40	4.7	3.9	0.09	0.44	4.9	3.71	155.24
	40-50	4.5	3.8	0.08	0.30	3.8	3.67	164.48
	50-60	4.9	4.1	0.07	0.39	5.6	3.35	114.96
	60-70	4.9	4.2	0.11	0.76	6.9	4.11	102.32
	80-90	5.1	4.5	0.10	0.74	7.4	4.97	109.32
	80-90	5.1	4.3	0.12	0.76	6.3	4.12	104.24
	90-100	5.3	4.5	0.12	0.79	6.6	4.05	103.72
	Mean	**4.9**	**4.2**	**0.11**	**0.81**	**6.4**	**4.12**	**135.50**
C	**0-10**	6.9	6.6	0.05	0.18	3.6	15.53	154.16
	10-20	6.7	6.3	nd	0.10	nd	14.06	119.76
	20-30	6.5	6.2	nd	0.06	nd	13.93	135.08
	30-40	6.3	6.2	nd	0.07	nd	14.23	138.66
	40-50	6.4	6.2	nd	0.05	nd	13.79	129.56
	50-60	6.3	6.1	nd	0.06	nd	13.17	133.96
	60-70	6.4	6.3	nd	0.05	nd	13.15	140.80
	70-80	6.4	6.1	nd	0.03	nd	10.08	135.16
	80-90	6.8	6.5	nd	0.03	nd	10.56	140.24
	90-100	6.5	6.1	nd	0.03	nd	11.11	146.08
	Mean	**6.5**	**6.2**	**0.05**	**0.07**	**3.6**	**12.96**	**137.35**
D	**0-10**	6.5	5.6	0.06	0.46	7.7	16.04	162.24
	10-20	6.4	5.3	nd	0.41	nd	15.99	148.80
	20-30	6.4	5.4	nd	0.06	nd	12.45	135.24
	30-40	6.5	5.8	nd	0.07	nd	11.03	123.84
	40-50	6.4	5.7	nd	0.05	nd	10.88	132.52
	50-60	6.4	5.7	nd	0.05	nd	15.68	140.34
	60-70	6.3	5.6	nd	0.06	nd	15.69	152.92
	70-80	6.1	5.7	nd	0.05	nd	16.00	160.62
	80-90	6.2	6.1	nd	0.06	nd	15.23	146.34
	90-100	6.2	5.9	nd	0.05	nd	11.92	158.60
	Mean	**6.3**	**5.7**	**0.06**	**0.13**	**7.7**	**14.09**	**146.15**

Table 5. 4. Continued.

Pedon	Depth	pH*		Total N†	C_{org}†	C/N†	Av. P†	Total P†
	(cm)	H_2O	$CaCl_2$	(%)	(%)		(mg kg^{-1})	
E-9	**0-10**	4.2	3.7	0.08	0.66	8.3	2.73	215.00
	10-20	4.3	3.7	0.06	0.33	5.5	2.70	209.06
	20-30	4.3	3.8	0.07	0.35	5.0	2.69	209.56
	30-40	4.3	3.7	0.08	0.32	4.0	2.55	204.80
	40-50	4.4	3.7	0.08	0.33	4.1	2.59	208.80
	50-60	4.4	3.8	0.08	0.34	4.3	2.46	204.04
	60-70	4.1	3.7	0.08	0.33	4.1	2.12	209.16
	70-80	4.2	3.8	0.06	0.22	3.7	2.32	207.24
	Mean	**4.3**	**3.7**	**0.07**	**0.36**	**4.9**	**2.52**	**208.46**
E-18	**0-10**	3.4	3.1	0.06	0.55	9.2	5.53	287.86
	10-20	3.9	3.6	0.04	0.21	5.3	5.57	284.62
	20-30	3.9	3.5	0.04	0.21	5.3	4.46	283.92
	30-40	3.9	3.6	0.03	0.09	3.0	3.61	281.10
	40-50	3.9	3.6	0.02	0.10	5.0	3.62	291.60
	50-60	3.9	3.7	0.02	0.11	5.5	3.45	269.04
	60-70	3.8	3.7	nd	0.08	nd	3.12	268.48
	70-80	3.8	3.7	nd	0.09	nd	3.10	291.52
	80-90	3.9	3.7	nd	0.09	nd	3.11	221.36
	Mean	**3.8**	**3.6**	**0.04**	**0.17**	**5.5**	**3.95**	**275.50**
E-21	**0-10**	4.0	3.4	0.10	2.02	19.2	3.13	362.80
	10-20	4.0	3.7	0.03	0.42	14.0	2.54	356.52
	20-30	3.8	3.7	nd	0.39	nd	2.36	338.28
	30-40	4.0	3.7	nd	0.33	nd	2.45	236.06
	40-50	3.9	3.7	nd	0.22	nd	2.27	265.16
	50-60	3.9	3.6	nd	0.15	nd	2.45	291.54
	60-70	3.9	3.7	nd	0.18	nd	2.57	295.40
	70-80	4.0	3.7	nd	0.11	nd	2.56	240.38
	80-90	4.0	3.7	nd	0.18	nd	2.55	299.40
	90-100	3.9	3.7	nd	0.18	nd	2.46	251.14
	Mean	**3.9**	**3.6**	**0.07**	**0.42**	**16.6**	**2.53**	**293.67**

* pH in H_2O and $CaC1_2$ was in a 1:2 soil to liquid suspension; † Total N = Total nitrogen.
C_{org} = Organic carbon; OM = Organic matter; nd = non-detectable;
Av. P = Available phosphorus; Total P = Total phosphorus.

According to the ratings of Blakemore, et al. (1987), the observed total N at the surface was low, then dropped to a very low content in the middle and later appreciated to a low content within the profile. The organic carbon content was high in the surface and decreased to the 50-60 cm and further increased down the profile. The C/N ratios of the profile showed irregular pattern from top to the bottom. The available P and total P contents were low in the profile and ranged between 3.35-5.50 mg kg^{-1} and 102.32-198.24, mg kg^{-1} respectively. However, they showed no trend in the profile.

The exchangeable bases occurred in the following order: Ca > Mg > K > Na with respective mean values of 1.60, 0.50, 0.41 and 0.33 $cmol_c\ kg^{-1}$ in the profile (Table 5.4). The concentration of bases in the surface soil were relatively low but generally increased down the profile. Within the profile, exchangeable Ca ranged between 0.92-2.50 $cmol_c\ kg^{-1}$. It was 1.23-1.27 $cmol_c\ kg^{-1}$ within the 30 cm from the surface and then decreased to 0.92-0.97 $cmol_c\ kg^{-1}$ within the 30-50 cm depth, but further increased to 2.50 $cmol_c\ kg^{-1}$ in the lower half of the profile. However, the ratings by FAO (2006), rates exchangeable Ca concentration in the profile as very low to low. Similarly exchangeable Mg was low and ranged between 0.25-0.70 $cmol_c\ kg^{-1}$ occurring in an irregular pattern in the profile. The exchangeable sodium level was within 0.28-0.42 $cmol_c\ kg^{-1}$ which values fell below the 1 $cmol_c\ kg^{-1}$ critical value beyond which sodium becomes detrimental to plant roots. Exchangeable K ranged between 0.36-0.47 $cmol_c\ kg^{-1}$ in the profile. Following the ratings by FAO, (2006), exchangeable K of the studied profile fell within low to medium bracket (0.2 to 0.4 $cmol_c\ kg^{-1}$) indicating a relatively low presence of K in the profile. The observed values of the exchangeable Na do not seem to a problem within the upper 100 cm of Pedon B. The CEC in the profile ranged between 6.2-7.5 $cmol_c\ kg^{-1}$, indicating low CEC as per the ratings of FAO, (2006). However, CEC was relatively low in the upper half but increased marginally at the lower half of

the profile. The exchangeable acidity of the studied profile was relatively high (2.6 $cmol_c$ kg^{-1}) in the surface soil (0-10 cm) but generally the same within the profile.

Base saturation increased down the profile with the observed values (32-55%) classified as medium as per the classification of Landon (1991).

5.2.3.3 Pedon C

The soil pH (H_2O) in the Pedon C was neutral and ranged between 6.3-6.9 with mean value of 6.5. The pH in $CaCl_2$ was also neutral (6.1 to 6.6) with a mean value of 6.2 (Table 5.3). Total nitrogen content was very low (0.05%) in the 0-10 cm and turned non-detectable down below the 10 cm depth. Whereas the organic carbon ranged from 0.03-0.18%, decreased drastically down the profile it was rated very low as per the ratings of (Blakemore et al. 1987; CSIR-SRI, 2007). The C/N ratio at the surface was 3.6 but not determined within the depth below 10 cm from the surface. The concentration of available P was relatively high at the surface (15.53 mg kg^{-1}) and tended to decrease to 10.08 in the profile. It was medium at the surface. Per the ratings of FAO (1980), the available P within the profile was low as the profile average was less than 15 mg kg^{-1}. However, the concentration of total P was high (Blakemore et al. 1987) and ranged between 94-146 mg kg^{-1} within the profile.

The concentration of exchangeable bases as observed in Pedon D, occurred in the following order: Ca > Mg > K > Na (Table 5.4). The observed exchangeable bases occurred in inconsistent patterns in the profile. Exchangeable Ca was highest (0.74 $cmol_c$ kg^{-1}) in the surface but tended to decrease down the profile in an irregular pattern. However, the observed exchangeable Ca was very low according to the ratings of FAO, (2006). The concentration of exchangeable Mg was within the range of 0.22-0.35 $cmol_c$ kg^{-1} which occurred in irregular pattern in the profile. The observed exchangeable Mg in the profile was low following the ratings of FAO, (2006).

Exchangeable Na in the profile occurred between a marginal range of 0.16-0.19 $cmol_c\ kg^{-1}$ which values fell below 1 $cmol_c\ kg^{-1}$ suggested as detrimental to plant roots. Similarly, exchangeable K was very low to low and ranged between 0.15-0.21 $cmol_c\ kg^{-1}$ in the profile indicating low K in the tailings. The cation exchange capacity (CEC) in the profile was very low and ranged between 2.7-3.7 $cmol_c\ kg^{-1}$ and rated as very low (FAO, 2006) and hence considered as soil with low fertility. The exchangeable acidity was very low (0.7-0.8 $cmol_c\ kg^{-1}$) and almost uniform in the profile. Percent base saturation in the profile ranged between 34-49% and occurred in irregular pattern. This is an indication that greater part of the part of the CEC was occupied by the exchangeable bases. The ratings by Landon (1991) puts BS < 20% as low, 20-60 as medium, and above 60% as high. Hence the base saturation was rated medium to high.

5.2.3.4 Pedon D

The soil reaction pH (H_2O) was neutral (6.1 to 6.5) and showed no trend in the profile. That of $CaCl_2$ was in a range of 5.3 to 6.1 (Table 5.3). The total nitrogen in the surface (0-10 cm) of the profile was 0.06 but became non-detectable in the depth below. The organic carbon contents within 20 cm from the surface was very low (0.41-0.46%). The organic carbon in the soil dropped significantly in the depth below. Whereas the C/N ratio was 7.7% at the surface (0-10 cm) of the profile, the depths below were not determined. The available P ranged between 10.88-16.04 mg kg^{-1} while total P ranged between 123.84-161.24 mg kg^{-1} in the profile. Whereas the available P was rated as low, the total P was very high (Blakemore et al. 1987; FAO, 1980).

The concentration of exchangeable bases in Pedon D occurred in the following order: Ca > Mg > Na > K (Table 5.4). Except for exchangeable Ca that generally decreased with depth, Mg, Na and K occurred in irregular patterns within the profile. Exchangeable Ca was high 0.92 $cmol_c\ kg^{-1}$ in the surface layer but decreased down the profile.

Table 5. 5. Exchangeable bases, CEC, and base saturation of the soil profiles.

Pedon	Depth	Exchangeable Cations ($cmol_c\ kg^{-1}$)				TEB*	Al+H	CEC*	BS*
	(cm)	Ca	Mg	Na	K	$cmol_c\ kg^{-1}$	$cmol_c\ kg^{-1}$	$cmol_c\ kg^{-1}$	(%)
A	**0-10**	0.97	0.75	0.40	0.60	2.7	4.1	14.6	19
(control)	**10-20**	0.91	0.69	0.37	0.56	2.5	4.5	12.8	20
	20-30	0.86	0.60	0.33	0.51	2.3	4.4	13.8	17
	30-40	0.86	0.55	0.31	0.50	2.2	3.6	13.4	17
	40-50	0.88	0.53	0.31	0.46	2.2	3.4	15.5	14
	50-60	0.84	0.57	0.26	0.48	2.1	3.2	15.9	14
	60-70	0.82	0.53	0.22	0.38	2.0	3.2	10.2	19
	70-80	0.83	0.51	0.22	0.25	1.8	3.0	10.1	18
	80-90	0.83	0.40	0.23	0.25	1.7	3.0	11.2	15
	90-100	0.82	0.39	0.24	0.28	1.7	3.0	10.7	16
	Mean	**0.86**	**0.55**	**0.29**	**0.43**	**2.1**	**3.5**	**12.8**	**17**
B	**0-10**	1.23	0.29	0.27	0.47	2.3	2.6	7.0	32
	10-20	1.24	0.25	0.30	0.44	2.2	2.3	6.8	33
	20-30	1.27	0.35	0.28	0.39	2.3	2.5	6.2	37
	30-40	0.97	0.47	0.28	0.36	2.1	2.4	6.3	33
	40-50	0.92	0.44	0.28	0.40	2.0	2.4	6.5	31
	50-60	1.89	0.52	0.38	0.37	3.1	2.4	6.6	48
	60-70	1.94	0.63	0.38	0.38	3.3	2.4	7.5	44
	80-90	2.16	0.70	0.42	0.45	3.7	2.3	7.4	51
	80-90	1.93	0.64	0.40	0.45	3.4	2.3	7.2	48
	90-100	2.50	0.69	0.30	0.38	3.9	2.5	7.0	55
	Mean	**1.60**	**0.50**	**0.33**	**0.41**	**2.8**	**2.4**	**6.8**	**41**
C	**0-10**	0.74	0.33	0.19	0.21	1.5	0.7	3.2	45
	10-20	0.69	0.33	0.18	0.21	1.4	0.7	3.1	46
	20-30	0.69	0.22	0.16	0.17	1.2	0.8	3.6	34
	30-40	0.67	0.33	0.18	0.15	1.3	0.8	3.4	39
	40-50	0.68	0.30	0.17	0.18	1.3	0.8	3.6	37
	50-60	0.69	0.34	0.18	0.19	1.4	0.7	3.6	39
	60-70	0.63	0.35	0.19	0.20	1.4	0.7	3.2	43
	70-80	0.60	0.30	0.17	0.19	1.3	0.7	3.0	42
	80-90	0.67	0.31	0.19	0.16	1.3	0.8	3.7	36
	90-100	0.65	0.28	0.19	0.20	1.3	0.8	2.7	49
	Mean	**0.67**	**0.31**	**0.18**	**0.19**	**1.3**	**0.8**	**3.3**	**41**
D	**0-10**	0.92	0.37	0.26	0.25	1.8	0.9	4.4	41
	10-20	0.90	0.38	0.27	0.26	1.8	0.8	4.1	45
	20-30	0.89	0.41	0.23	0.21	1.7	0.9	3.7	47
	30-40	0.88	0.40	0.21	0.21	1.7	0.7	4.1	42
	40-50	0.88	0.37	0.24	0.19	1.7	0.8	4.1	41
	50-60	0.89	0.38	0.24	0.20	1.7	0.8	4.7	37
	60-70	0.85	0.37	0.24	0.18	1.6	0.9	4.8	34
	70-80	0.73	0.39	0.23	0.19	1.5	0.9	4.5	34
	80-90	0.77	0.39	0.23	0.20	1.6	0.9	4.6	34
	90-100	0.79	0.36	0.23	0.21	1.6	0.8	2.7	59
	Mean	**0.85**	**0.38**	**0.24**	**0.21**	**1.7**	**0.8**	**4.2**	**41**

Table 5. 4. Continued.

