UNIVERSITY OF CAPE COAST

HEAVY METAL CONCENTRATION IN CASSAVA CULTIVATED ON RE-VEGETATIVELY RESTORED MINED SPOILS AT AN ANGLOGOLD CONCESSION, OBUASI-GHANA

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BY

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Thesis submitted to the Department of Soil Science of the School of Agriculture, College of Agriculture and Natural Sciences, University of Cape Coast in partial fulfilment of the requirements for the award of Master of Philosophy degree in Land use and Environmental Science

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DECLARATION

Candidate's Declaration

I hereby declare that this thesis is the result of my own original research and that no part of it has been presented for another degree in this university or elsewhere.

Candidate's Signature:………………………… Date:……………………

Name: Abubakar Muhammed

Supervisors' Declaration

We hereby declare that the preparation and presentation of the thesis were supervised in accordance with the guidelines on supervision of thesis laid down by the University of Cape Coast.

Principal Supervisor's Signature:…………………. Date: ……………………

Name: Prof. Daniel Okae-Anti

Co-Supervisor's Signature:………… ………….. Date:……………………..

Name: Prof. B. A. Osei

ABSTRACT

Mining is one of the main causes of environmental pollution by heavy metals. Re-vegetation of mine spoils is an effective way of restoring the mined land to its pristine status. The study was conducted to assess the physico-chemical soil properties, heavy metal concentrations in two re-vegetatively restored mined spoils at an AngloGold concession at Obuasi. Concentration of heavy metals in cassava were used as an indicator for comparism. A simple random sampling technique was used to obtain soils at 0-15 cm depth and crop samples. Data analyses were done using GENSTAT statistical package. The results showed that the soil was slightly acidic, loamy textured, well aerated with a maximum bulk density of 1.33 Mg m^{-3}. The mean maximum concentration values for K^+, Ca^{2+}, Mg^{2+}, Na^+ in the soil; 7.39, 12.72, 12.62, and 1.57 mg kg^{-1} respectively were very low. These were attributed to leaching losses and low level of organic carbon content of the soil. Also, the mean maximum concentrations of As, Cu, Fe, Mn, Pb and Zn levels in the soil were; 0.98, 34.50, 213.45, 56.11, 3.35 and 43.04 mg kg^{-1} respectively, while Cd was below detection. Metals detected in the soil were at higher concentrations compared to values in the cassava tubers. These values detected were however below phytotoxicity levels. Further studies on heavy metal accumulation in the cassava were suggested to be made at different seasons in order to help advise farmers on the consumption pattern of the crop.

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DEDICATION

To the memory of my late father Alhassan Mohammed, Hajia Suada Mohammed, my mother, Abubakar Aasium Charmawla and the entire family

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KEY WORDS

Heavy metal

Mined spoil

Mined spoil reclamation

Phytotoxicity

Re-vegetation

Translocation factor

CHAPTER ONE

INTRODUCTION

Background to the Study

Mining is the world's second oldest and most important industry after agriculture. It is currently the fifth largest industry in the world and it plays a vital role in world economic development (Down & Stocks, 1977). However, all these benefits have been offset by considerable negative impacts on the environment and on the health and safety of mine workers and mining communities (Blinker, 1999). Increased growing public awareness of these costs and the challenges of sustainable development, societies around the globe are progressively expecting the mining industry to apply higher standards of environmental safety and community management to all projects through the application of modern technologies and management tools (Blinker, 1999).

An increase in the concerns for the environment has made the concurrent post-mining reclamation of degraded land an integral feature of the whole mining spectrum (Ghose, 1989). Conservation and reclamation efforts to ensure continued beneficial use of land resources are essential. Reclamation is the process by which derelict or highly degraded lands are returned to productivity, and by which some measures of biotic function and productivity are restored (Sheoran et al., 2010). Long term mine spoils reclamation requires the establishment of stable nutrient cycles for plant growth and microbial processes (Sheoran et al., 2010). The soil provides the foundation for this process, so its composition and density directly affect the future stability of the restored plant community. Reclamation strategies must address soil structure, soil fertility, microbe populations, top soil management and nutrient cycling in order to return the land as closely as possible to its pristine condition and continue as a self-sustaining ecosystem.