Pedon	Depth	Exchangeable Cations ($cmol_c\ kg^{-1}$)				TEB*	Al+H	CEC*	BS*
	(cm)	Ca	Mg	Na	K	$cmol_c\ kg^{-1}$	$cmol_c\ kg^{-1}$	$cmol_c\ kg^{-1}$	(%)
E-9	**0-10**	1.05	0.76	0.27	0.23	2.3	3.8	19.4	12
	10-20	0.97	0.74	0.27	0.20	2.2	3.8	17.4	13
	20-30	0.98	0.75	0.25	0.24	2.2	3.9	17.0	13
	30-40	0.84	0.67	0.27	0.25	2.0	3.8	15.0	14
	40-50	0.95	0.71	0.31	0.23	2.2	3.9	15.5	14
	50-60	0.96	0.78	0.32	0.26	2.3	3.9	16.9	14
	60-70	0.87	0.76	0.32	0.27	2.2	4.5	16.6	13
	70-80	0.69	0.79	0.30	0.27	2.1	4.5	15.5	13
	Mean	**0.91**	**0.75**	**0.29**	**0.24**	**2.2**	**4.0**	**16.7**	**13**
E-18	**0-10**	1.06	0.43	0.28	0.27	2.0	3.9	14.0	15
	10-20	0.99	0.40	0.28	0.26	1.9	4.2	13.9	14
	20-30	0.97	0.38	0.26	0.27	1.9	3.8	12.1	16
	30-40	0.97	0.39	0.27	0.25	1.9	4.1	12.5	15
	40-50	0.87	0.31	0.20	0.27	1.7	3.8	12.1	14
	50-60	0.84	0.31	0.21	0.28	1.6	3.3	12.6	13
	60-70	0.86	0.29	0.21	0.25	1.6	3.1	12.4	13
	70-80	0.80	0.32	0.22	0.27	1.6	3.4	13.5	12
	80-90	0.84	0.31	0.23	0.27	1.7	3.2	12.2	14
	Mean	**0.91**	**0.35**	**0.24**	**0.27**	**1.8**	**3.6**	**12.8**	**14**
E-21	**0-10**	1.32	0.52	0.34	0.26	2.4	7.0	19.7	12
	10-20	0.92	0.51	0.27	0.23	1.9	6.7	19.4	10
	20-30	0.90	0.50	0.26	0.22	1.9	6.6	15.7	12
	30-40	0.78	0.53	0.22	0.27	1.8	6.5	13.9	13
	40-50	0.77	0.54	0.24	0.26	1.8	6.4	12.8	14
	50-60	0.84	0.48	0.23	0.25	1.8	6.6	13.6	13
	60-70	0.83	0.47	0.23	0.26	1.8	5.7	14.8	12
	70-80	0.77	0.49	0.24	0.27	1.8	6.6	15.0	12
	80-90	0.85	0.51	0.23	0.29	1.9	6.5	14.7	13
	90-100	0.84	0.51	0.24	0.26	1.9	5.4	14.2	13
	Mean	**0.88**	**0.51**	**0.25**	**0.26**	**1.9**	**6.4**	**15.4**	**12**

* TEB = Total exchangeable bases; Ex. Acidity (Al+H) = Exchangeable acidity; CEC = Cation exchange capacity; BS = Base saturation.

Based on the ratings of FAO (2006), the observed exchangeable Ca in the profile was very low. The concentration of exchangeable Mg was within the range of 0.36-0.41 $cmol_c\ kg^{-1}$ which occurred in an irregular pattern in the profile. The exchangeable Na occurred between the range of 0.21-0.27 $cmol_c\ kg^{-1}$ while the exchangeable K fell between 0.18-0.26 $cmol_c\ kg^{-1}$. Whereas low

exchangeable Na connotes a good attribute, low exchangeable K on the other hand, implies K deficiency within the profile. The soil showed a very low cation exchange capacity within the range of 2.7-4.8 $cmol_c\ kg^{-1}$ in the profile. Exchangeable acidity was very low and occurred between the range of 0.7-0.9 $cmol_c\ kg^{-1}$ in the profile indicating a good attribute. However, the base saturation was generally high and ranged between 34-59% and occurred in an irregular pattern within the profile.

5.2.3.5 Pedon E-9

In Pedon E-9, soil pH was low (4.1-4.4) in the profile and is classified as strongly acid soil. The pH H_2O followed an irregular pattern within the profile. The pH in $CaCl_2$ (3.7-3.8) was lower than that of H_2O throughout the profile (Table 5.3). The total nitrogen content was very low and almost uniform throughout the profile. The organic carbon (0.22-0.66%) was generally low and occurred in irregular patterns within the profile. Except for the surface soil (0- 10 cm) that had 8.3 C/N ratio, the remaining depth (10-80 cm) fell between the range of 3.7-5.5 in the profile indicating decreasing with depth. The available P was very low (2.12-2.73 mg kg^{-1}) irrespective of high total P (204.80-215.00 mg kg^{-1}) observed in the profile.

In Pedon E-9, the exchangeable bases occurred in the following order: Ca > Mg > Na > K $cmol_c\ kg^{-1}$ showing inconsistent patterns in the profile (Table 5.4). Exchangeable calcium was highest (1.05 $cmol_c\ kg^{-1}$) at the surface (0-10 cm) depth and decreased inconsistently to 0.69 $cmol_c\ kg^{-1}$ at the bottom (70-80 cm). The concentration of calcium was rated very low according to the ratings of FAO (2006). Exchangeable Mg occurred in an irregular pattern within the range of 0.67-0.79 $cmol_c\ kg^{-1}$ in the profile. As per the ratings of FAO (2006), exchangeable Mg in the profile is rated as low indicating Mg deficiency in the profile. However, exchangeable Na and K occurred between the range 0.27-0.32 $cmol_c\ kg^{-1}$ and 0.20-0.27 $cmol_c\ kg^{-1}$ respectively which occurred inconsistently

within the profile. Exchangeable acidity was 3.8 $cmol_c\ kg^{-1}$ at the surface but generally increased down the profile to 4.5 $cmol_c\ kg^{-1}$ at the bottom indicating high exchangeable acidity in the profile. The CEC of the studied profile ranged between 15.0-19.4 $cmol_c\ kg^{-1}$ which occurred in an irregular pattern. Following the ratings of FAO, (2006), CEC of the soil is rated as medium. However, base saturation (12-14%) was very low despite the relatively high CEC of the profile.

5.2.3.6 Pedon E-18

The pH (H_2O) (3.4-3.9) was extremely acid throughout the profile following an irregular pattern. That of $CaCl_2$ fell within the range of 3.1-3.7 (Table 5.3). The total nitrogen occurred within the upper 60 cm and turned non-detectable within the lower 30 cm. However, it continued to reduce from 0.06% in the surface (0-10 cm) to 0.02% in 50-60 cm. Per the ratings of Blakemore, (1987) the observed total nitrogen level within the upper

60 cm was very low. Whereas the organic carbon ranged between 0.08-0.55%, in Pedon E-18, it was very low (Blakemore et al. 1987) and tended to decrease inconsistently within the profile. The C/N ratio in the 0-60 cm was very low and ranged between 3.0-9.2 but was not determined below the 60 cm depth due to no total nitrogen found therein. Although the total phosphorous concentration was substantial (221.36-291.60 mg kg^{-1}) in the profile, available P was very low (3.10-5.57 mg kg^{-1}) (Blakemore et al. 1987).

The concentration of exchangeable bases in Pedon E-18 were high at the surface but generally decreased down the profile (Table 5.4). Generally, the mean concentration of bases in the entire profile was in order Ca > Mg > K > Na $cmol_c\ kg^{-1}$. Exchangeable calcium was relatively high (1.06) at the surface but decreased to 0.80 in the profile. Exchangeable Mg, Na, and K occurred within 0.29-0.43 $cmol_c\ kg^{-1}$, 0.20-0.28 $cmol_c\ kg^{-1}$ and 0.25-0.27 $cmol_c\ kg^{-1}$ respectively. Following the ratings of FAO (2006), Ca, Mg, Na and K were all low. The exchangeable acidity was high

and ranged between 3.1-4.2 $cmol_c\ kg^{-1}$ and occurred in an irregular pattern in the studied profile. CEC was medium (12.1-14.0 $cmol_c\ kg^{-1}$) and occurred in irregular pattern throughout the profile. The mean exchangeable acidity of the soil was high in the profile. However, base saturation ranged between 12-16% with a mean value of 14% in the profile implying that the exchangeable bases occupy a very little portion of the CEC.

5.2.3.7 Pedon E-21

The pH (H_2O) ranged from 3.8-4.0 and classified as very strongly acid to extremely acid. That of $CaCl_2$ was 3.4-3.7 (Table 5.3). Total nitrogen was 0.10% in 0-10 cm and 0.03% in 10-20 cm depth and turned non-detectable in the remaining depth. However, organic carbon was relatively high (2.20%) at the surface and tended to decrease down the profile. Both total nitrogen and organic carbon content in the studied profile were rated very low following the ratings of Blakemore et al. (1987). The C/N ratio was moderate (14-19.2) within the upper 20 cm but below was not detectable. It was, however, high in the surface soil (0-10 cm) and medium in the subsoil (10-20 cm) per the ratings of Blakemore et al. (1987). The total phosphorous ranged between 338.28-362.80 mg kg^{-1} within the upper 30 cm and tended to range between 236.06-299.40 mg kg^{-1} within the lower 70 cm of the profile. According to the ratings of Blakemore et al. (1987), total phosphorus observed in the profile was very high but the available phosphorous (2.27-3.13 mg kg^{-1}) tended to be very low. However, it was high at the surface, but the concentration declined along the profile.

The exchangeable bases of the soil occurred in the following order: Ca > Mg > K > Na $cmol_c\ kg^{-1}$ in the profile (Table 5.4). Values for exchangeable bases were relatively high in the surface soil but decreased continuously with depth. Exchangeable Ca was relatively high (1.32 $cmol_c\ kg^{-1}$) at the surface and decreased inconsistently with depth. It however, occurred between the range 0.77-

1.32 $cmol_c\ kg^{-1}$ in the profile. Following the ratings by FAO (2006), exchangeable Ca levels in the profile was rated as low. Concentration of exchangeable Mg ranged between 0.47 and 0.54 $cmol_c\ kg^{-1}$ which occurred in irregular pattern within the profile. According to the ratings by FAO (2006), the values of Mg in the various depth are rated low connoting Mg insufficiency in Pedon B. Sodium was 0.34 $cmol_c\ kg^{-1}$ at the surface but decreased to 0.22 $cmol_c\ kg^{-1}$ within the profile. The observed values were less than 1 $cmol_c\ kg^{-1}$ which is the highest level for sodium to be detrimental to plant roots. This implies that Pedon E-21 is free from Na toxicity. Moreover, exchangeable K occurred between 0.22 and 0.29 $cmol_c\ kg^{-1}$ in the profile. As per the ratings of FAO (2006), K level in the soil was low indicating that the soil is naturally deficient of K. CEC ranged between 12.8-19.7 $cmol_c\ kg^{-1}$ and was higher in the upper half of the profile than the lower half. However, they are rated as low to medium according to the ratings of FAO, (2006). Exchangeable acidity was highest (7.0 $cmol_c\ kg^{-1}$) in the surface soil but decreased to 5.4 $cmol_c\ kg^{-1}$ in the bottom of the profile.

5.3 Discussions

5.3.1 Morphological characteristics

5.3.1.1 Colour, structure and consistence

The Soil color (moist) *value* and *chroma* of the natural forest, increased with soil depth owing to decrease in organic matter content with depth. The upper 50 cm of the profile had a darker *hue* (10YR) due to accumulation of organic matter therein compared to the lower 50 cm (Table 5.1), but it became yellower (7.5YR) with depth. The yellow colour in subsoil horizons could be due to the oxidation of iron and manganese oxides, indicating the soil is well drained and well aerated.

Soil colour, showed an irregular pattern in Pedon B. Soil colour (moist) was brown (7.5YR5/6) at the surface (0-10 cm), yellowish brown (10YR5/8), light gray (7.5YR8/1), yellow orange (7.5YR7/8) and brown (10YR4/4) in the lower 50 cm. The brown colour at the surface is due to its relatively high organic matter content of the topsoil used for the reclamation. The generally yellow colour within 10-50 cm depth of the subsoil, could be the result of oxidation of iron and manganese oxides. However, the brown colour depicted by the lower 50 cm was also due to relatively high organic matter accumulated in that portion of the profile. Higher organic matter content in the lower 50 cm, relative to the immediate upper part is due to the fact that vegetation was incorporated in the subsoil at the lower portion of the profile during reclamation. The vegetation, upon decomposition over the years is clear testament to the brown colour and subsequently high organic matter in the lower 50 cm. The consistence (moist) being friable in the upper half of the profile was due to the presence of organic matter and intense activities of the earthworms and other soil microbes in that portion of the pedon.

Pedons C and D had a uniform soil colour (moist) of dull brown (7.5YR6/3) and grayish yellow brown (10YR5/2) respectively throughout the entire profile. These two sites were reclaimed with tailings which is a waste product of milled ore, transported in a form of slurry into an embankment or pond. The tailings in the pond become saturated and that oxygen availability becomes limited causing iron and manganese existing in their reduced states. The subdued shade of gray in Pedon D is an indication of reduced condition. However, the dull brown colour depicted by Pedon C might have emanated from the other overburden materials that was added to the tailings during reclamation.

Soil colour (moist) was bright reddish brown (5YR5/8), dull reddish brown (2.5YR6/4) and bright brown (2.5YR6/8) throughout the entire profiles of Pedons E-9, E-18 and E-21, respectively. The

generally reddish brown colours in the studied profiles are often the result of oxidized iron (Fe^{3+}) and manganese (Fe^{2+}) in the soil (Buol et al. 2003). This condition is an indication that the soils were well drained and well aerated.

Generally, the structure of the natural forest soil in the profile showed strong variation from surface to subsurface horizon revealing that there is strong variability in the development of soil structure. Higher content of organic matter at the upper 30 cm and higher proportion of clay in the entire profile are responsible for the variation. The structure of the soil was granular-subangular blocky in the upper half but tended to be subangular blocky in the lower half of the profile. The upper half was high in sand and low in clay and the vice versa in the lower half. These variations may be attributed to the proportion of sand and silt in the two halves of the profile.

In Pedons C and D the structure was single grained and uniform in the entire 100 cm profiles. The observed structure was due to higher proportion of sand, coupled with low contents of clay and organic matter in the profiles. Moreover, the absence of structure observed in these two reclaimed sites suggested minimal pedogenic activities in the tailings considering the upper 100 cm.
Generally, the soil structure was uniform in each Pedon E. However, except for Pedon E-18 that was uniformly granular throughout, Pedons E-9, and E-21 were uniformly subangular blocky in the two profiles. The uniform structure observed in the studied profiles was due lack of reorganization of original grains due to anthropogenic activities rather than cumulative impact of soil-forming processes. Moreover, the observed high clay content relative to sand and silt in each profile is responsible for their high stability.

However, soil consistence (dry) was slightly soft in 0-10 cm depth and tended to slightly hard within 10-20 cm due to high organic matter content. It was hard in 30-50 cm and remained very

hard in the lower half of the profile as a result of high clay content. Whereas consistence (moist) was friable in the upper 30 cm due to higher organic matter, it remained firm in the lower 70 cm of the profile due high content of clay therein. However, consistence (wet) was generally slightly sticky due to change in particle size distribution.

Within Pedon B, soil consistence (dry) was slightly hard in the upper 50 cm depth due to relatively higher sand and lower clay content and became hard within the lower 50 cm in the profile due to relatively low and high contents of sand and clay respectively in that portion. However, consistence (moist) was friable in the upper half of the profile due to the presence of organic matter and intense activities of the earthworms and other soil microbes in that portion of the pedon, while the lower half turned firm due to change in particle size distribution. Meanwhile consistence (wet) remained slightly sticky due to relatively lower proportion of sand.

The consistence (dry, moist, wet) was loose, loose and slightly sticky considering the upper 100 cm in both profiles. This was due the fact that, whereas Pedon C was uniformly packed with a mixture of tailings, waste rock and other overburden materials, Pedon D was uniformly packed with only tailings within 100 cm from the surface. High sand content of the reclamation material and their uniformly packing arrangement is testament to the uniform consistence (dry, moist and wet) throughout the entire profiles.

Soil consistence (dry) was generally very hard, (moist) friable and (wet) slightly sticky throughout the entire profile of Pedon E-9. The consistency of soil at dry, moist and wet exhibited by Pedon E-9 could be attributed to high clay content within the profile. In Pedon E-18, soil consistence (dry) was soft, (moist) very friable and (wet) slightly sticky in all depth of the profile due to the distribution of the particle size. However, in Pedon E-21 soil consistence (dry, moist and wet)

remained hard, friable and sticky from top to the bottom of the profile due to the combined effect of silt and clay.

5.3.1.2 Roots distribution and coarse fragments

The root distribution of the Pedon A was many medium, many fine and many very fines in the upper 30 cm but changed to very few fines in the 30-50 cm with no root found in the lower half of the profile. The natural forest had no or minimal impact of human activities, yet limited roots were found the subsoil. This situation might be attributed to thick litter floor in the forest which prevented germination of lower plants leading to sparse population of lower plants above ground. Again, higher bulk density and the acidic nature of the soil hampered root development and growth in the lower plants.

The roots in the profile of Pedon B were many medium, many fine and many very fines in the upper 30 cm. The abundance of root at upper 30 cm is the result of high population of shrubs, herbs vegetation above the ground. However, no root was found in the deeper horizon and this may be due to the acid nature of the soil, high bulk density and the fact that deep rooted plants were infrequent on site. Soil property such as low pH (Ryan et al. 2001; Rahman and Ranamukhaarachchi, 2003) and high bulk density (Mukhopadhyay and Maiti, 2011) have been reported to decrease root growth and development.

In the profile of Pedon C, root distribution has been very fine, very few fine roots in the upper 20 cm, with no root found in the lower 80 cm of the profile. The observed very fine and very few fine roots found in the 0-20 cm, relative to deeper depth is an indication of high population of shallow rooted plants on site with limited deep-rooted plants. Moreover, the extreme bulk density of Pedon C further impeded root growth and development (Mukhopadhyay and Maiti, 2011).

In Pedon D, root distribution ranged from many medium, many fine and many very fine roots in the surface layer (0-10 cm) to very few fine roots 10-30 cm in the profile indicating abundance of lower plants on the field. However, no root was found between 30 to 100 cm depth of the profile due to the sparse growth of higher plants as a result of frequent fall out of plants during heavy rainfall. The tailings become too loose to anchor the roots of higher plants during heavy rains thereby reducing the population of plants. Furthermore, the sandy nature of the tailings is a clear impediment to growth and development of some plants.

In Pedon E-9, many medium, many fine and few fine roots were found in the upper 20 cm. In Pedon E-18, there were presence of many medium, very fine and few fine roots within 30 cm from the surface. However, there were of few fine and very few fine in the upper 20 cm of Pedon E-21. The roots in the three profiles were few and even not found in the deeper horizons. This limitation of root, indicated that the population of shrubs above the ground were uncommon. Moreover, the strongly to extremely acid condition of the soil and the high bulk density in the profiles accounted for few roots in the profiles as acidic condition and high bulk density hampers root growth and development.

Coarse fragments increased with depth in the profiles of Pedons A and B. However, it remained constant throughout Pedons C, D, and E. In the natural forest, this variation and increase in coarse fragment with depth is the result of intense weathering and cumulative impact of pedogenic activities in Pedon A unlike Pedon B which was heavily influenced by man. Pedon B was reclaimed following the conventional protocol and that packing was done in a backfilled manner to mimic the natural settings.

Pedon C was reclaimed with a mixture of tailings, waste rocks and other overdurden materials. The inclusion of these materials as reclamation materials in Pedon C are responsible for the abundant coarse fragments as classified by Schoeneberger, et al. (2012).

However, no coarse fragment was found in Pedon D since the entire 100 cm profile was uniformly packed only tailings which are the product of finely ground rock after beneficiation of gold. The tailing is basically sand as depicted by the particle size distribution (Table 5.2).

In Pedon E-9, coarse fragments were very few (<2) within the entire profile. It was common coarse (5%) and few coarse fragments in the entire Pedons E-18 and E-21, respectively. The coarse fragments were uniformly distributed in each profile. This indicates that each profile was uniformly packed with the same material.

5.3.2 Physical properties

5.3.2.1 Texture and silt/clay ratio

The mean particle size distributions of the natural forest soil (Pedon A) occurred in the following order of magnitude: sand (25.8%) > silt (27.4 %) > clay (46.8%) with an average clay texture. The upper 20 cm of the natural forest pedon was dominated by sand (43.0-44.5%) but decreased from 44.5% to 20.0% in the subsurface horizons. Conversely, the clay content was relatively low (30.1-35.0%) in the upper 20 cm but increased from 30.1 to 52.5% in the subsurface horizons indicating an increase in clay content with profile depth. High clay content in the natural forest soil is due to the intense weathering activities of the area. However, the consistent increase in clay content with depth may be attributable to translocation of clay particles from the overlying horizons. This corroborates the findings similar in soils elsewhere (Buol et al. 2011; Lambin and Esu, 2011;

Prasad and Govardhan, 2011) who reported clay buildup in subsurface horizons and attributed it to the in-situ development of secondary clays and the weathering of primary minerals in the B-horizon.

In Pedon B, particle size distribution showed average values of 70%, 5.5% and 24.5% for sand, silt and clay, respectively, in the profile. The results on texture indicate that the upper 30 cm was dominated by sandy loam followed by sandy clay loam texture occupying the lower 70 cm. Whereas sand content in the profile decreased with depth, clay on the other hand, increased down the profile indicating migration of clay from top to the underlying horizons. Higher sand content within the profile may be ascribed to the fact that the soil used for the reclamation was stock piled for some years and that intense rainfall of the area had dissolved and washed away the finer fractions to the immediate surroundings leaving behind a relatively coarse textured soil.

The particle size distribution of Pedon D showed mean values of 74%, 13.8% and 12.2% for sand, silt and clay respectively resulting in sandy loam texture. The mean values of 81.5% sand, 6.3% silt and 12.2% clay producing a sandy loam textured soil. Both Pedons C and D were reclaimed with tailings hence higher proportion of sand in the profiles. Tailing has been characterized with high sand content (Lottermoser, 2010).

The texture of the soils from Pedon E was mainly of a clay, except for Pedon E-21 which was clay loam (Table 5.2). Among these sites, the surface layer was generally dominated by clay loam followed by clay in the underlying layers. However, the lower 30 cm of Pedon E-21 was entirely silty clay. Whereas the clay fractions showed consistent increasing pattern down the profile, the silt content showed an irregular increase with depth except for Pedon E-9. The higher clay content depicted by these three sites stems from the fact they were reclaimed with non-plinthic B-horizon.

The B-horizon of the natural forest (Pedon A) has high accumulation of clay (Table 5.2). Also, the increasing trend of clay down the profile is as a result of translocation of finer fractions down the profile.

The ratio of silt to clay at the natural forest pedon decreased with profile depth and was < 1 in all depth. This is an indication that the soil is at an advanced level of development (Abayneh, 2005; Basava et al. 2005), connoting the occurrence of both clay migration and translocation in the profile. Within the profile of Pedon B, the ratio of silt to clay decreased with depth and was < 1 in all depth. The low ratio of silt to clay within the profile shows that the soil is at an advanced level of development as noted by Abayneh, (2005) and Basava et al. (2005). The soil was reclaimed with the original excavated material in a backfilled manner. In Pedon C, the ratio of silt to clay was < 1 in the upper 20 cm but tended to increase to 1 and above in the lower 80 cm. Low silt to clay ratio of the soil in the upper 20 cm is due to the intense weathering activities at the surface to the disadvantage of the lower 80 cm. Moreover, the inclusion of other overburden materials to the tailings as the reclamation material, contributed to the relative higher silt content within the profile. However, the Pedon D had a silt to clay ratio that was < 1 throughout the profile. The relatively lower silt to clay ratio of Pedon D may be due to the relatively older age of the site and the fact that no material other than the tailings was used for the reclamation. Although both Pedons C and D were all of tailings, the differences in silt to clay ratio is mainly attributed to the nature and properties of the parent material or the rocks from which the tailings were produced. Except for Pedon E-21, the ratio of silt to clay was < 1 in Pedons E-9 and E-18. The low ratio of silt to clay of these two pedons indicates advanced level of development (Abayneh, 2005; Basava et al. 2005) irrespective of their ages of reclamation. These sites were reclaimed with natural soils from B-horizons hence higher clay content relative to silt. Although Pedon E-21 is the oldest reclaimed

site, the ratio of silt to clay was generally more than 1 in the upper 40 cm and the lower 30 cm indicating a young development. However, the middle portion (40 to 70 cm) of the profile recorded a less than 1 silt to clay ratio. This variation within the profile may be ascribed to the heterogenous nature of the material used for the reclamation.

5.3.2.2 Bulk density

Generally, the soil bulk density of the natural forest (Pedon A) showed consistent increase with depth. Bulk density was 1.33 Mg m^{-3} in the surface soil which fell below the optimum value (1.40 Mg m^{-3}) for clay loams (USDA-NRCS, 2008). In the upper 10 cm it was less than levels that inhibit root development (Hunt and Gilkes, 1992; McKenzie et al. 2004). It further increased from 1.33 to 1.98 Mg m^{-3} within the lower 90 cm of the profile falling above the optimum value of 1.40 Mg m^{-3} for clay in the lower 90 cm. The bulk density of the natural forest increased with soil depth as a result of a decline in organic matter levels, aggregation, and root propagation (Tsimba et al. 1999) and compaction from the weight of the above strata. Alexander (1980) had earlier reported that soils and horizons with a high concentration of organic matter tend to have a lower bulk density, most likely as a consequence of increased biological activity that leads in the formation of more soil pores.

In Pedon B, bulk density of the studied profile increased with depth (Table 5.2). In the upper 30 cm bulk density ranged between 1.45 to 2.06 Mg m^{-3} which fell higher than the optimum value (1.40 Mg m^{-3}) for sandy loams (USDA-NRCS, 2008). It further increased down the profile to levels higher than the optimum value for sandy clay loam suggested by USDA-NRCS, (2008). The observed results show that bulk density was generally high in the profile and this may be due to compaction caused by the heavy mining equipment used for the reclamation.

Bulk density was typically high in Pedon C. It ranged from 1.98 to 2.29 Mg m^{-3} in the profile and highest among all reclaimed sites. The observed bulk density in the upper 10 cm was higher than the 1.60 (USDA-NRCS, 2008) optimum value for loamy sand soils and 1.40 Mg m^{-3} for sandy loam in the lower 90 cm. This indicates that the bulk density in the profile fell above levels inhibiting root growth (Hunt and Gilkes, 1992; McKenzie et al. 2004). The soil was reclaimed with a mixture of tailings, waste rocks and other overburden materials. Therefore, dumping and spreading of the materials by the heavy mining equipment consequentially compacted the soil resulting in increased bulk density. Moreover, the inclusion of waste rocks in the reclamation materials further exerted weight on the underlying soils. Again, higher sand content in this soil might have caused the observed high bulk density since coarser soils have higher bulk density than finer soils (Weil and Brady, 2017).

Within the upper 30 cm of Pedon D, the bulk density fell below the 1.60 (USDA-NRCS, 2008) optimum value for loamy sand soils. Low bulk density in the upper 30 cm is attributable to the burrowing activities of earthworms, perforation by roots of lower plants and the fact the soil was reclaimed with no or minimal impact of heavy mining equipment. The soil was reclaimed using the spot planting approach which reduces or eliminates the use of heavy mining equipment. However, the bulk density increased down the profile to levels higher than the suggested optimum value for loamy sand and sandy loam. This increase is due to the sandy nature of the tailing and the fact that the overlying materials exert weight on the underlying material.

Generally, bulk density increased with depth in all the three soils of Pedon E with their mean values higher than higher than their respective optimum values of 1.10 Mg m^{-3} for clays and 1.40 Mg m^{-3} for clay loams. The generally higher bulk density is the results of compaction caused by

heavy mining equipment used for the reclamation activities. However, the consistent increase in bulk density with depth is due to compaction caused by the weight of the overlying layers.

5.3.2.3 Aggregate stability

Generally, the most stable soil aggregates were found in the top 0-10 cm, while the least stable aggregates were found in the subsurface layer 10-20 cm profile in the natural forest (Table 5.2). The aggregate stability was 4.16 in 0-10 cm and 3.68 in 10-20 cm depth with an average stability of 3.92 mm within the plough layer. The aggregate stability of the natural forest soil (Pedon A) was very high (> 2 mm). This property of the natural forest might be attributed to high organic matter content, high clay content and probably low bulk density of the soil. The results corroborate the findings of Tisdall and Oades (1982) who observed that the stability of soil aggregates is dependent on soil organic matter and clay content. Similarly, Siddique et al. (2017) reported that soils with high organic matter generally have high aggregate stability. They described organic matter as not only a binding agent but an aid to bringing the negatively charged clay matrix together for flocculation which is crucial for aggregate stability. In a study conducted on forest soils, Chenu et al. (2000) reported greater aggregate stability with the distribution been dominated by the >2 mm size fraction and attributed this trend to high organic carbon of the forest soils. Similarly, Six et al. (1998); DeGryze et al. (2004) reported that, naturally, undisturbed soils have good soil structure and are richer in soil organic carbon than disturbed lands. The natural forest under studied recorded greater aggregate stability than all reclaimed sites in the study area.

The stability of the soil aggregates in Pedon B was high (> 2 mm) in both 0-10 cm and 10-20 cm depth in the profile. Although the mean sand content at the plough layer was very high (77.5%) as against a low mean clay content of 17.5%, the aggregate stability within the plough layer was nonetheless high. Meanwhile, high sand content has been reported by Siddique et al. (2017) to

decrease aggregate stability since the sand possesses no charge. However, the relatively high stability of this soil, regardless of high sand content may be ascribed to the high organic matter in the plough layer.

The aggregate stability reclaimed sites with tailings provenance (Pedon C and D) were all low (< 2 mm). The mean stability of 0.67 mm and 0.43 mm for Pedon C and D, respectively, was noted. These are anthropogenic soils which recorded, high sand content, low clay content, low organic matter content, high bulk density, poor structure characteristically caused the low aggregate stability in these two profiles. Several studies have observed a parallel decline in aggregate stability with decreasing soil organic matter (Greacen and Peckman, 1953; Kemper and Koch, 1966, Tisdall and Oades 1980) and clay content (Siddique et al. 2017). Similarly, high sand content has been reported by Siddique et al. (2017) to decrease aggregate stability since the sand possesses no charge. Therefore, the observed aggregate stability of Pedon C and D are consistent with the findings of these authors.

Generally, aggregate stability in the surface (0-10 cm) were consistently lower than in the 10-20 cm in Pedons E-9, E-18, and E-21, which were under the same mode of reclamation. This was contrary to the aggregate stability results reported from Pedon A (natural forest). Pedons E-9, E-18, and E-21 are degraded but reclaimed mine sites whose soil matrices have been drastically altered following mining and reclamation activities. Several authors have reported that mining activities result in decrease in soil organic matter content of the surface layer. Consequently, the stability of soil aggregates may be affected as a result of drop in soil organic matter content remaining in the soil. However, the stability of the studied profiles was high (>2mm) regardless of

their low organic matter content. High aggregate stability in this regard, might be attributed to higher clay content present in the soils.

5.3.2.4 Moisture characteristics

The soil water content at field capacity (1/3 bars) and permanent welting point (15 bars) varied from 31.6 to 43.2% and 22.0 to 31.3%, respectively, in Pedon A. Although, variation in moisture retention was inconsistent, relatively higher values were recorded in the upper 40 cm, while lower values were observed in the subsurface horizons (Table 5.2). Higher values of available water capacity in the upper portion of the profile were influenced by high content of organic matter, silt, and relatively low bulk density. Haynes and Beare (1996); Rawls et al. (2003) found that available water capacity related positively to organic matter and silt content. Similarly, Charman and Murphy (1998) reported that available water capacity to a large extent depends on the bulk density and the silt content. However, lower values of available water capacity observed in the lower 60 cm is due to the relatively high bulk density, low content of silt and organic matter in that portion of the profile. Yet, higher clay content in the lower 60 cm kept the AWC within the high range according to the ratings of Miller, et al. (2010).

According to the ratings by USDA-NRCS, (1998), moisture content at both suctions (1/3 bar and 15 bars) were low within Pedon B. However, the available water capacity in a range of 6.6 to 8.5% were moderate according to the ratings of Miller, et al. (2010). The generally low moisture content may be due to the higher sand content, low silt content and high bulk density in the profile.

In Pedon C, the moisture content at field capacity and permanent wilting point ranged from 13.0 to 18.8% and 8.0 to 10.9% respectively. Moisture content of Pedon D at field capacity was between 13.6 to 17.8% and permanent wilting point ranged from 8.0 to 10.9% in the entire profile. Whereas

the available water capacity in Pedon C ranged from 5.0 to 9%, that of Pedon D ranged from 5.9 to 6.9% (Table 5.2). The values observed at the two-suction pressure in the two pedons were rated low according to USDA-NRCS, (1998). However, the available water capacity in both sites were moderate as rated by Miller, et al. (2010). Low values of moisture content in Pedons C and D might be attributed to high sand content, high bulk density, low content of silt and organic matter. Generally, moisture content at FC and PWP and the available water capacity in Pedons E-9, E-18 and E-21 were high (Table 5.2). According to the ratings by the USDA-NRCS 1998, moisture content at field capacity in all three sites was rated as low to high, whereas moisture at permanent wilting point fell within medium to high range. However, the available water capacity was rated as high to very high in all three sites according to the ratings by Miller, et al. (2010). The values of AWC were influenced by silt and clay contents across the three profiles regardless of their low organic matter content and high bulk density.