Vegetation has an important role in protecting the soil surface from erosion and allowing accumulation of fine particles. It can reverse the degradation process by stabilizing soils through development of extensive root systems. Once established, plants increase soil organic matter, lower soil bulk density, and moderate soil pH and bring mineral nutrients to the surface and accumulate them in available forms. Their root systems allow them to act as scavengers of nutrients not readily available. The plants accumulate these nutrients; and redeposit them on the soil surface through plant litter and subsequently organic matter from which nutrients are much more readily available by microbial breakdown (Sheoran et al., 2010). Plants to be used for re-vegetation must be drought-resistant, fast growing crops or fodder which can grow in nutrient deficient soils. They should also be easy to establish, grow quickly, and have dense canopies and root systems (Sheoran et al., 2010). The AngloGold Ashanti Obuasi Mine has made investments to reclaim the degraded lands at mined sites in its concession by re-vegetating with multipurpose tree species and grasses including *Acacia mangium, Senna siamea*, *Ceiba pentandra* and grasses. The re-vegetating strategies are necessary to improve the soil fertility for crop production and to restore the ecosystem function closer to pre-mining status. This study, therefore, aims to evaluate the impact of the re-vegetation strategy on some soil fertility indicators at the re-vegetatively restored mined spoils and assess the concentration of heavy metals both in soil and cassava tubers at the re-vegetated mined spoils.

Although mining-contaminated lands constitute a relatively small proportion of the total extent of degraded lands in the world, the scale of mining is increasing and the impacts are generally more severe (Kumar, 2013). Some surface mined areas and mine spoil dumps are nutritionally deprived habitats characterized by infertile soils having extreme pH values, low cation exchange capacity, low water holding capacity, and low nutrient availability (Kumar, 2013). Given the growing food insecure populations of the world (835.2 million undernourished people in the developing world), it is important to raise agricultural productivity on all types of lands including the mine-contaminated ones that are re-vegetated (FAO, 2010).

Statement of the Problem

Despite the benefits of mining, mining areas are associated with a high environmental cost including decreased access to potable water, loss of biodiversity, decline in soil fertility, degradation of agricultural lands, and increased fragmentation of communities which result in the decline in the livelihoods of individuals, groups and communities. These issues are of great concern to conservationists, ecologists, policy-makers and all environmental advocates.

Studies on heavy metal pollution in various environments and or lands within mining communities in Ghana have been undertaken during the last few decades (Hilson, 2002; Golow and Adzei, 2002; Adimado and Amegbey, 2003; Manu et al., 2004; Essumang et al., 2007; Obiri, 2007; Yidana et al., 2008; Armah et al., 2010; Nude et al., 2011). Heavy metal accumulation in soils is of concern in agricultural production due to the adverse effects on food safety and marketability, crop growth due to phyto-toxicity, and environmental health of soil organisms (Nagajyoti et al., 2010). Plants growing in metal-polluted sites exhibit altered metabolism, growth reduction, lower biomass production and metal accumulation. Various physiological and biochemical processes in plants are affected by metals. A few metals, including copper, manganese, cobalt, zinc and chromium are, however, essential to plant metabolism in trace amounts. Heavy metals when taken up by plants may enter the food chain in significant amounts. Hence, people could be at risk of adverse health effects from consuming crops grown in soils containing elevated heavy metals concentrations. For instance, it is estimated that approximately half of human lead (Pb) intake is through food, with around half originating from plants (Nagajyoti et al., 2010). Root crops such as cassava and carrot can accumulate heavy metals, e.g. Cd, Cu, Mn, Pb and Zn in their tissues (Marisa & John, 2006). Care must therefore be made in growing these plants on soils with little or trace amount of such heavy metals.

Over the years, AngloGold Ashanti Obuasi Mine has employed various reclamation techniques and resources to impact on conservation values of degraded sites in anticipation of returning some pre-disturbance functions. In fulfilling these objectives re-vegetation has since been used for reclamation purposes by the Mine. Even though the reclamation is perceived to go in parallel with mining so that lands will be made available to the resource-poor farmers after decommissioning, knowledge about the quality of the cassava crop grown on this re-vegetatively restored mined spoils is yet to be established.

Objectives of the Study

Main Objective

To assess the impact of re-vegetatively restored mined spoils on heavy metals concentration in cassava cultivated on such spoils at the AngloGold Ashanti Mine concession at Obuasi.

Specific Objectives

The specific objectives of the study were to assess:

1. The magnitude of some selected soil physical and chemical (Fe, Zn, Cu, As, Mn, Pb and Cd) properties in two re-vegetatively restored mined spoils.
2. The levels of heavy metals (Fe, Zn, Cu, As, Mn, Pb and Cd) in the cassava crop grown on the re-vegetatively restored mined spoils.
3. The relationship between heavy metal concentration in the soil and that of the cassava tubers at the re-vegetatively restored mined spoil.