5.3.3 Chemical properties

5.3.3.1 Soil pH

Generally, the soils from the study area are acidic. Except for the reclaimed tailings (Pedons C and D) which were in neutral range, the remaining reclaimed sites and the natural forest (Pedon A) were strongly acid to extremely acid (Table 5.3). The pH of the tailings from Pedons C and D were generally higher than the natural forest soil and the other reclaimed sites. This was consistent with Chileshe, (2014) who reported a higher pH of tailings over natural soils. However, Ganjegunte et al. (2009) attributed this trend of high pH of reclaimed sites relative to the natural soils to the effects of anthropogenic activities. On the other hand, other authors such as McClure et al. 2009; Ndur and Buah 2015; Yassir et al. 2015; Faber, 2017 and Guo et al. 2022 have ascribed this

phenomenon to the presence of carbonates and bicarbonates in the tailings. But Festin et al. (2018) hold the view that tailings are not always neutral to alkaline but can also be acidic depending on the nature and property of the parent material.

The acidic condition depicted by the natural forest and other reclaimed sites could be attributed to high exchangeable acidity and low base saturation (see Tables 5.3 and 5.4) and high rainfall pattern of the study area. This is because, as base saturation is low, more exchange sites are filled with acidic cations, leading to a lower pH. Zhang, (2017), reported that acidic conditions in soils are mainly caused by the acidic parent material, organic matter decay and rainfall. He further singled out rainfall as the major and effective cause of acidic condition in soils. His assertion confirms why pH was generally low in the study area which is located in the high rain forest zone and receives between 1750 and 2000 mm of annual precipitation on average (EAU, 1990). The observed low pH values from the reclaimed soils and the natural forest were consistent with the findings of Tetteh, (2010) who reported low pH on soils from the study area. The extremely acid condition in the upper 40 cm of the natural forest could be the result of high organic matter content in that portion of the profile (Table 5.3). The natural forest (Pedon A) has a thick cover of litter, which decomposes further to release H^+ into the soil, causing extremely acid in the upper 40 cm which was reported to be high in organic matter. This corroborated the findings of Riha et al. (1986) who attributed low pH under canopies to decomposition of organic matter.

Generally, the reclaimed sites had relatively higher pH than the natural forest. Tetteh et al. (2010) also observed a similar trend on some soils from the study area. Ganjegunte et al. (2009) had earlier ascribed high pH values in disturbed soils to anthropogenic effect. However, Pedons E-9, E-18 and E-21 had very low pH regardless of been drastically disturbed through mining and reclamation activities. This condition is mainly attributed to high exchangeable acidity, low base saturation

(see Tables 5.3 and 5.4), cumulative impact of heavy rainfall pattern or the soils may have originated from acidic rock rather than anthropogenic effect as asserted by Ganjegunte et al. (2009).

5.3.3.2 Total organic carbon and total nitrogen

Except for Pedon A, the organic carbon of the reclaimed soils was low and decreased down the profile (Table 5.3). This trend is due to accumulation and decomposition of leaf litterfall, twigs and roots of plants, and other organic material on the forest floor. This was consistent with Obeng (2000) who reported higher accumulation of soil organic matter in the surface horizons of soils in the forest belt of Ghana. He ascribed this trend to biomass effects on surface. Soil organic carbon in the forest soil ranged from very low to medium (0.30 to 4.12) in the profile according to the ratings by Blakemore et al. (1987) and was highest in the surface but tended to decrease down the profile. High soil organic carbon at the surface is due to correspondingly high organic matter recorded by the surface soil. Meanwhile Ghosh et al. (1983) had earlier reported that total organic carbon greater than 0.75% indicates good fertility. This is a clear indication that the natural forest is fertile and suitable for crop production.

Higher organic carbon was found in the surface of Pedon B relative to Pedons C, D and E. This was due to accumulation of biomass at the surface as a result of type of vegetation established on the site. Binkley and Giardina (1998), reported that tree species differ in their biomass production and tissue nutrient concentrations. The floristic composition of the site reveals high population of shrubs, herbs and grasses reported to have less lignin content. Again, the site was reclaimed following the conventional protocol of reclaiming degraded mine soils and the original topsoil was used in situ. Moreover, cleared vegetation was incorporated in the reclamation material hence presence of organic matter even at the bottom of the profile. Similarly, high organic carbon content

was found in the surface but declined with profile depth. Per the ratings of Ghosh et al. (1983), the organic carbon at the plough layer indicates good fertility.

The carbon content was generally low in Pedon C and D. Within these Pedon, the organic carbon occurred between 0.03 to 0.18% and 0.05 to 0.46%, respectively. The very low organic carbon content exhibited by these two sites is due to the fact they are basically tailings. Tailings have been reported to have low contents of organic carbon (Cooke and Johnson 2002; Titshall et al. 2013) due to their impoverished nature. The tailings are not natural soils and are very young per their existence. They are completely human caused soils derived from milled ore or rocks.

In accordance with CSIR-SRI, (2007) ratings, organic carbon content of Pedons E-9, E-18 was low except for Pedon E-21, which was in the range of low to high (Table 5.3). The general low levels of soil organic carbon in the three soils reflects the impact of anthropogenic activities associated mining and reclamation. Indorante et al. (1981), reported that degraded or disturbed lands usually have a very low content of organic carbon due to erosion, leaching and faster decomposition of soil organic matter on the disturbed surface. Similarly, Offiong and Iwara (2012), ascribed decline in soil organic matter content on disturbed lands to changes in temperature, soil moisture, and the quality of biomass returned to the soil. Furthermore, the upper 100 cm of each site under consideration was packed haphazardly with non-plinthic B-horizon making the soils impoverished and deprived of soil organic which hitherto resides in the topsoil. However, organic carbon obtained in the surface of Pedon E-21 was due to the age of the tree species resulting in accumulation of leaf litter fall leading to the relatively high organic carbon production.

The total nitrogen in the study area was generally low. In accordance with the ratings by Tekalign et al. (1991), total N < 0.05% as very low, 0.05-0.12 low, 0.12-0.25 medium and > 0.25 high. Therefore, the overall total N values in the reclaimed soils and the natural forest soil, ranged from low to medium (Table 5.3). The total nitrogen content followed the trend of the organic carbon in the profile since much of the former is derived from the latter and are strongly correlated.

High total nitrogen recorded in the upper 30 cm in natural forest (Pedon A) was due to a correspondingly high organic carbon in that portion of the profile. This was consistent with Darmawan et al. (2006), who observed greater nitrogen content in the surface soil layer. These authors attributed this trend to accumulation and mineralization of organic matter at the surface. With regard to the reclaimed sites, a generally low total nitrogen observed across the profiles was due to a corresponding low organic carbon. Low levels and to some extent, non-detectable total nitrogen observed among reclaimed sites was due to their low carbon content, their acidic nature and the type of vegetation established on them. On the other hand, the relatively high total nitrogen portrayed by topsoil of Pedon B was due to its high organic carbon content. Organic carbon content correlate positively with total nitrogen status since much of soil nitrogen is derived from organic matter. Galloway et al. (2004) had earlier reported mineralization of organic carbon and atmospheric nitrogen by nitrogen-fixing bacteria, as the main sources of soil nitrogen. Although leguminous plants were established on the reclaimed sites, their contributions to nitrogen were negligible due to the acidic nature of the soils as noted by Hart et al. (2013). Hart et al. (2013) asserted that in acid soils, legume roots may have few nodules, or their nodules may not be effective at N fixation by bacteria of the genus Rhizobium. Hence the N fixing ability of the leguminous plants on the field might be hampered. Meanwhile Lawrie, (1981) indicated that 75% of soil nitrogen emanates from leguminous plants. It is therefore, clear that low organic input and

low pH causing ineffective nodulation and N fixation by the legumes, is a clear testament to their poor N status in the reclaimed soils and by extension the study area.

5.3.3.3 Carbon to nitrogen ratio

The C/N ratio, which is also an index of soil fertility (Tisdale et al. 1990) is shown in Table 5.3. The C/N ratio was generally low in the study area. In accordance with the ratings by CSIR-SRI, (2007) the C/N ratio of the selected sites was in good to moderate quality category. However, in some instances, the ratio of carbon to nitrogen was not determined in lower depth across a number of profiles. This situation is highly attributed to the occurrence of no total nitrogen in that portion of the profile. According to CSIR-SRI, (2007), C/N <13 indicates good quality organic material. Therefore, except for Pedon E-21 which recorded C/N ratio between 14 and 19.2 (rated as moderate quality), the remaining reclaimed sites and the natural forest occurred below 13. The quality of C/N ratio is largely dependent on the quality organic matter. The moderate soil quality of Pedon E-21 indicates that the organic matter was not fully decomposed and serves an indication of decline in soil fertility (Lal, 1973).

5.3.3.4 Total P and available P

In general, the available P content in all the soils was low and showed a decreasing trend with depth. The surface soil had relatively higher available P than the subsurface soil which might be attributed to a decrease in soil organic carbon content with depth. Andrews (1998) reported that the availability of phosphorus in soils is dependent on the organic matter content and pH of the soil substrate. Organic matter has a positive effect on P dynamics in soils.

The available P content of Pedons C and D was relatively higher than the other reclaimed sites and the natural forest. According to the ratings by CSIR-SRI, (2007), available P of the tailings falls

within the moderate category. Although, the tailings recorded a very low content of organic matter, the available P was relatively higher compared to the other sites. The relatively high concentration of available P in the tailings is a reflection of neutral soil reaction. This was consistent with Havlin et al. (1999) who reported soil pH between the range of 6.0 and 7.5 as ideal for P availability.

According to the ratings by CSIR-SRI, (2007), the available P content in the other reclaimed soils and the natural forest soil falls under very low to low category across the entire profiles indicating that available P could be limited in the soils from the study area. From the observed data, the available P content was generally low in the forest soil regardless of its high content of organic matter. The available P was also low in the other reclaimed sites. The low available phosphorus recorded by the soils from the reclaimed sites and the natural forest was due to the strongly acid to extremely acid nature of the soils. Under very strongly acid to extremely acid condition P is highly fixed (Havlin et al. 1999, Muindi, 2019) which situation accounted for the low available P in the study area. The observed low available P in the soils under study was consistent with Muindi, (2019) who reported that soils with low pH will have reduced phosphorus availability. The observed results indicate that available P is limited in the reclaimed mine soils and natural forest.

5.3.3.5 Exchangeable bases and CEC

In accordance with the ratings by Blakemore et al. (1987), the exchangeable bases of the natural forest soil and the reclaimed soils were generally low and varied within and across profiles. In the Pedon B, and tailings (Pedons C and D), the concentration of cations was very low. This was due to their high sand content, low clay and organic matter levels, which resulted in very few exchange sites. Soil organic matter content and clay have more exchange sites and are, therefore, reported to have high concentration of cations. The natural forest soil had the highest organic matter content and high proportion of clay, yet recorded low exchangeable bases. Similarly, the other reclaimed

sites (Pedons E-9, E-18, E-21) generally had high clay content relative to sand and silt, yet recorded low concentration of the basic cations. Conclusively, low levels of exchangeable basic cations observed from the studied soils may be a result of the heavy rainfall trend of the study area causing excessive leaching of the basic cations. It may as well be due to paucity of these cations in the parent material. The predominantly low pH of the soils from the study area accounts for the low concentrations of these basic cations. The findings of this work corroborate that of Suharta (2010), who reported a low to very low concentrations of exchangeable bases (Ca, Mg, K, and Na) in acidic minerals.

The CEC at pedon A was low to medium in the profile (Blakemore et al. 1987). The highest CEC (15.9 $cmol_c\ kg^{-1}$) was found in (Bt) horizon indicating the presence and accumulation of clay therein. In general, CEC decreased down the profile and might be attributed to decrease in organic matter with depth. Soils with high organic matter content generally have high CEC. Although the natural forest was conspicuous of high contents of organic matter and clay, the overall CEC in the profile was rated medium (Blakemore et al. 1987). This is due to the high rainfall pattern in the study area resulting in leaching the basic cations.

Within Pedon B, CEC varied between 6.2 and 7.5 $cmol_c\ kg^{-1}$. The CEC occurred in an irregular pattern in the profile. However, the Blakemore et al. (1987) ratings put the observed CEC into very low to low category. This result is attributed to the high sand content of the soil and the leaching effects on the basic cations.

The low CEC recorded by Pedons C and D was due to their high sand content. According to Brady and Weil, (2008), fine-textured soils (clay) have high CEC, while coarse-textured soils (sand) have low CEC. Again, low exchangeable bases and low organic inputs in the tailings might have contributed to such low CEC.

The CEC of soils across the three pedons (E-9, E-18, E-21) varied between 12.1 and 19.7 $cmol_c$ kg^{-1} and are rated medium according to the ratings by Blakemore et al. (1987). The relatively higher CEC among these three pedons (E-9, E-18, E-21) was due to higher content of clay found in each profile. In addition, the CEC of the soil profiles followed the trend of exchangeable basic cations, indicating that basic cations are the primary ion contributors in exchange complexes. Therefore, low exchangeable bases of these sites are clear reflections on the amount of CEC observed. However, the consequential effects of the heavy rainfall pattern in the area leaches the basic cations, which consistently renders the soil acidic leading to a general low CEC among the studied profiles.

5.3.3.6 Exchangeable acidity (Al+H) and base saturation

The exchangeable acidity in the natural forest and the reclaimed sites were high across the profiles. The concentration of exchangeable acidity observed from the forest soil was high and varied inversely to the soil pH (see tables 5.3 and 5.4). It was high in the reclaimed sites as well. Largely, exchangeable acidity values greater than 0.30 $cmol_c$ kg^{-1} are noted to be high (Funakawa et al. 2016; Onwuka et al. 2016). The observed results indicate that the exchangeable acidity from Pedon A and the reclaimed sites (Pedons B, C D and E) were all greater the 0.30 $cmol_c$ kg^{-1}. This was due to leaching caused by excessive rainfall in the study area.

Base saturation in the soils from the study area were generally low and occurred in irregular patterns in the profiles. The generally very low base saturation of the soils from Pedon A, might be credited to the very strongly acid to extremely acid condition and the very low basic cations found in the soil. The base saturation in the forest soil was <50% throughout the profile and this indicates low soil fertility status (Msanya, et al. (2001).

According to the ratings by Blakemore et al. (1987), base saturation of Pedon B was rated low to medium (31-51%) and occurred in an irregular pattern in the profile. Msanya, et al. (2001) rated soils with base saturation > 50% as fertile soils. However, the 41% mean base saturation in the profile indicates that the soil is not fertile as suggested by Msanya, et al. (2001) but performed better than the other reclaimed sites and the natural forest due to their relatively lower pH.

The base saturation ranged between low to medium in Pedons C and D with a mean base saturation of 41% each representing the proportion of bases occupying the CEC. Base saturation was higher in these two pedons than the other reclaimed pedons and the natural forest. This was consistent with Tetteh, (2010), who reported higher percent base saturation in reclaimed soils with high pH values.

As per the ratings by Blakemore et al. (1987) the base saturation in Pedons E-9, E-18 and E-21, fell in the very low category (< 20%). The very low base saturation of these three reclaimed sites implied that a less than 20% basic cations occupied the exchange sites, making the soil low in fertility. This could be attributed to their low pH which occurred from leaching of the basic cation. Since base saturation shows the degree of basic cation leaching of basic cations, the results from the studied profiles indicate that the soils were highly leached (14-20%) and infertile.

5.4 Conclusions

Four modes of reclamation were adopted by AngloGold Ashanti Iduapriem Mines, Tarkwa.

Except for the Pedon B, that was reclaimed following the conventional protocol of reclaiming degraded mine soils, the other reclaimed sites did not follow the recommended protocol.

The mode of reclamation affected the morphology and physicochemical characteristics of the reclaimed sites considering the upper 100 cm. Like Pedon A, the Pedon B had varied soil colour,

structure, consistency and coarse fragments linearly in the profile. However, the other reclaimed sites, had these characteristics uniformly occurred in their profiles.

The use of heavy mining equipment in transporting, dumping and spreading the reclamation materials, compacted the soils. This resulted in high bulk densities, poor soil structure and limited root distribution within profiles of the reclaimed sites. However, the reverse was the case in the natural forest.

The generally strongly acid to extremely acid nature of the soils and the depleted exchangeable cations was due to high amount of exchangeable acidity and the intense rainfall pattern in the area. But the neutral pH class of the tailing provenance reclaimed sites (Pedons C and D) was due to the fact during gold beneficiation, pH is raised to prevent the formation of toxic hydrogen cyanide gas and to optimize gold recovery.

The natural forest recorded the highest soil organic carbon and total nitrogen. With regard to the reclaimed sites, it occurred in the following order: Pedon B > Pedon E-21 > Pedon E-9 > Pedon E-18 > Pedon D > Pedon C. Considering the different modes of reclamation, the Pedon B which followed the conventional protocol came close to the natural forest compared to the other older reclaimed sites.

Considering the vertical cross section of the individual profiles of the reclaimed soils, it is clear to conclude that the similar and uniform properties observed for the reclaimed profiles is due to cumulative impact of anthropogenic activities rather than pedogenic.

CHAPTER SIX

QUALITY OF SOME REHABILITATED SOILS

6.1 Introduction

Surface mining is an anthropogenic activity that causes drastic disturbances to the natural soil environment (McSweeney and Jansen, 1984) and the terrestrial ecosystems. In effect, surface mining and its allied activities create huge overburden materials, tailings, chemicals and voids on the mine. The overburden materials which mostly consists of large boulders and loose rocks fragments, and other mine waste materials are dumped on the mine. These waste dumps, chemicals and voids on the mine, create horrid and devastating scenes in the mining environment. However, mining companies are obliged by the mining and environmental laws to reclaim these areas to condition that most resemble the original status.

The reclamation process involves earthworks, addition of subsoil, topsoil, soil amendment, and vegetation establishment. Over time, natural succession also speeds the process of soil regeneration (Javurek, 1999). As a requirement, reclamation strategies should bring about an improvement in soil quality in mining areas (Tripathi et al. 2016) and the development of pedogenic processes to ultimately support revegetation (Dimitriu et al. 2010). There are various methods to monitor and assess successful reclamation, but one of the most crucial metrics for reestablishing healthy ecosystems after mining is soil quality.

Soil quality refers to the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, and to sustain plant productivity while maintaining or enhancing water quality, supporting human health and habitation, and reducing soil degradation (Doran et al. 1994; Karlen et al. 1997; Karlen et al. 2003). Soil quality is a complex functional concept and cannot be

measured directly in the field or laboratory (Stockings, 2003), but can only be inferred from soil biogeochemical characteristics (Diack and Scott, 2001).

The assessment of soil quality is crucial in soil reclamation process (Zhao et al. 2013) and must include soil attributes that are sensitive to management practice. Soil quality index is a valuable tool for evaluating the productivity of reclaimed land and soil management practices (Tesfahunegn, 2014). Identification of soil attributes that are sensitive to management is crucial in soil quality assessment. Several authors have proposed a number of soil indicator parameters to assess soil quality. Gil-Sotres et al. (2005) indicated that soil indicator parameters can be used individually as simple indices or integrated into index. However, the integrated soil quality indices are reported to be highly suitable since it provides a reliable index than the individual properties (Gil-Sotres et al. 2005). Consequently, several soil quality indices have been established to assess the quality status (Andrews et al. 2002; Velasquez et al. 2007; Rossi et al. 2009; Sinha et al. 2009; Amacher et al. 2007; Asensio et al. 2013; Mukhopadhyay et al. 2013; Zhao et al. 2013). However, the most widely reported methods of soil quality determination are the expert opinion and statistical tools (e.g., linear or multiple regression, pedotransfer function, principal component analysis (PCA)) (Bouma, 1989; Larson and Pierce, 1991; Doran and Parkin, 1994; Li and Lindstorm, 2001).

Previous studies in Ghana have shown that the quality of reclaimed soils at various ages of rehabilitation differs from that of the native soils (Dorgbetor et al. 2012; Tetteh et al. 2015). However, these studies seldom pay attention to the integration of biogeochemical characteristics of soils. Dorgbetor et al. (2012) estimated the quality of some rehabilitated mined soils within the concessions of AngloGold Ashanti Obuasi Mines following the method outlined by Amacher et al. (2007). They reported a 36.5% soil quality index for the forests native soil compared to 32.5%

for the reclaimed mined soils. In this method, soil quality indicators were selected from only chemical and physical soil properties according to an expert-based judgment of the whole minimum data set. The selection of indicator parameters was done on the principle that certain soil characteristics wield more influence on soil quality than others. The selected soil properties were assigned threshold values based mainly on literature and the researcher's opinion. The individual index values were summed up using a simple additive method to obtain a total soil quality index. Tetteh et al. (2015), also assessed the quality of reclaimed soils and natural forest within the concessions of AngloGold Ashanti Iduapriem Mines, Tarkwa based on qualitative assessment of unscreened total data sets of chemical and physical soil properties. They opined that nutrient levels of reclaimed soils were higher than that of the native soil since the reclaimed soils had heterogeneous and dynamic substance with variable C content and improved pH compared to the native soil. These authors assessed the quality of reclaimed mined soils using only physical and chemical indicators following the expertise-based approach. Meanwhile, the integration of physical, chemical and biological indicators produces a more reliable index value (Li et al. 2013). The approach adopted by these authors may however produce unrealistic index values (Lu, 2008) since the expertise-based approach is subject to the discretion of the researcher (Asensio et al. 2013) which is prone to high level of disciplinary bias.

The use of a statistics-based models to estimate soil quality index is robust and can integrate soil biological, chemical, and physical indicator parameters (Mandal et al. 2008) to develop a reliable soil quality index (Andrews et al. 2002; Li et al. 2013) to better reflect the status of soil quality (Masto et al. 2009).

The objectives of this study were to:

a. Investigate the properties of composite surface soils of the rehabilitated mined sites.

b. Assess the quality of rehabilitated mined soils.

6.2 Results

Composite soil samples were collected at the study sites namely, Site A (natural forest or control), Site B (rehabilitated soil with conventional protocol), Site C (rehabilitated soil with a mixture of tailings, waste rocks and other overburden materials), Site D (rehabilitated soil with only tailings), and Site E (rehabilitated soil with only subsoil). Data on selected properties for soil quality assessment especially those influenced by human activity, are presented in the following sections.

6.2.1 Physical indicators

The analytical results of the physical characteristics of the soils are given in Table 6.1.

6.2.1.1 Texture and bulk density

The texture at Site A (control site) was clay loam. Among reclaimed sites, the texture at Site B was sandy loam whereas at Sites C and D it was loamy sand. Except for Site E-21 which was clay, the soils at Sites E-9 and E-18 were clay loam. At Site B, the bulk density was 1.45 Mg m^{-3} which was higher than the 'optimum' value (1.40 Mg m^{-3}) for sandy loams (USDA-NRCS, 2008). The Site C soil had bulk density of 2.00 Mg m^{-3}, which was highest among all reclaimed sites. The bulk density at Site C soil was higher than the 1.60 (USDA-NRCS, 2008) optimum value for loamy sand soils. At Site D, the bulk density was 1.48 Mg m^{-3} which fell below the 1.60 (USDA-NRCS, 2008) optimum value for loamy sand soils. Bulk density values were 1.42 Mg m^{-3}, 1.75 Mg m^{-3} and 1.40 Mg m^{-3} respectively at Sites E-9, E-18 and E-21. These values were higher than their respective optimum values of 1.40 Mg m^{-3} for clay loams, 1.40 Mg m^{-3} for sandy loams and 1.10 Mg m^{-3} for clays. At the control natural forest (Site A) bulk density of 1.30 Mg m^{-3} was lower than the 'optimum' value (1.40 Mg m^{-3}) for clay loams (USDA-NRCS, 2008).

6.2.1.2 Aggregate stability and AWC

The aggregate stability of soils from the study area ranged from 0.44 mm to 4.17 mm MWD. In the control soil (Site A), the stability of soil aggregate was highest with 4.17 mm MWD. The stability of soil aggregate was 2.57 mm MWD at Site B, 0.67 mm at Site C, 0.44 mm at Site D. At Sites E, aggregate stability occurred in the following order 3.97 mm > 2.94 mm > 1.23 mm for E-21, E-9 and E-18 respectively.

The AWC of the study area varied among sites and across different mode of reclamation. Whereas the available water capacity at Site B was 9.28%, it was 6.34% at Site C and 6.09% at Site D. At Sites E-9, E18 and E-21, the AWC were 12.56%, 10.3% and 12.81%, respectively. It was 12.01% at Site A.

6.2.1.3 Coarse fragments

At the reclaimed sites the highest coarse fragments (> 2 mm size) was found at Site C (>45%). The coarse fragments of the other reclaimed sites ranged from 0 to 8% coarse fragments. However, in the natural forest (Site A) no coarse fragments were recorded.

6.2.2 Chemical indicators

6.2.2.1 Soil pH

The pH of soils from the study area was generally low. At Site B, the pH was 4.9 which is classified as very strongly acid. The respective pH at Site C and D of 6.7 and 6.4 were classified as neutral. In the other reclaimed soils, Sites E-9, E-18 and E-21 had pH values of 3.88, 3.72 and 3.81, respectively. Soils from Site E were extremely acid. The natural forest was extremely acid with a pH value of 3.6.

6.2.2.2 Total organic carbon, total N and total P

The total organic carbon content in the natural forest (Site A) was 3.93%. That of the reclaimed sites ranged from 0.14% to 0.94%. In the reclaimed sites, Sites B, C, D, E-9, E-18 and E-21 had TOC contents of 0.94%, 0.14%, 0.30%, 0.39%, 0.52% and 0.82%, respectively. Whereas the total nitrogen content was adequate (0.41%) in the natural forest (Site A) and moderate (0.17%) at Site B, the remaining reclaimed sites fell below 0.1% which is classified as low. The total P content at Site A was 305 mg kg^{-1}. It was 187 mg kg^{-1} at Site B, 127 mg kg^{-1} at Site C and 159 mg kg^{-1} at Site D. Among Site E, E-9 had 210 mg kg^{-1}, E-18 had 281 mg kg^{-1} and E-21 had 359 mg kg^{-1}.

6.2.2.3 Exchangeable acidity (Al+H) and CEC

The exchangeable acidity of soils from the study sites was high. At Sites A, B, C, D and E, exchangeable acidity was higher than 0.30 $cmol_c\ kg^{-1}$. CEC is often used to measure soil fertility and nutrient retention capacity. The CEC of the soil at Site B was 6.72 $cmol_c\ kg^{-1}$. Whereas the CEC at Site C was 3.51$cmol_c\ kg^{-1}$, that at Site D was 3.83 $cmol_c\ kg^{-1}$. The respective CEC of soils at Site E was, E-9, E-18 and E-21 for 4.72 $cmol_c\ kg^{-1}$, 3.98 $cmol_c\ kg^{-1}$ and 14.20 $cmol_c\ kg^{-1}$. Except for Site E-21 that had a moderate CEC, the remaining sites were low. However, the natural forest (Site A soil) had a moderate CEC of 13.04 $cmol_c\ kg^{-1}$.

6.2.3 Biological indicators

6.2.3.1 Microbial biomass carbon, nitrogen and labile carbon

Microbial biomass carbon (C_{mic}) in all selected site was in the range of 12.84 – 63.75 mg kg^{-1} of soil. Among the reclaimed sites, Site B (56.99 mg kg^{-1}) was high to the level close to the natural forest after the eight years of rehabilitation. Whereas Site C had 37.53 mg kg^{-1}, that at Site D was 46.96 mg kg^{-1}. Among those sites under the same mode of reclamation, Site E-9 was 12.84 mg kg^{-1}, Site E-18 was 39.10 mg kg^{-1} and Site E-21 recorded 46.02 mg kg^{-1} showing an increase in

microbial biomass carbon with age. The natural forest site was highest with 63.75 mg kg^{-1}. The microbial biomass N at the study sites vary from 10.64 mg kg^{-1} to 38.99 mg kg^{-1}. Generally, the labile carbon of the study area followed the trend of the C_{mic} which increased with age of reclamation. Whereas the labile carbon of soil at Site B was 0.24 mg kg^{-1}, that of Sites C and D were 0.18 mg kg^{-1} and 0.20 mg kg^{-1}, respectively. The soils at Sites E-9, E-18 and E-21, the labile carbon occurred values were 0.14 mg kg^{-1}, 0.18 mg kg^{-1} and 0.19 mg kg^{-1}, the natural forest (Site A) was 0.30 mg kg^{-1}.

6.2.3.2 Soil dehydrogenase activity

Soil dehydrogenase activity (DHA) was highest in the control forest (236.8 µg TPF g^{-1} h^{-1}). In Site B soil, the DHA was (97.5 µg TPF g^{-1} h^{-1}). The values in the Site C soil and Site D soil were 15.6 µg TPF g^{-1} h^{-1} and 30.6 µg TPF g^{-1} h^{-1}, respectively. The DHA values were 4.35 µg TPF g^{-1} h^{-1}, 35.92 µg TPF g^{-1} h^{-1} and 80 µg TPF g^{-1} h^{-1} for Sites E-9, E-18 and E-21, respectively (Table 6.1).

6.2.3.3 Soil fauna diversity

Earthworm diversity at the study site was generally low. The diversity of earthworms at Site B was 0.89. In Site C and Site D, earthworm diversity was 0.48 and 0.57 respectively. Whereas the Sites E-9 and E-21 had no diversity, Site E-18 recorded a 0.83 diversity of earthworms. In the natural forest (Site A) earthworm diversity was 0.58.

6.2.4 Principal component analysis

In all, five individual PCs (PC1, PC2, PC3, PC4 and PC5) with eigenvalues >1.0 were selected (Table 6.2). Whereas PC-1 selected clay, available water capacity, aggregate stability, pH and total P as the highly weighted variables (bold-face values) as initial MDS, PC-2 selected C_{mic} and labile pool.

Table 6.1. Soil quality indicator parameters from the study sites at 0-20 cm depth.

Indicator[a]	Sites						
	B	**C**	**D**	**E-9**	**E-18**	**E-21**	**A**
	Physical						
Sand (%)	52 ± 0.10	80 ± 0.66	83 ± 0.20	30 ± 0.78	35 ± 1.00	25 ± 0.56	43 ± 0.40
Silt (%)	30 ± 0.40	11 ± 0.20	5 ± 0.10	34 ± 0.20	35 ± 0.40	30 ± 0.10	22 ± 0.10
Clay (%)	18 ± 0.30	9 ± 0.30	12 ± 0.10	36 ± 0.50	30 ± 0.90	45 ± 0.79	35 ± 0.60
BD (Mg m^{-3})	1.45 ± 0.05	1.95 ± 0.02	1.48 ± 0.02	1.42 ± 0.10	1.65 ± 0.05	1.4 ± 0.09	1.3 ± 0.10
AWC (%)	9.28 ± 0.20	6.34 ± 0.32	6.09 ± 0.08	12.56 ± 0.50	10.27 ± 0.35	12.81 ± 0.54	12.01 ± 0.02
MWD (mm)	2.57 ± 0.14	0.67 ± 0.05	0.44 ± 0.03	2.94 ± 0.30	1.23 ± 0.10	3.97 ± 0.15	4.16 ± 0.14
CF (%)	8 ± 0.50	45 ± 1.00	None	2 ± 0.20	5 ± 0.87	2 ± 0.50	None
	Chemical						
pH	4.89 ± 0.18	6.7 ± 0.03	6.4 ± 0.06	3.88 ± 0.04	3.72 ± 0.15	3.81 ± 0.05	3.6 ± 0.03
Ex. acidity ($cmol_c$ kg^{-1})	2.48 ± 0.03	1.81 ± 0.04	1.74 ± 0.02	3.75 ± 0.03	2.29 ± 0.03	1.92 ± 0.09	2.45 ± 0.01
CEC ($cmol_c$ kg^{-1})	6.72 ± 0.02	3.51 ± 0.02	3.83 ± 0.03	4.72 ± 0.03	3.98 ± 0.03	14.2 ± 0.05	13.04 ± 0.05
Organic C (%)	0.94 ± 0.01	0.14 ± 0.01	0.3 ± 0.04	0.39 ± 0.03	0.52 ± 0.02	0.82 ± 0.04	3.93 ± 0.10
Total N (%)	0.17 ± 0.02	0.07 ± 0.02	0.03 ± 0.01	0.03 ± 0.01	0.05 ± 0.01	0.04 ± 0.00	0.41 ± 0.01
C/N	5.58 ± 0.63	2.14 ± 0.76	11.14 ± 5.14	13 ± 1.00	10.81 ± 2.59	20.5 ± 1.00	9.588 ± 0.32
Total P (mg kg^{-1}**)**	186.67 ± 1.53	127 ± 2.00	158.67 ± 1.15	210 ± 0.04	281 ± 0.92	359.33 ± 1.15	305 ± 2.00
	Biological						
C_{mic} (mg kg^{-1}**)**	56.09 ± 0.05	37.53 ± 0.50	46.97 ± 0.40	12.84 ± 0.19	39.1 ± 0.36	46.02 ± 0.03	63.75 ± 0.20
N_{mic} (mg kg^{-1}**)**	20.37 ± 0.07	24.68 ± 0.29	38.99 ± 0.01	10.63 ± 0.35	29.83 ± 0.15	24.98 ± 0.28	35.16 ± 0.70
Labile pool (mg kg^{-1})	0.24 ± 0.03	0.18 ± 0.04	0.2 ± 0.05	0.14 ± 0.01	0.18 ± 0.02	0.19 ± 0.03	0.29 ± 0.02
DHA (µg TPF g^{-1} h^{-1})	97.53 ± 0.41	15.6 ± 0.53	30.6 ± 1.47	4.35 ± 0.10	35.92 ± 0.16	180 ± 2.00	236.8 ± 2.50
SFD	0.89 ± 0.05	0.48 ± 0.05	0.57 ± 0.04	0 ± 00	0.82 ± 0.08	0 ± 00	0.58 ± 0.09

[a)] AWC: Available water content; MWD: Mean weight diameter; CF: Coarse fragments; CEC: Cation exchange capacity; C_{mic}: Microbial biomass carbon; N_{mic}: Microbial biomass carbon; DHA: Dehydrogenase activity; TPF: Triphenylformazan; SFD: Soil fauna diversity (earthworms).