Research Hypothesis

H_o: Heavy metal concentration of re-vegetatively restored mined spoils are within phytotoxic limits

H_o: Concentration of heavy metals in the cassava crop grown at the re-vegetativley restored mined spoils are within phytotoxic limits

Significance of the Study

The study will establish the effectiveness of the re-vegetation strategies employed at the mined spoils. Once the method proved to be successful, it could be used to restore similar mined spoils by mining companies. This study will also help in ascertaining the quality of cassava tubers produced at the re-vegetatively restored mined spoil/sites and give practical recommendations for the consumption of this crop.

Delimitations

Two re-vegetatively restored mined soils at AngloGold concession Obuasi, were used for this study. Levels of heavy metals in the cassava were also determined only in the tubers. Cassava peels, stems and leaves were excluded from the study.

Limitations

The study was conducted during the months of June – July, 2014. This was a rainy season and might have affect the levels of soil physical and chemical properties and some heavy metals in the soil.

Definition of terms

Bioavailability: The degree and rate at which a trace element is absorbed into a living system or is made available at the site of physiological activity.

Heavy metal: A metal of relatively high density (<5 cm^{-3}) or of high relative atomic weight

Mine spoil: All waste except for tailings generated as result of gold mining.

Phytotoxicity: Level at which a compound or trace element can be toxic to the plant

Mine Reclamation: The process of restoring land that has been mined to a natural or economically usable state.

Re-vegetation: The process of replanting and rebuilding the soil of disturbed land.

Restoration: The process involved in returning an altered, dysfunctional and or degraded landscape to a functional healthy ecosystem.

Organisation of the Study

The first chapter introduces the research conducted, describing the need for reclamation of mined land, stating the research problem, main and specific objectives to attain.

Chapter 2 is devoted to the literature review of the study. Among the topics discussed were; the impact of mining on soil quality, re-vegetation and its importance and behaviour of heavy metals in contaminated soils. The chapter concludes with a brief summary of the literature review.

In chapter 3, a brief account as well as the description of the study area and the methodological approaches carried out in order to achieve the set out objectives as well as software used for the data analysis.

The results of the study are discussed in chapter 4. The chapter states the levels of soil physico - chemical properties, concentration of heavy metals both in the soil and cassava tubers and finally the partial correlation between heavy metals in the soil and the cassava tubers.

Chapter 5 concludes the research with some recommendations whilst references and appendices are given on separate sheets after recommendations.

CHAPTER TWO

LITERATURE REVIEW

This chapter reviews literature relevant to the topic. Issues covered include the impact of mining on soil quality, chemical and physical properties of soil important for plant growth, behaviour of heavy metals in contaminated soils, soil-to-plant transfer factor, re-vegetation importance and methods used in re-vegetation**.**

Impacts of Mining Projects on Soil Quality

Surface mining is perhaps the greatest agent of land degradation, utilizing over 13% out of the 240,000 km^2 of the remaining forest in Ghana for mining activities (Awotwi, 2003). It has been widely recognized since the late 20th century that re-vegetation as a reclamation strategy is a desirable and necessary remedy ‘‘to return the mined areas to an acceptable environmental condition whether for resumption of the former land use or for a new use’’ (Redgwell, 1992), or to allow such lands to achieve their optimum economic value as much as possible (Bastida, 2002). In addition, reclamation is generally considered as an ongoing programme because of progressively growing environmental effects as mining evolves through the different stages of development (Walde, 1993). Mining is a temporary use of land and mined land reclamation is clearly justified from the perspective of sustainable development. Thus, it has become an important part of the sustainable development strategy in many countries (Gao et al., 1998).

Surface mined soils are mechanically, physically, chemically and biologically deficient, characterized by instability and limited cohesion, with low contents of nutrients and organic matter and high levels of heavy metals. Apart from the local disturbance of the physical properties, toxic metals can cause a more widespread contamination of soil, sediments and food crops leading eventually to a loss of biodiversity and a potential health risk to residents in the vicinity of the mining area (Zhang et al., 2002; Galan et al., 2003). Land is one of the most important resources on which human beings depend. The rate of exploitation and consumption of mineral resources is continuously increasing with the advancement of science and technology, economic development, industrial expansion, acceleration of urbanization and growth of population. Growth of our society and civilization thus heavily rely upon the mining industry to operate and maintain comfort. The effects for mining activities on the surface is mining wastes and alteration of land forms which are of concern to society and it is desired that the pre-mine conditions are restored (Sheoran et al., 2010).

Mining can contaminate soils over a large area. Agricultural activities near a mining project may be particularly affected. According to a study commissioned by the European Union, mining operations routinely modify the surrounding landscape by exposing previously undisturbed earthen materials. Erosion of exposed soils, extracted mineral ores, tailings, and fine material in waste rock piles can result in substantial sediment loading to surface waters and drainage ways. In addition, spills and leaks of hazardous materials and the deposition of contaminated windblown dust can lead to soil contamination (International Institute for Environment and Development, 2002).

According to Sheoran et al. (2010) human health and environmental risks from soils generally fall into two categories; contaminated soil resulting from windblown dust, and soils contaminated from chemical spills and residues. Fugitive dust can pose significant environmental problems at some mines. The inherent toxicity of the dust depends upon the proximity of environmental receptors and type of ore being mined. High levels of arsenic, lead, and radionuclides in windblown dust usually pose the greatest risk. Soils contaminated from chemical spills and residues at mine sites may pose a direct contact risk when these materials are misused as fill materials, ornamental landscaping, or soil supplements (Sheoran et al., 2010).

Chemical and Physical Properties of Soil Important for Plant Growth

Selected soil chemical and physical properties important for plants growth are discussed below.

Soil fertility

The three major macronutrients, namely nitrogen, phosphorus and potassium are generally found to be deficient in overburden dumps. All newly created mine soils, and many older ones, will require significant fertilizer element applications for the establishment and maintenance of any plant community. Organic matter is the major source of nutrients such as nitrogen, and available P and K in unfertilized soils. A level of organic carbon greater than 0.75% indicates good fertility. Organic carbon is positively correlated with available N and K and negatively correlated with Fe, Mn, Cu, and Zn. Initial applications of fertilizers have been shown to increase the specific numbers, plants co-density and growth rates of vegetation (Sheoran et al., 2010).

The impact of soil organic carbon (SOC) or soil organic matter (SOM) on other aspects of soil quality have also been widely reported with contradictory evidence as to whether increased levels have a positive or negative effect on soil function or indeed any effect at all. Loveland and Webb (2003) in a review of reported data on availability of micronutrients concluded “there is no consistent effect of SOC on crop uptake of Zn or Cu”. Higher levels of SOC are desirable from both an agronomic and environmental perspective and soil function may be adversely affected by a lack of SOC (John et al., 2009). They continued that it is however very difficult to determine at what level of SOC content, soil structure and function are adversely affected. A “trigger or threshold value” is questionable, rather a critical level for any soil will be dependant. As SOC declines towards lower levels the yield potential and workability of the soil will decline in parallel with adverse environmental performance. Whilst there is no specific level of SOC below 2% that can be used to determine soils in poor condition, observation of the functionality of the soils below 2% SOC, in terms of crop production is a useful indicator of unsustainable farming practices and prompting to adjust management or tillage practices (John et al., 2009).

Some of the important metallic micronutrients that are essential for plant growth are iron, manganese, copper and zinc. These micronutrients are available in the soil due to continuous weathering of minerals mixed with primary minerals. These metals are more soluble in acidic solution, and they dissolve to form toxic concentrations that may actually hinder plant growth (Sheoran et al., 2010). According to Lindsay and Norvell (1978), if the concentration of micronutrients in the soil was higher than 4.5 mg kg-^{1} for Fe, 1.0 mg kg-^{1} for Mn, 1.0 mg kg-^{1} for Zn and 0.4 mg kg^{-1} for Cu, the values are rated as highly sufficient for ecological sustainable reclamation.

Soil pH

Soil pH is a measure of soil reaction and is the most commonly used indicator of mine soil quality. The pH of a given mine soil can change rapidly as the rock fragments weather and oxidize. Pyritic minerals (FeS_2), when present, oxidized to sulfuric acid and drastically lower the pH, while carbonate ($Ca/MgCO_3$) bearing minerals and rocks tend to increase the pH as they weather and dissolve. Unweathered (or unoxidized) mine soils that contain a significant amount of pyritic-S in excess of their neutralizers (carbonates) will rapidly drop the pH to a range of 2.2 - 3.5 after exposure to water and oxygen. Vegetation achieves optimal growth in soil at a neutral pH. When the soil pH drops below to 5.5, reduced legume and forage growth occur due to metal toxicities such as aluminum or manganese, phosphorus fixation, and reduced population of N-fixing bacteria. The situation inhibits plant root growth and many other metabolic processes. A mine soil pH in the range of 6.0 to 7.5 is ideal for forages and agronomic or horticultural uses (Sheoran et al., 2010).