However, PC-3 selected exchangeable acidity and C: N. In PC-4 only coarse fragment was selected while PC-5 selected both bulk density and soil fauna diversity. Under PCs 1, 2, 3 and 5, the PCA retained more than one variable. Pearson correlation matrix (Appendix 5) retained the non-correlated parameters under PCs 1, 2, 3 and 5 as important MDS. Whereas the PCA selected aggregate stability, total P, C_{mic}, exchangeable acidity, C: N, coarse fragments, bulk density and soil fauna diversity as the final MDS (bold-faced and underlined), their observed values were transformed into a unitless score (0-1.00) and then integrated into SQI (see Appendix 6 and 7).

The influence of each MDS to the final soil quality index of each selected site was calculated (see Appendix 7). Results from each reclaimed site under the four modes of reclamation i.e., Site B (rehabilitated soil with conventional protocol), Site C (rehabilitated soil with a mixture of tailings, waste rocks and other overburden materials), Site D (rehabilitated soil with only tailings), and Site E (rehabilitated soil with only subsoil) were compared with Site A (natural forest or control) (Figure 6.1). Although aggregate stability, total P, C_{mic}, exchangeable acidity, C: N, coarse fragments, bulk density and soil fauna diversity were selected as the most important indicator parameters, their contributions vary from site to site. In site A the contribution of C_{mic} (0.205) was highest followed by aggregate stability (0.200) and then total P (0.163). The coarse fragments contributed 0.066, soil fauna diversity (0.017), bulk density (0.016), C: N (0.011) and exchangeable acidity (0.009) resulting in a final SQI of 0.687. The contribution of the MDS in Site B also followed order of C_{mic} (0.194) > aggregate stability (0.141) > total P (0.092) > coarse fragment (0.037) > soil fauna diversity (0.022) > bulk density (0.017) > C: N (0.016) > exchangeable acidity (0.009). This aggregated into a 0.527 SQI. That at Site C show that C_{mic} was 0.088, total P was 0.041 and C; N had 0.020. Soil fauna diversity was 0.014, exchangeable acidity had 0.013, aggregate stability was 0.011, bulk density was 0.009 while coarse fragment contributed

0.00. Site C had a final SQI of 0.197. In Site D the contribution of the MDS show that C_{mic} (0.114) was highest followed by coarse fragment (0.066). The total P also contributed 0.064 while C: N, soil

Table 6.2. Results of PCA of soil indicators at the reclaimed sites and the natural forest.

Principal components	PC1	PC2	PC3	PC4	PC5	PC6	PC7
Eigenvalues*	**9.25**	**5.23**	**1.86**	**1.22**	**1.01**	0.43	2.85687E-3
Variance	48.7	27.52	9.78	6.43	5.32	2.25	0.00E+00
Cumulative variance	48.7	76.22	86	92.43	97.75	100	1.00E+00
Soil indicators[P]	**Eigenvectors**[*]						
Sand	-0.285	0.182	0.065	-0.024	0.252	0.008	-0.202
Silt	0.215	-0.216	-0.253	0.121	-0.410	0.208	-0.124
Clay	**0.306**	-0.128	0.108	-0.063	-0.081	-0.193	-0.195
BD	-0.261	-0.062	-0.030	-0.329	**-0.423**	-0.283	0.315
AWC	**0.310**	-0.131	-0.081	-0.074	-0.035	-0.083	-0.519
MWD	**0.307**	0.027	-0.089	-0.239	0.139	0.218	0.171
CF	-0.224	0.001	-0.131	**-0.605**	-0.230	-0.077	-0.348
pH	**-0.301**	0.085	0.126	-0.140	0.220	0.221	-0.154
Ex. Acidity	0.127	-0.255	**-0.446**	0.127	0.335	-0.140	-0.068
CEC	0.271	0.156	0.178	-0.305	0.019	0.208	-0.119
Organic C	0.204	0.291	-0.199	-0.074	0.149	-0.405	0.100
Total N	0.145	0.324	-0.333	-0.103	0.121	-0.238	0.106
C: N	0.223	-0.146	**0.467**	0.115	0.064	0.034	0.003
Total P	**0.293**	0.018	0.217	-0.019	-0.315	-0.177	0.300
C_{mic}	0.062	**0.410**	0.046	0.042	-0.162	0.351	0.097
N_{mic}	-0.041	0.335	0.336	0.258	-0.063	-0.490	-0.295
Labile pool	0.107	**0.400**	-0.161	0.024	0.038	0.138	0.151
Dehydrogenase activity	0.247	0.264	0.073	-0.208	-0.017	0.125	-0.215
Soil fauna diversity	-0.115	0.253	-0.281	0.416	**-0.420**	0.129	-0.260

PC: principal component.

[*] Boldface eigenvalues match the PCs examined for the index.

[P] AWC: Available water content, MWD: Mean weight diameter (aggregate stability), CF: Coarse fragments; CEC: Cation exchange capacity, C_{mic}: Microbial biomass carbon, N_{mic}: Microbial biomass nitrogen.

[*] Boldface eigenvectors are the highly weighted; boldface–underlined factors match the parameters included in the index.

fauna diversity, bulk density, exchangeable acidity, aggregate stability occurred in respective of 0.019, 0.016, 0.014, 0.012 and 0.004 with final SQI of 0.31. However, Site E-9 the aggregate

stability was 0.113, total P was 0.107 and the coarse fragment was 0.064. Whereas bulk density contributed 0.015, $C_{mic,}$ C: N, exchangeable acidity contributed 0.013, 0.007, 0.003. Soil fauna diversity contributed 0.00. The final SQI of 0.322. At Site E-18 the results show that total P (0.151) > C_{mic} (0.124) > coarse fragment (0.053) > aggregate stability (0.043) > soil fauna diversity (0.021) > bulk density (0.011) > exchangeable acidity (0.010) = C: N (0.010) with final SQI of 0.424. The contributions of the MDS in Site E-21 also occurred in the following order aggregate stability (0.196) > total P (0.182) > C_{mic} (0.152) > coarse fragments (0.037) > bulk density (0.014) > exchangeable acidity (0.007) > C: N (0.002) > soil fauna diversity (0.00). Site E-21 had an aggregate of 0.589 SQI.

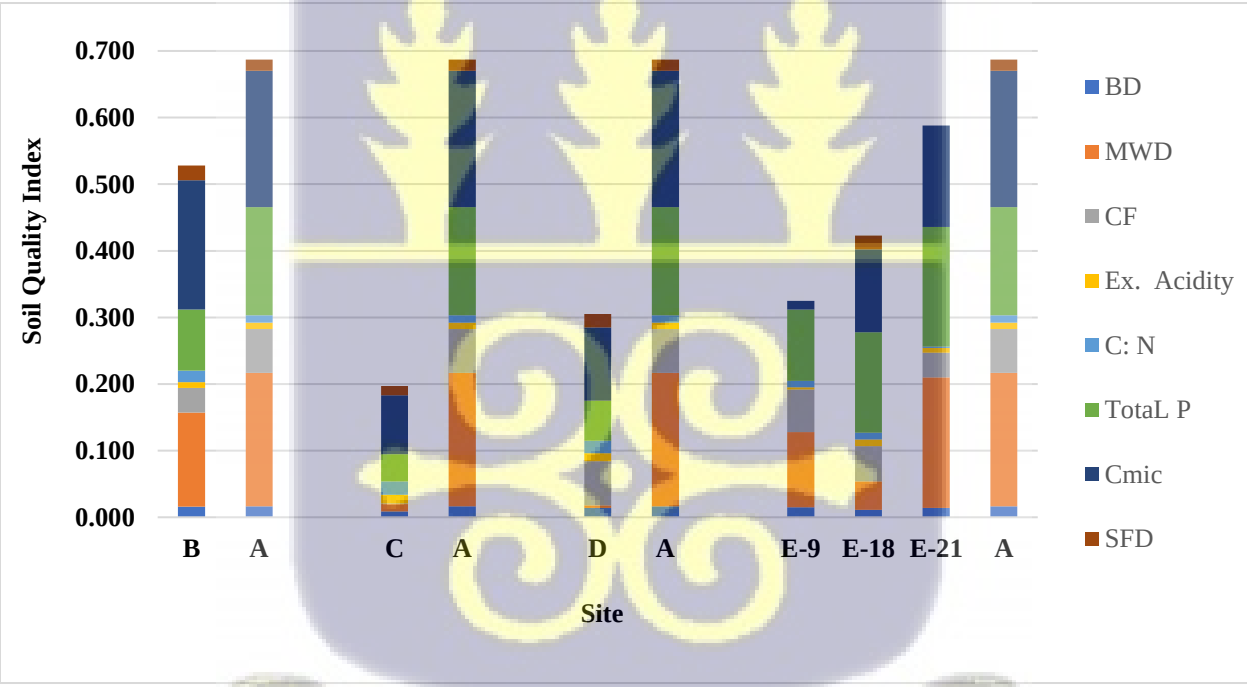


Figure 6.1. Influence of each MDS to final soil quality index (SQI).

Table 6.3. SQI values for the selected sites.

Site	A (control)	B	C	D	E-9	E-18	E-21
SQI	0.687	0.527	0.197	0.310	0.322	0.424	0.589

6.3 Discussion

6.3.1 Physical indicators

6.3.1.1 Particle size distribution and texture

The sand content at Sites B, C and D were higher than E-9, E-18, E-21 and A (natural forest). High sand content of B1 Pit was due to the fact that the pit was reclaimed with stockpiled topsoil. Long exposure of the stockpiled materials to the mercy of the intense rainfall of the area, resulted in washing out the very fine particles from the heaped soil making the sand content high. On the other hand, Sites C and D were basically tailings which were derived from milled rocks. The tailing material is not natural soil rather waste product that remains after beneficiation of gold from the milled ore. In Sites E-9, E-18 and E-21, the clay contents were higher than sand. These sites were reclaimed with materials excavated from the non-plinthic B and Bt horizons. The B and Bt horizons are noted for their high clay accumulation. The Site A was clay loam due to intense weathering activities of the area.

6.3.1.2 Bulk density

Except for Site D, the bulk densities of the reclaimed soils (Sites B, C, E-9, E-18 and E-21) were higher than their optimum bulk density values for their respective textural classes. High bulk density among these reclaimed soils was generally attributed to compaction associated with heavy mining machinery used for the reclamation program and low organic input of the soils.

In the Site C soil, the 1.95 Mg m^{-3} recorded was extremely high for loamy sand soils which could severely affect root growth. The soil at Site C was reclaimed from a mixture of tailings, waste rocks and other overburden materials. The use of heavy mining equipment to dump and spread these materials, heavily compacted the soil in the process resulting in high bulk density. This high

bulk density is also ascribed to the high sand content of the soil and the fact that the waste rocks in the reclamation material exert weight on the underlying.

The 1.48 Mg m^{-3} bulk density of soil at Site D fell below the 1.60 Mg m^{-3} (USDA-NRCS, 2008) optimum value for loamy sand soils. Though sandy, it had low bulk density due to no or zero impact of compaction from heavy mining equipment. The OTSF was reclaimed using the spot planting approach which invariably eliminates the impact of heavy machinery.

The bulk density of the natural forest (Site A) which was 1.30 Mg m^{-3} fell below optimum value (1.40 Mg m^{-3}) for clay loams (USDA-NRCS, 2008). This performance was due to good soil texture, structure, and high organic matter content in the soil. Moreover, the natural forest is characterized by trees and shrubs with varied rooting systems that are capable of including organic matter and perforate to loosen up the soil, thus reducing bulk density over time.

Bulk density values varied within and among the four modes of reclamation and were generally higher than those of the natural forest. These variations might be attributable to differences in reclamation strategy, soil texture, level of organic input, and soil compaction. Higher bulk density in the reclaimed sites relative to the natural forest was mainly due to heavy mining equipment used for excavating and replacing top and subsoil material in the reclamation process coupled with low levels organic matter on the reclaimed sites.

6.3.1.3 Aggregate stability and AWC

The aggregate stability of soils from the study area was generally high. In the natural forest, the stability of soil aggregate was the highest due to high organic matter content, good soil structure and high content of clay. Among the reclaimed sites, the stability of soil aggregate at Site B (2.57 mm) was high due to its relatively high organic matter and good soil structure. The Site C soil

(0.67 mm) and Site D (0.44 mm) recorded the lowest aggregate stability due to their poor structure, high sand contents and low amount of organic input. The high aggregate stability at Site E-9 (2.94 mm), Site E-18 (2.94 mm) and Site E-21 (3.97 mm) may be attributed to the high clay content of these soils.

Compared to the other reclaimed sites, the available water capacity soils at Sites B, C and D were relatively low. This is probably due to high bulk density, high sand content and low amount of organic input of these soils. However, in the Sites E soils, the AWC were high and increased according to their clay content. The AWC of the natural forest was high. This is attributable to the high clay content, low bulk density and high organic matter content of the forest soil. While the soils at Sites B, C, and D were typically sandy soils, the Site E and Site A soils were clayey. Generally, clay soils have greater total pore size distribution and therefore can retain more water. The available water capacity of soils is strongly affected by texture, organic matter and compaction (Ratliff et. al., 1983; da Silva and Kay, 1996).

6.3.1.4 Coarse fragments

Coarse fragment is an influential soil physical property that affects the output of degraded mined soils since excessive number of coarse fragments reduces the volume of fine earth for root proliferation and anchorage, moisture-holding capacity, and nutrient availability (Rodrigue and Burger, 2004; Maiti, 2013; Mukhopadhyay et al. 2013).

Highest coarse fragment (>2 mm size) was found at Site C (<45%) while Site D and Site A had no coarse fragments. In the other reclaimed sites, coarse fragment contents ranged from 2% to 8%. This variation is due to different modes of reclamation, type and heterogeneous nature of mine soil used for the reclamation. The high coarse fragments found at Site C was due to the presence of waste rocks and other overburden materials in the reclamation materials. On the other hand, Site

D had no coarse fragments since the reclamation material was entirely tailings. The natural forest (Site A) also had no coarse fragments within the first 20 cm. The topsoil of the natural forest is characterized with accumulation of highly decomposed organic material and the fact that weathering activity in the study area is high. Daniels and Zipper, (1997), found that in an undisturbed soil, reduction of coarse fragments with time occurs in surface horizons due to weathering processes.

6.3.2 Chemical indicators

6.3.2.1 Soil pH

Generally, soils from the study area were extremely acid to strongly acid except for Site C and Site D which were neutral. The observed data is in agreement with Tiimub et al. (2020) who reported a pH of extremely acid to very strongly acid in Ajopa, an adjacent concession to Iduapriem. This is due to a significant amount of exchangeable acidity present in these soils and the fact the Iduapriem area falls within the high rain forest zone with its characteristic acid soils.

However, the Site C and Site D were neutral pH due to the fact that soils from these two sites were tailings, a product of milled rocks. Tailings and waste rocks have been reported to have high pH depending on the geochemistry and the mineralogy of the ore. Since they are not natural soils and very young, their duration of exposure to the wetting effects of the high precipitation of the area is minimal compared to the other sites.

Comparing each reclaimed site with the natural forest, a general decreasing trend in pH towards the natural forest was observed. This insidious trend is primarily due to long period of exposure of soils to precipitation which then caused increased leaching of base cations and lowered the soil pH. The trend of reclaimed mine soils having higher pH than the natural forest as observed in this

study, was also reported by Ganjegunte et al. (2009), Juwarkar et al. (2010) and Tiimub et al. (2020). However, Ganjegunte et al. (2009) had attributed the high pH values in reclaimed sites relative to the natural forest to the effects of anthropogenic activities.

6.3.2.2 Total organic carbon, total N and total P

The total organic carbon content in the study area varies considerably among the selected sites. This variation may be due to the mode of reclamation, age of reclamation, materials used for the reclamation and the type of vegetation established on the sites.

Ghosh et al. (1983) had reported that total organic carbon greater than 0.75% indicates good fertility. Generally, TOC were less than 0.75% indicating low fertility status. Low level of TOC in Sites C and D were due to the fact that the tailings were product of milled rocks devoid of natural soil and that they are very young and impoverished.