Maiti and Ghose (2005) reported that pH varied from 4.9 to 5.3 in a mining dump site and thus indicated the acidic nature of the dumps. This acidic nature arose due to the geology of the rock present in the area. It has been reported earlier that at pH less than 5, the bioavailability of toxic metals such as nickel, lead and cadmium also increases (Sheoran et al., 2010).

Soil aggregation

Soil aggregation controls soil hydrology, affects soil diffusion and the degree of nutrient availability to the soil, may reduce erosion potential and constitutes a pathway of organic carbon stabilization and long term sequestration (Heras, 2009). Aggregate structure breaks down as successive layers of soil are removed and stockpiled elsewhere on the site when mining begins.

The resulting compaction reduces water holding capacity and aeration. Macro aggregate stability is largely responsible for macroporosity, which determines soil drainage rate and aeration; it changes seasonally and is often affected by cultivation and cropping regime. Micro-aggregate stability is more resilient than macro-aggregate stability as the organic matter responsible for binding the soil particles together reside in pores too small for microorganisms to access, thus micro-aggregates are less sensitive to cropping practices than macro-aggregates and are responsible for crumb porosity which controls the amount of available water for vegetation (Sheoran et al., 2010).

Soil texture

The relative amounts of sand (2.0 - 0.05 mm), silt (0.05 - 0.002 mm), and clay (< 0.002 mm) particles determine the texture of soil. Mine soils with sandy textures cannot hold as much water or nutrients as finer textured soils like loams and silts. The silts are finer textured soils and have a tendency to form surface crusts, and often contain high level of soluble salts, and have a poor "tilth" or consistence. The particle size distribution of the soils with loamy textures is generally ideal. Silt loam textures are common where spoils are dominated by siltstones (Ghose, 2005).

Moisture, bulk Density, compaction and available rooting depth

Moisture content in a dump is a fluctuating parameter which is influenced by the time of sampling, height of dump, stone content, amount of organic carbon, and the texture and thickness of litter layers on the dump surface. During the winter, the average moisture content of 5% was found to be sufficient for the plants growth (Sheoran et al., 2010). Gravimetric soil water contents of air-dry (25 °C) mineral soil are often less than 2 per cent, but, as the soil approaches saturation, the water content may increase to values between 25 and 60 per cent, depending on soil type. Volumetric soil water content, may range from less than 10 per cent for air-dry soil to between 40 and 50 per cent for mineral soils approaching saturation. Water contents for stony or gravelly soils can be grossly misleading. When rocks occupy an appreciable volume of the soil, they modify direct measurement of soil mass, without making a similar contribution to the soil porosity. For example, gravimetric water content may be 10 per cent for a soil sample with a bulk density of 2000 kg m^{-3}; however, the water content of the same sample based on finer soil material (stones and gravel excluded) would be 20 per cent, if the bulk density of fine soil material was 1620 kg $m^{-3.}$ (Campbell & Henshall, 2001).

Bulk density of productive natural soils generally ranges from 1.1 to 1.5 Mg m^{-3}. High bulk density limits rooting depth in mine soils. Soil compaction directly limits plant growth, as most species are unable to extend roots effectively through high bulk-density mine soils. Severely compacted (bulk density > 1.7 Mg m^{-3}) mine soils, particularly those with less than 60 cm of effective rooting depth, shallow intact bedrock and the presence of large boulders in the soil simply cannot hold enough plant-available water to sustain vigorous plant communities through protracted drought. About 90 to 120 cm of loose non-compacted soil material is required to hold enough water to sustain plants through prolonged droughts (Sheoran et al., 2010).

Soil Vegetation interactions

Soil vegetation interactions is vital in determining the effects of soil on re-vegetation success in degraded mined areas. There is a variety of research available on the carbon sequestration potential of diverse types of re-vegetation projects and with those comes minor analysis of other variables such as moisture and nutrient enrichment. (Yadav et al., 2009; Wang et al., 2011; Dean et al., 2012).

From subjective and observational investigation it is clear that there is a relationship between soils and vegetation, nevertheless there is a lack of synthesis of this relationship with regards to the interactions of soil condition and re-vegetation efforts. It seems there is no attempt to measure the success of a planting based simply on the changes in soil conditions over time and it would seem that this is a necessary aspect of assessment when approaching a planting from an ecological restoration perspective as the whole ecosystem must function as it once did, abiotic factors included. Wilson (2008), studied the effect of scattered paddock trees on surface soil properties and among his major findings were that there was a significant decrease in pH, carbon, nitrogen and extractable phosphorous systematically with distance from individual sites. This indicates the potential influence vegetation can have on soil properties.

The storage of carbon within soil and decaying vegetation is a theme which has come to the forefront of scientific research recently as communities search for advanced methods of removing carbon from the atmosphere and store it in a stable form (Shrestha & Lal, 2006; Yadav et al., 2009; Batjes, 2011). A large body of literature exists on a number of variations in soil organic carbon (SOC) storage under plantations (Turner & Lambert, 2000; Turner et al., 2005; Kasel & Bennett, 2007), under native grasslands and woodlands (Jonson and Freudenberger, 2011; Pringle et al., 2011; Dean et al., 2012), the potential to store carbon in reclaimed mine sites and the effects of vegetation restoration and rehabilitation on soil carbon amounts (Wilson, 2008; Son et al., 2003; Wang et al., 2011). The potential of re-vegetated areas to support crop production has been mapped as agricultural landscapes return to native vegetation in other countries (Alberti et al., 2011), though notably absent is assessment of Ghanaian soils. The potential for the accumulation pattern to be detectable means that SOC accumulation could be a valuable indicator for the effectiveness of re-vegetation whilst also providing information about the possibility of carbon off-set schemes based on re-vegetation and restoration.

Contradiction however exists within the literature on this topic. Dean et al. (2012) discusses this through suggesting that a ‘breakeven’ point in carbon soil dynamics may occur whereby there is a net emission directly after re-vegetation, a time lapse where soil carbon turns from emission to accumulation which then reaches the ‘breakeven’ point where the soil has returned to pre-re-vegetation carbon levels and only after which net sequestration occurs. This could suggest a reason for some of the contradictions in the literature, hence sampling done before the breakeven point would show a decrease in soil organic carbon (Turner and Lambert, 2000) and those sampled afterwards would show an increase (Wang et al., 2011).

Soil moisture and nutrients

As with all things within an ecosystem, even one which has been highly modified, the interaction of vegetation with soil cannot be overlooked. It has been documented that leguminous species provide litter fall which is low in lignin and high in degradable organic carbon and nitrogen (Alegre et al., 2004). As such these species are recognized as being primary boosters of soil nutrients (typically N) within rehabilitation environments (Alegre et al., 2004). The deficiencies in nutrients has been recognized as a key factor affecting re-vegetation success (Schwenke et al., 2000). A decline in N in conventionally ploughed and rehabilitated sites at a Bauxite mine near Weipa, Western Australia was observed in sites with poorly performing rehabilitation. Losses of up to 69% in mineralised N within the top 0-10cm of soil occurred in the first 18 months after rehabilitation, this was attributed to the highly disturbed nature of mine site rehabilitation locations (Schwenke et al., 2000). In an exceptional study conducted by Alberti et al. (2011) the effect of rainfall on organic carbon content was inextricably linked to the types of vegetation which persisted in particular areas and the amount of N retained within the soil. 70% of the variation in their data was explained by the amount of precipitation within an area, where dry sites (<750 mm annually) gained soil organic carbon (SOC) after colonisation of native species, and wet sites lost SOC. The trend in carbon dynamics was linked to the nitrogen dynamics in the area as carbon losses occurred only when there was a decrease in soil nitrogen stocks, which occurs in wetter sites due to leaching.

Soil moisture is also another relevant factor when looking at the potential success of re-vegetation sites. It is unquestionable that moisture is needed in some form within the soil profile for vegetation to survive. It is evident that soil moisture retention does influence the success of rehabilitated sites due to it becoming a limiting factor in germination rates, especially in areas where conditions are harsh with little time for moisture to become available to the seed (Bochet et al., 2007). The presence of moisture within the soil is generally related to the decomposition rates of leaf litter within rehabilitation sites (Cortez, 1998). This process works by the assumption that leaf litter insulates the soil from desiccation, the presence of moisture and leaf litter helps to provide a favourable environment for decomposing fungi and microorganisms, which in turn influences the levels of nutrients within the top layer of soil which directly comes into contact with any viable seed dropped from re-vegetated species.