On the other hand, low levels of TOC at Sites E-9 and E-18 were due to the impoverished nature of the soil since non-plinthic B-horizon was used in place of topsoil during the reclamation process and the fact that these soils have been severely damaged. The highest TOC obtained at Site E-21 was due to the advanced age of the reclaimed trees resulting in accumulation of leaf litterfall and its decomposition to form humus.

The TOC at Site B was attributed to the fact that vegetation (debris) was incorporated in the soil during reclamation and the fact that vegetation established on this site are those with low lignin content hence high decomposition rate. Moreover, a striking feature in organic carbon content is apparent. The Site B reclaimed site, even though 8 years old, had a relatively higher value than the other sites. Records indicated that the reclamation of this site followed conventional protocol as a

result the ecological equilibrium for organic matter accumulation might have been established faster than the other, older reclaimed sites that did not follow proper procedures.

The relatively high nitrogen contents recorded at Site A and Site B, were due to the high organic carbon recorded by these sites. This trend corroborates the findings of Agus et al. (2014) who noted that organic carbon and total nitrogen are strongly correlated since much of soil nitrogen is derived from the organic matter. Although leguminous plants were established on the reclaimed sites, their contributions to nitrogen were negligible due to the acidic nature of the soils. In the works of Hart et al. (2013), the authors reported that leguminous plants are very sensitive to soil acidity because soil acidity inhibits N fixation by bacteria of the genus rhizobium. These bacteria require high soil levels of Ca and Mo, which are limited at low pH. They also asserted that under acidic soil conditions, legume roots may have few nodules or their nodules may be ineffective at N fixation.

Phosphorus (P) plays important role in plants metabolism and energy dynamics (Weil and Brady, 2017). Zublena, (1997), underscores the crucial role of P in converting solar energy into the chemical energy desired for synthesizing starches, sugars and proteins. However, the total amount of phosphorous in the soils from the study area were low. The amount of P recorded at Site A (natural forest) and the Sites B, C, D and E (reclaimed sites), were rated low according to the ratings by Blakemore et al. (1987). Low level of total phosphorus in these soils are attributed to low P bearing rocks and parent materials. Low phosphorus availability is linked to the low pH of the soils from the study area since in acidic soils, phosphorus is bound to iron (Fe) or aluminun. Weil and Brady, (2017) had earlier attributed low total P level in soils to long period of intense weathering. The soils from the study area are highly weathered soils.

6.3.2.3 Exchangeable acidity (Al+H) and CEC

The exchangeable acidity in the soils from the study area were high. In Sites A, B, C, D and E, the amount of exchangeable acidity observed were high according to the ratings by Funakawa et al. (2016) and Onwuka et al. (2016). High exchangeable acidity of these sites is mainly attributed to the excessive precipitation in the area. Again, the generally high clay contents of soils from the study area, with the exception of Sites C and D might have accounted for this high exchangeable acidity. Weil and Bray (2017) had earlier attributed high exchangeable acidity in soils to high clay content.

CEC is often used to quantify the fertility and nutrient retention capacity of soils. The CEC of the soils from the study area was generally low. Whereas Site A and Site E-21 soils had moderate CEC, Sites B, C, D, E-9 and E-18 soils had low CEC. The presence of clay and soil organic carbon together, were very important in increasing the CEC values of soil from the study sites. This was evident in Site A and Site B that had high contents of clay and soil organic carbon. However, Sites B, C and D are sandy loam, loamy sand and loamy sand, respectively and hence had few exchange sites for cations. These Sites had low amounts of clay and a relatively low organic carbon content, resulting in very few exchange sites. Although Sites E-9 and E-18 are clay loam textured soils, they recorded low levels of CEC due to their relatively low organic carbon content. The generally low CEC of all the soils might be attributed to high level of precipitation culminating in leaching the cations and the fact that there is low activity clay of these soils.

6.3.3 Biological indicators

6.3.3.1 Microbial biomass carbon (C_{mic}) and nitrogen (N_{mic})

Soil microbial biomass carbon (C_{mic}) is a source of plant nutrients in mine soils (Singh et al. 1989), and its concentration increases with age of reclamation. Generally, C_{mic} of the study area was low.

A relatively higher C_{mic} observed in Site B (56.99 mg kg^{-1}) came close to the forest soils (Site A) with C_{mic} of 63.75 mg kg^{-1}. During reclamation process, Site B received direct-returned forest topsoil, which increased sharply the C_{mic} to reach levels equivalent to the forest soils after the eight years of rehabilitation. The speedy development of C_{mic} at Site B is an indication of faster recovery of soil microbial communities as well as the efficient restorative potential of the established vegetation.

Sites C and D with a relatively low C_{mic} of 37.5 mg kg^{-1} and 46.96 mg kg^{-1}, respectively, might be attributed to poor site conditions, scant ground vegetation, and low tree density. At Site E, a general pattern of C_{mic} increasing from a lower level at Site E-9 (12.84 mg kg^{-1}) to a higher level at Site E-18 (39.10 mg kg^{-1}) and at Site E-21 (46.02 mg kg^{-1}) year of reclamation was noted. A steady rise in C_{mic} with age in our study implies that degraded mined soils are continually regenerating.

Our results indicate that C_{mic} in all reclaimed sites was lower than the natural forest. This suggest that disturbances delay the onset of microbial biomass accumulation in the rehabilitated sites but it may not restrict the ultimate level reached. Clearly, the younger sites are in a period of development, and it will take time for their soil nutrient levels to achieve parity with those of surrounding natural forests. The microbial biomass N of the study area was low which ranged from 10.64 mg kg^{-1} to 38.99 mg kg^{-1}. The low N_{mic} of the study area may be attributed to low organic matter since much of soil N_{mic} was derived from organic matter.

6.3.3.2 Labile carbon

The labile carbon at the study area was generally low. This is the manifestation of low C_{mic} since it increases rapidly as a result of the buildup of labile carbon during reclamation. An increase in labile carbon typically leads to an increase in microbial biomass carbon. The labile carbon of the study area followed the trend of the C_{mic} which increased with age of reclamation.

6.3.3.3 Soil dehydrogenase activity

Generally, the soil dehydrogenase activity of the study area was low which ranged from 4.35 µg TPF g^{-1} h^{-1} to 236.8 µg TPF g^{-1} h^{-1}. DHA was highest in the natural forest due to higher vegetation cover and higher organic matter content of the forest soil (Site A). In Site B soil, DHA was (97.5 µg TPF g^{-1} h^{-1}) whereas that of Site C and Site D were (15.6 µg TPF g^{-1} h^{-1}) and (30.6 µg TPF g^{-1} h^{-1}) respectively. A relatively higher DHA value at Site B was because of relatively higher content of soil organic carbon. However, low DHA values of Sites C and D were due to low organic input of the tailings. At Site E, DHA occurred in order of Site E-21 (180 µg TPF g^{-1} h^{-1}) > Site E-18 (35.92 µg TPF g^{-1} h^{-1}) and the least at Site E-9 (4.35 µg TPF g^{-1} h^{-1}). Although the DHA activity at Site E was low, it increased with reclamation age as plant cover, organic matter, and microbiological activity increased in older sites.

The generally low DHA values recorded the by reclaimed sites demonstrates that disturbance due to mining and reclamation activities have decreased microbial activity and that recovery requires time. However, aside from the age of reclamation, the nature and properties of the soil for reclamation and the type the plant species and litter quality will impact carbon buildup and, by extension, dehydrogenase activity.

6.3.4 Soil quality index

The Soil quality index (SQI) values for the reclaimed sites under four modes of reclamation were compared with the natural forest (Figure 6.1). The contribution of each soil indicator parameter to SQI is also shown to give an insight into the cause for the measured SQI.

The SQI for Site B was 0.527 compared to the 0.687 index value for Site A (natural forest). Although Site B was 8 years old, it was close to the natural forest in terms of quality which

indicates a faster recovery of soil. This faster recovery of soil is attributable to the fact Site B was reclaimed following the conventional protocol of reclaiming degraded mined site.

In Site C, the 0.197 SQI observed was very low because of the lower scores for all eight MDS included in the SQI. With the exception of microbial biomass carbon (C_{mic}), the contributions of the remaining parameters to the calculated SQI were very low due to the impoverished nature and the haphazard way of packing the reclamation materials. This site was reclaimed haphazardly using a mixture of tailings, waste rocks and other overburden materials which contravenes the conventional protocol of reclaiming degraded mined land.

A generally lesser score was observed in Site D for all the eight parameters included in the SQI which culminated in a low 0.310 quality index value. The low SQI observed was due to the impoverished nature of the reclamation materials. This site was reclaimed with only tailings devoid of natural soil material following the unconventional protocol. From the study, the contributions of aggregate stability, Ex acidity, C: N and BD to the quality index of Site D were abysmally low but that of coarse fragments, total P, microbial biomass carbon and soil fauna diversity were rather substantial.

At Site E, the values of SQI increased with increase in age of reclamation which occurred in the following order: Site E-21 (0.589) > Site E-18 (0.322) and Site E-9 (0.310). The index values for Sites E-9 and E-18 were low and very close despite wide age margin. Although the reclamation procedures at Site E did not follow the conventional protocol, hence accounting low index values at Sites E-9 and E-18, the SQI at Site E-21 was substantial. The SQI recorded at Site E-21 (0.589) was close to Site A (natural forest) (0.687), which shows the soil recovery over time. According to Anderson et al. (2008), it often takes 20 years or more for disturbed mined soils in semiarid

areas to reach the level of native soils. Although the study area is in the humid region, the results from Site E-21 seem to follow the trend in the semiarid region. However, the recovery process has been slow among all three soils in Site E due to the impoverished nature of the reclamation materials which is worsened by the haphazard approach adopted in reclaiming these soils.

Except for Site B which was reclaimed following the conventional protocol of reclaiming degraded mined soils, Sites C, D, E-9, E-18 and E-21 were done haphazardly which consequently affected scores of their respective parameters and the overall SQI. There was a general low score for all the parameters included in the SQI for these sites. Although SOC was not selected as MDS, it has a direct and indirect influence on a number of selected MDS such as bulk density, aggregate stability, pH, total P and C_{mic}. Chaudhuri et al. (2013) describe it as the most influential soil quality indicator whose buildup over time indicates the recovery and quality of the soil, was almost deficient in Sites C, D, E-9 and E-18. Therefore, the very low content of SOC observed among these sites with respect to their ages, is a clear manifestation of their low SQI. On the other hand, the substantial amount of SOC observed at Sites B, E-21 and A (natural forest), might have contributed to their high SQI.

6.4 Conclusion

The AngloGold Ashanti Iduapriem Mines adopted four modes of reclamation viz (i) Site B; (ii) Site C; (iii) Site D and (iv) Site E – E-9, E-18 and E-21. Except for Site B, the Mines did not follow the conventional protocol for reclaiming Sites C, D, E-9, E-18 and E-21. They were done haphazardly indicating that the mining company did not adhere to the guidelines set out by the EPA on reclamation of degraded mined soils. The PCA selected the most important minimum data set (MDS) for the computation of the soil quality index which included physical, chemical and

biological properties. Therefore, the PCA based model produced a more realistic index value which could be used to assess the quality of mined soils elsewhere. Among the different modes of reclamation, the conventionally reclaimed site (Site B), comparatively performed better than those sites reclaimed haphazardly regardless of the reclamation duration. However, within the same mode of reclamation (Site E), SQI increased with age of reclamation. It is therefore concluded that the mode of reclamation and the materials used are more influential than age of reclamation.

CHAPTER SEVEN

GENERAL SUMMARY, CONCLUSIONS AND RECOMMENDATIONS

7.1 Summary

The study was conducted to evaluate the quality of six mined sites under four different modes and varied ages of reclamation. The reclaimed sites, which were categorized into four, consisted of: (i) an 8-year-old rehabilitated soil which followed the conventional protocol on reclamation of mined soils; (ii) a 7-year-old soil reclaimed with tailing mixed with waste rocks and other overburden materials; (iii) a 20-year-old soil reclaimed with only tailings; and (iv) three soils of varying ages (9-, 18- and 21-year-old) with the same mode of rehabilitation (profiles packed with stored subsoil materials only) were compared with the Neung natural forest as the control site.

In chapter four, inventory of plant species and earthworms were taken from each reclaimed site and the control site for biodiversity estimation. The floristic composition showed that a total of 2225 plants were identified. The natural forest (Site A) recorded the highest Shannon-Wiener diversity index value of 3.13. This was followed by Site B with an index value of 2.64. At Site C the Shannon-Wiener diversity index was 1.19, while Site D had 1.62. However, among Site E, the Shannon-Wiener diversity index occurred in the following order: 1.23 (E-9) <1.76 (E-21) <1.82 (E-18). A total number of 954 individual earthworms were collected from the study area. A population of 30 individual earthworms were collected from the natural forest. However, 540 earthworms were collected from Site B, 43 from Site C and 129 from Site D. Within the same mode of reclamation, it was 6 in Site E-9, 5 in Site E-18 and 201 in Site E-21. However, the Shannon-Wiener earthworm diversity index was in order of Site B (0.89) > Site E-18 (0.82) >Site A (0.58) > Site D (0.57) > Site C (0.48). however, Sites E-18 and E-21 had 0 diversity index for each site.

In chapter five, soil and core samples were collected at 10 cm interval from a profile at each site. Each profile was described for their morphological characteristics, the collected soil samples were examined for their physicochemical properties. Morphologically, except for Site A and Site B, that had varied soil colour, structure, consistency (dry, moist, wet) and coarse fragments, in their profiles, the other reclaimed sites had these in uniform. However, no carbonate was detected in all profiles. The physical analysis of the soil samples indicated that, the texture of the natural forest soil (Site A) was clay. In the reclaimed sites the texture was sandy clay loam at Site B, sandy loam at Sites C and D. Whereas Sites E-9 and E-18 were clay, E-21 was clay loam. The results from physical analysis also revealed higher bulk densities and poor soil structure within profiles of the reclaimed sites rather than the natural forest. However, except for Site C and Site D, the remaining reclaimed sites (Sites B, E-9, E-18 and E21) and the natural forest (Site A), had great aggregate stability and moisture content. The chemical analysis also showed that except for Sites C and D that had a neutral pH, Sites B, E-9, E-18, E-21 and Site A (natural forest) soil were in strongly acid to extremely acid conditions. The natural forest recorded the highest soil organic carbon and total nitrogen. In the reclaimed sites, the concentration of these properties occurred in the following order: Site B > Site E-21 > Site E-9 > Site E-18 > Site D > Site C. However, both reclaimed sites and natural forest had low CEC and high exchangeable acidity.

In chapter six, after the laboratory analysis, the whole dataset of soil parameters for all six reclaimed sites and the control site was subjected to a statistics-based model (PCA) to develop individual PCs and to determine the most important soil attributes (MDS). From the study, the PCA selected bulk density (BD), aggregate stability, coarse fragments (CF), exchangeable acidity, total P, C: N, microbial biomass carbon (C_{mic}) and soil fauna diversity (SFD) as the most appropriate indicators to estimate SQI. The results show that, SQI for the natural forest was 0.687.

In the reclaimed sites, SQI was 0.527 at Site B, 0.197 at Site C, and 0.310 at Site D. Among the same mode of reclamation (Site E), the value of SQI increased with increase in age of reclamation which occurred in the following order: Site E-21 (0.589) > Site E-18 (0.424) and Site E-9 (0.320).

7.2 Conclusions

The flora survey revealed higher dissimilarity between the natural forest and the reclaimed sites since the natural forest was conspicuous of native trees, whereas the reclaimed sites were more of exotic species. None of the reclaimed sites attained the 40% native species inclusion indicating a clear disregard for the 40% native species and 60% exotic species recommended by EPA-Ghana. With regard to diversity of plants and earthworms, Site B had the highest species richness, abundance, evenness and Shannon-Wiener diversity index relative to the other reclaimed sites (Sites C, D, E-9, E-18 and E-21).

The morphology and physicochemical characteristics within the profile of each reclaimed soils were highly influenced by their modes of reclamation. Unlike Site A (natural forest) and Site B (conventionally reclaimed site), that had varied soil colour, structure, consistency and coarse fragments linearly in their profiles, Sites C, D, E-9, E-18 and E-21 (haphazardly reclaimed sites), had these characteristics uniformly occurred in their profiles. The concentration of soil organic carbon and total nitrogen occurred in the following order: Site A (control forest) > Site B >Site E-21 >Site E-9 >Site E-18 >Site D >Site C. This result indicates that Site B, that was reclaimed following the conventional protocol, came close to the natural forest compared to the other reclaimed sites. In considering the vertical cross section of the individual profiles of the reclaimed soils, it is clear to conclude that the similar and uniform properties observed in the reclaimed profiles is due to cumulative impact of anthropogenic activities rather than pedogenic.