Soil fauna and microorganisms

Though soil fauna is not a component of this research it is important to discuss the significance soil microbes have on soil condition and therefore vegetation and in turn re-vegetation. Bourne et al. (2008) have suggested from the results of their study the effect of soil biota on growth of *Eucalypt microcarpa*, that variation in soil fauna has the potential to greatly influence the growth rates of re-vegetation on retired agricultural lands, such as those being re-vegetated in the Braidwood district. Bourne et al. (2008) also found that soil fauna provided benefits in terms of release of nitrogen and support of decomposition rates, but also to leaf robustness and biomass increases overall. Conversely Jouquet et al. (2010) have found that though the addition of compost and vermicompost (worm castings) both improved soil parameters (increased nutrients, C and pH) and seedling growth and biomass amounts, the interaction of local endogenic earthworms with these additions can have a negative effect on soil parameters and plant growth through leaching of the soil nutrients.

A major part of re-vegetation within Ghana takes place within the mining industry as mine spoil and impacted lands are rehabilitated as part of general practice within mine leasehold areas. Because of the limited private companies' funds for rehabilitation, costs-effective methods need to be used for rehabilitation. This has been the major objective of mining industries efforts in rehabilitating mined spoils through re-vegetation. One such comprehensive study was conducted on the importance of inoculation to increase the numbers of filamentous and non-filamentous bacteria and fungi within the soil (Wildman 2009). It was recognized that these microorganisms provide beneficial decompositional services within the leaf litter biomass and top soil layers making it advantageous to improve numbers of these microorganisms to improve re-vegetation success (Wildman, 2009). Inoculation of the soil to be re-vegetated results in an increase in actinomycetes (filamentous microbe spp.) and fungi in a measureable gradient from disturbed state to rehabilitated state. Hence this progression in soil microorganisms could provide an assessable gradient for decomposition rates, soil nutrient rates and therefore increases in vegetation development.

Behaviour of Heavy Metals in Contaminated Soils

Since the beginning of the industrial revolution, heavy metal contamination of the biosphere has increased considerably to become a serious environmental concern. Contamination by heavy metals can be considered as one of the most critical threats to soil and water resources as well as to human health. During the past decades, the annual widespread release of heavy metals reached 22000 Mg for Cd, 939000 Mg for Cu, 1350000 Mg for Zn, and 738000 Mg for Pb (Ahmadpour et al., 2012).

The two different pathways that heavy metals enter the environment are from natural and anthropogenic sources. Natural sources of heavy metals contamination usually result from the weathering of mines, which are themselves created anthropogenically. The toxicity of heavy metals is a very serious issue, because they have long persistence in the environment. The half-life of these toxic elements is more than 20 years. According to Sarma, (2011), this environmental problem has threatened the health of more than 10 million people in many countries. Heavy metal pollution has spread throughout the world. There are 53 elements that are classified as heavy metals. Their densities exceed 5 Mg m^{-3}, and they are known as universal pollutants in industrial areas (Sarma, 2011).

Heavy metal pollution is a crucial environmental concern throughout the world. It occurs in the soil, water, living organisms, and at the bottom of the sediments. Environmental contamination by heavy metals as a result of industrial and mining activities became widespread in the late 19th and early 20th centuries (Ahmadpour et al., 2012). Heavy metals, including Cd, Cu, Cr, Zn, Ni and Pb as critical pollutants, have an adverse effect on the environment, specifically at high concentrations in areas with severe anthropogenic activities. Although they are natural components of the earth’s crust, heavy metals’ biochemical equivalence and geochemical cycles have changed noticeably due to human activities. These metals are just being transformed from one form to another, because of their inability to degrade naturally. The heavy metals, namely copper, iron, zinc, molybdenum and magnesium are micronutrients and are considered to be essential to maintaining life in biological systems. However, at higher concentrations, these metals become highly toxic and threaten the health of animals and humans by influencing the quality of crops, water and atmosphere. The heavy metals Cd, Cu, Ni and Hg create greater phytotoxicity than Zn and Pb (Ahmadpour et al., 2012). The pollution of soil is a crucial matter that has attracted considerable public attention over the past few decades. A large proportion of land has become hazardous and non-arable for humans and animals, because of extensive pollution. It is unusual to have soils without at least traces of heavy metals, and the levels of these elements become more toxic due to anthropogenic or natural activities that are harmful for living systems (Ahmadpour et al., 2012).