The PCA-SQI, revealed that within the same mode of reclamation (Site E), soil quality index increased with age of reclamation. However, across the different modes of reclamation, the site reclaimed with the conventional protocol, comparatively performed better than those sites reclaimed haphazardly. It is therefore concluded that the recovery of reclaimed gold mined soils is highly influenced by the mode of reclamation.

The findings of the study show that topsoil management and replacement is very crucial in the recovery of disturbed lands. However, the topsoil from the study sites was not properly managed and used by the AngloGold Ashanti Iduapriem Mines which consequentially resulted in slow recovery of the reclaimed sites. Among the four modes of reclamation adopted by AngloGold Ashanti Iduapriem Mines, it was only Site B that followed the recommended protocol of reclaiming degraded gold mined soils. The remaining reclaimed sites viz Site C, D and E were reconstructed haphazardly. The effects of this breach of the recommended protocol translated into their low performance.

7.3 Recommendations

Based on the results obtained from the study, the following recommendations were made:

1. The regulatory institutions, Civil Society Organizations and all stake holders should put pressure on the mining company to ensure strict adherence and compliance to the environmental standards so that the 40% native and 60% exotic species recommended by EPA-Ghana would be followed to restore the natural ecology of the mined sites. However, more attention should be given to the native species to ensure their survival and performance on the mined sites.
2. Post planting maintenance activities must be routinely carried out by the mining company on the rehabilitated sites to establish the survival rate of native species while checking the

invasive exotic species. A more scientific approach in this case ArcGIS for site inspection, assessments, and remedial investigation should be used instead of the usual walk through.

3. Proper topsoil (A-horizon) should be used instead of the non-plinthic B-horizon. This will help bring back the native plant species, the soil microfauna and the organic matter status of the soil.
4. The plantation managers must add biogeochemical restoration of the soils to that of management of the vegetation.
5. The Principal Component Analysis should be employed as a model to assess the quality of degraded mined soils since it has the capacity to integrate biogeochemical properties into a meaningful or realistic SQI values
6. The mining company, should follow the conventional protocol set out by the EPA to reclaim degraded mined soils instead of the haphazard way. This will facilitate a faster recovery of the degraded mined lands.

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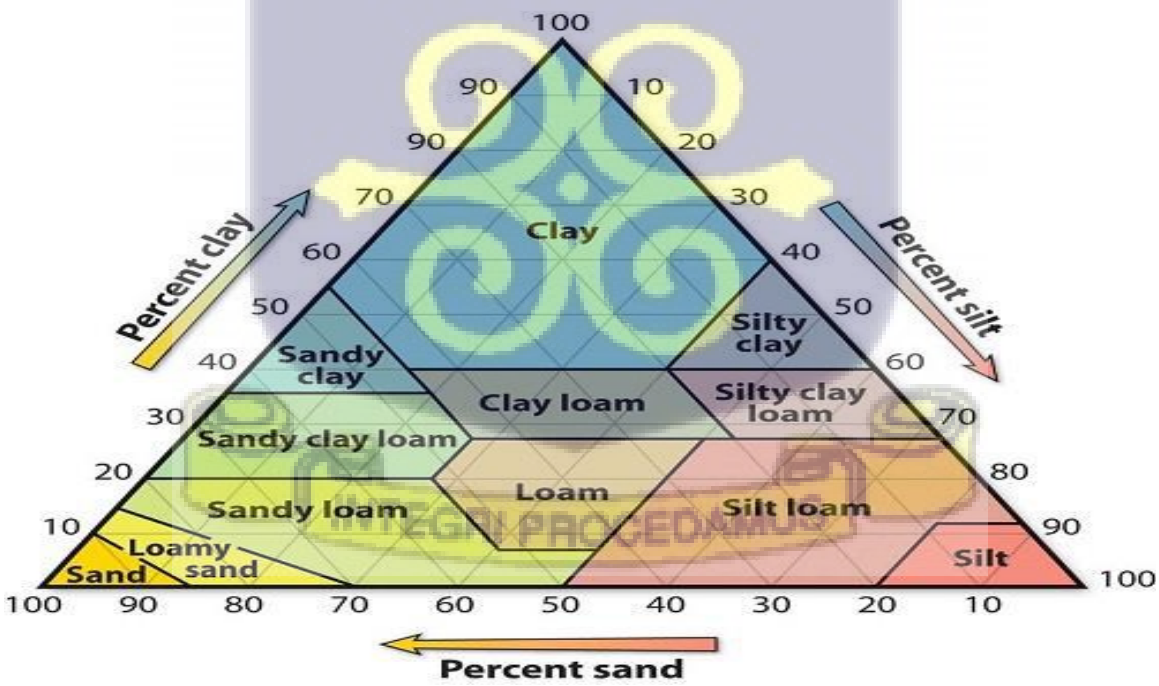
APPENDICES

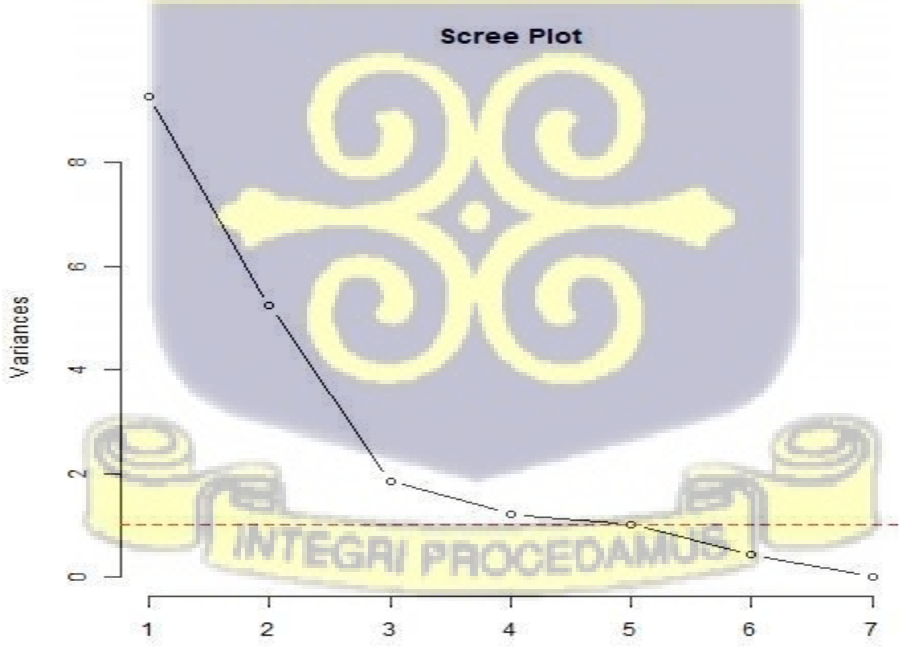
Appendix 1: Waste rocks found in Site C (ITSF).

Appendix 2: Textural triangle used for the texture class determination.



Appendix 3: Thick litter floor at Site A (natural forest).

Appendix 4: Selected principal components with Eigenvalue > 1.0.



Appendix 5: Correlation between the highly loaded variables of the principal component analysis.

	Sand	Silt	Clay	BD	AWC	MWD	CF	pH	Ex. Acidity	CEC	Organic C	Total N	CN	Total P	C_{mic}	N_{mic}	Labile pool	DHA	SFD
Sand																			
Silt	**-0.913**																		
	-0.004																		
Clay	**-0.936**	0.711																	
	-0.002	*-0.074*																	
BD	0.527	-0.334	-0.62																
	-0.224	*-0.463*	*-0.137*																
AWC	**-0.959**	**0.8**	**0.963**	-0.646															
	-0.001	***-0.031***	***(<.001)***	*-0.117*															
MWD	-0.752	0.549	**0.822**	-0.735	**0.883**														
	-0.051	*-0.202*	***-0.023***	*-0.06*	***-0.008***														
CF	0.536	-0.387	-0.59	**0.899**	-0.557	-0.478													
	-0.215	*-0.391*	*-0.164*	***-0.006***	*-0.194*	*-0.278*													
pH	**0.952**	**-0.848**	**-0.91**	0.628	**-0.943**	**-0.772**	0.639												
	-0.001	***-0.016***	***-0.004***	*-0.131*	***-0.001***	*-0.042*	*-0.123*												
Ex. Acidity	-0.55	0.618	0.414	-0.375	0.587	0.393	-0.322	-0.531											
	-0.201	*-0.139*	*-0.356*	*-0.407*	*-0.166*	*-0.383*	*-0.481*	*-0.22*											
CEC	-0.532	0.246	0.705	-0.627	0.663	**0.875**	-0.391	-0.569	-0.091										
	-0.219	*-0.594*	*-0.077*	*-0.132*	*-0.104*	***-0.01***	*-0.385*	*-0.182*	*-0.846*										
Organic C	-0.246	0.062	0.37	-0.561	0.431	0.658	-0.339	-0.477	0.079	0.678									
	-0.595	*-0.894*	*-0.414*	*-0.19*	*-0.334*	*-0.108*	*-0.456*	*-0.279*	*-0.867*	*-0.094*									
Total N	-0.08	-0.008	0.144	-0.418	0.256	0.537	-0.162	-0.314	0.052	0.536	**0.957**								
	-0.865	*-0.987*	*-0.758*	*-0.35*	*-0.579*	*-0.214*	*-0.729*	*-0.492*	*-0.912*	*-0.215*	***-0.001***								
C:N	-0.66	0.385	**0.807**	-0.596	0.656	0.515	-0.679	-0.581	0.108	0.558	0.019	-0.248							
	-0.106	*-0.394*	***-0.028***	*-0.158*	*-0.11*	*-0.237*	*-0.093*	*-0.171*	*-0.818*	*-0.193*	*-0.967*	*-0.592*							
Total P	**-0.812**	0.575	**0.904**	-0.563	**0.815**	0.745	-0.568	**-0.842**	0.041	**0.809**	0.486	0.271	**0.755**						
	-0.026	*-0.177*	***-0.005***	*-0.188*	***-0.026***	*-0.055*	*-0.183*	***-0.017***	*-0.93*	***-0.028***	*-0.269*	*-0.557*	*-0.05*						
C_{mic}	0.19	-0.257	-0.107	-0.277	-0.12	0.226	-0.144	0.01	-0.582	0.518	0.635	0.689	-0.144	0.25					
	-0.683	*-0.579*	*-0.82*	*-0.547*	*-0.798*	*-0.626*	*-0.758*	*-0.983*	*-0.171*	*-0.233*	*-0.125*	*-0.087*	*-0.758*	*-0.588*					
N_{mic}	0.444	-0.599	-0.248	-0.045	-0.403	-0.255	-0.154	0.239	-0.726	0.139	0.359	0.314	-0.026	0.105	0.673				
	-0.319	*-0.155*	*-0.592*	*-0.923*	*-0.371*	*-0.581*	*-0.742*	*-0.605*	*-0.065*	*-0.766*	*-0.429*	*-0.493*	*-0.956*	*-0.823*	*-0.098*				
Labile pool	0.089	-0.163	-0.012	-0.422	0.049	0.399	-0.211	-0.14	-0.267	0.545	**0.849**	**0.907**	-0.217	0.24	**0.921**	0.535			
	-0.85	*-0.727*	*-0.979*	*-0.345*	*-0.917*	*-0.375*	*-0.649*	*-0.765*	*-0.563*	*-0.205*	***-0.016***	***-0.005***	*-0.641*	*-0.604*	***-0.003***	*-0.216*			
DHA	-0.389	0.147	0.545	-0.611	0.531	**0.797**	-0.375	-0.51	-0.17	**0.948**	**0.836**	0.744	0.344	0.726	0.726	0.323	**0.776**		
	-0.388	*-0.754*	*-0.206*	*-0.145*	*-0.22*	***-0.032***	*-0.407*	*-0.243*	*-0.716*	***-0.001***	***-0.019***	*-0.055*	*-0.45*	*-0.065*	*-0.065*	*-0.479*	***-0.04***		
SFD	0.391	-0.135	-0.558	0.206	-0.486	-0.411	0.093	0.215	-0.325	-0.326	0.149	0.333	-0.641	-0.286	0.562	0.442	0.503	-0.042	
	-0.386	*-0.773*	*-0.193*	*-0.657*	*-0.268*	*-0.359*	*-0.842*	*-0.644*	*-0.477*	*-0.475*	*-0.749*	*-0.466*	*-0.121*	*-0.534*	*-0.189*	*-0.32*	*-0.25*	*-0.929*	

Appendix 6: Calculation of Soil quality index.

SQI = 0.498 (Aggregate stability/Total P) + 0. 282 (microbial biomass carbon) + 0.100 (Ex. Acidity/C: N) + 0.066 (Coarse fragments) + 0.054 (Bulk density/soil fauna diversity).

Final SQI = 0.245 × *S*(Aggregate stability) + 0.245 × *S*(Total P) + 0.282 × *S*(C_{mic}) + 0.02 × *S*(Exchangeable acidity) + 0.02 × *S*(C:N) + 0.066 × *S*(Coarse fragments) + 0.027 × *S*(Bulk density) + 0.027× *S*(SFD).

Appendix 7: PC weight and scores of indicator parameters of the selected sites.

PC weight and scores of indicator parameters of the soil from Site A (NF)

Site	Indicator	PC weight (W)	Score (S)	W*S
A	BD	0.027	0.605	0.016
	Ag. stability	0.245	0.818	0.200
	CF	0.066	1.000	0.066
	Ex. Acidity	0.020	0.474	0.009
	C: N	0.020	0.550	0.011
	Total P	0.245	0.663	0.163
	C_{mic}	0.282	0.726	0.205
	SFD	0.027	0.620	0.017
SQI (ΣW*S) =				**0.687**

PC weight and scores of indicator parameters of the soil from Site B

Site	Indicator	PC weight (W)	Score (S)	W*S
B	BD	0.027	0.583	0.016
	Ag. stability	0.245	0.574	0.141
	CF	0.066	0.563	0.037
	Ex. Acidity	0.020	0.466	0.009
	C: N	0.020	0.826	0.017
	Total P	0.245	0.375	0.092
	C_{mic}	0.282	0.687	0.194
	SFD	0.027	0.826	0.022
SQI (ΣW*S) =				**0.527**

PC weight and scores of indicator parameters of the soil from Site D

Site	Indicator	PC weight (W)	Score (S)	W*S
D	BD	0.027	0.526	0.014
	Ag. stability	0.245	0.016	0.004
	CF	0.066	1.000	0.066
	Ex. Acidity	0.020	0.617	0.012
	C: N	0.020	0.966	0.019
	Total P	0.245	0.262	0.064
	C_{mic}	0.282	0.404	0.114
	SFD	0.027	0.609	0.016
SQI (ΣW*S) =				**0.310**

PC weight and scores of indicator parameters of the soil from Site C

Site	Indicator	PC weight (W)	Score (S)	W*S
C	BD	0.027	0.343	0.009
	Ag. stability	0.245	0.045	0.011
	CF	0.066	0.017	0.001
	Ex. Acidity	0.020	0.657	0.013
	C: N	0.020	0.981	0.020
	Total P	0.245	0.166	0.041
	C_{mic}	0.282	0.313	0.088
	SFD	0.027	0.504	0.014
SQI (ΣW*S) =				**0.197**

PC weight and scores of indicator parameters of the soil from Site E-9

Site	Indicator	PC weight (W)	Score (S)	W*S
E-9	BD	0.027	0.552	0.015
	Ag. stability	0.245	0.460	0.113
	CF	0.066	0.976	0.064
	Ex. Acidity	0.020	0.147	0.003
	C: N	0.020	0.364	0.007
	Total P	0.245	0.437	0.107
	C_{mic}	0.282	0.046	0.013
	SFD	0.027	0.000	0.000
SQI (ΣW*S) =				**0.322**

PC weight and scores of indicator parameters of the soil from Site E-18

Site	Indicator	PC weight (W)	Score (S)	W*S
E-18	BD	0.027	0.422	0.011
	Ag. stability	0.245	0.176	0.043
	CF	0.066	0.807	0.053
	Ex. Acidity	0.020	0.516	0.010
	C: N	0.020	0.475	0.010
	Total P	0.245	0.616	0.151
	C_{mic}	0.282	0.438	0.124
	SFD	0.027	0.795	0.021
SQI (ΣW*S) =				**0.424**

PC weight and scores of indicator parameters of the soil from Site E-21

Site	Indicator	PC weight (W)	Score (S)	W*S
E-21	BD	0.027	0.518	0.014
	Ag. stability	0.245	0.800	0.196
	CF	0.066	0.563	0.037
	Ex. Acidity	0.020	0.367	0.007
	C: N	0.020	0.100	0.002
	Total P	0.245	0.734	0.180
	C_{mic}	0.282	0.540	0.152
	SFD	0.027	0.000	0.000
SQI (ΣW*S) =				**0.589**