Mine wastes are often characterized by shales, cobbles and pebbles, which have a very low water holding capacity. In addition to low pH and elevated metal concentration at these sites, other adverse factors includes absence of topsoil, periodic sheet erosion, drought, compaction, and absence of soil forming fine ingredients. Metal contaminated soils inhibit soil forming processes and plant growth. There is commonly shortage of essential nutrients supportive of biological growth (P, N, and K) and contains almost no organic matter. Toxic metals can also adversely affect the number, diversity and activity of soil organisms, inhibiting soil organic matter decomposition and N mineralization processes. The original soil of mined sites is usually lost or damaged, with only skeletal material. There is change in soil texture, loss of structure; and low availability of soil moisture; uncertain structure and unstable slopes due to hilly terrain. Sometimes mine waste sites (overburden, mine tailings) contain pyrites and sulphide minerals. These minerals when exposed to air and moisture oxidize to produce acid and soluble salts (Ahmadpour et al., 2012).

Pyrite oxidation and hydrolysis give rise to large amounts of H^+ ions which, by decomposition and exchange reactions with other spoil minerals, can give rise to high concentration of Al, Mn, Fe, Zn, Cu and other metals depending upon the composition of the originating mineral (Sheoran et al., 2012).

The possible source of heavy metals in the mining areas is crushed and exposed bedrock containing these metals through blasting. The metals considered in this study are Zinc, Copper, Lead, Arsenic, Iron, Manganese and Cadmium. These metals may be released in solution minerals during rains thereby making them available for detection.

Zinc

Zinc as trace element found in varying concentrations in all soils, plants and animals is needed in small but critical concentrations. The zinc which is available to plants is that present in the soil solution, or is adsorbed in a labile form. The soil factors affecting the availability of zinc to plants include; the total zinc content, pH, organic matter content, clay content, calcium carbonate content, soil moisture status, concentrations of other trace elements, concentrations of macro-nutrients, especially phosphorus and climate among others (Alloway, 2008).

Alloway (2008) explains that availability of zinc decreases with increasing soil pH due to increased adsorptive capacity. At neutral pH values, Zn^{2+} readily forms complexes with soil organic matter and clay minerals, becoming relatively immobile and accumulates in the surface soil horizons. The phytotoxicity level of zinc in soils varies between 100 - 500 mg kg^{-1} depending on plant genotype and soil pH. Alkaline, calcareous and heavily limed soils tend to be more prone to zinc deficiency than neutral or slightly acid soils. High levels of phosphorus may decrease the availability of zinc. Also higher concentrations of copper in the soil solution, relative to zinc, can reduce the availability of zinc to a plant (and *vice versa*) due to competition for the same sites for absorption into the plant root. This could occur after the application of a copper fertilizer (Alloway, 2008).

Zinc phytotoxicity is reported relatively often, especially for acid and heavily sludge soils. The physiology and biochemistry of the toxic effects of Zn in plants are likely to be similar to those reported for other trace metals; however, Zn is not considered to be highly phytotoxic. The toxicity limit for Zn depends on plant species and genotypes, as well as on a growth stage. A Zn content at about 300 mg kg^{-1} is reported to be toxic to young barley, whereas about 400 mg kg^{-1} is toxic to oats at the beginning of tillering (Kabata-Pendias, 2011).

Copper

Available copper in soils is held mainly as Copper cation (Cu^{2+}) on surfaces of clay minerals or in the association with organic matter. Copper present as an impurity in silicate minerals or carbonates is largely unavailable. Organic matter and soil pH are the predominant factors influencing copper availability. Copper availability decreases as organic matter in soil increases. Organic matter binds copper more tightly than any other micronutrient. This reduces fixation by soil minerals and leaching, as well as availability to crops.

Increasing the soil pH by liming increases the amount of copper held by clay and organic matter, thereby decreasing copper availability. Phytotoxic levels of Cu^{2+} vary from 100 - 1636 mg kg^{-1} depending on plant genotype and soil pH (Schulte and kelling, 1999).

Lead

Lead naturally occurs in magmatic and sedimentary rocks, and typically forms sulfide and carbonate minerals. The average lead content in the Earth's crust is estimated as 15 mg kg^{-1}. In the terrestrial environment, two kinds of lead are known: primary and secondary. Primary lead is of a geogenic origin and was incorporated into minerals at the time of their formation, and secondary Pb is of a radiogenic origin from the decay of Uranium and *Thorium* (Kabata-Pendias, 2011). Generally, lead is accumulated near the soil surface, mainly due to its sorption by SOM. Studies conducted by Kabata-Pendias (2011) suggest that SOM plays a decisive role in the Pb adsorption, but the fixation by clay minerals is much stronger. The mobilization of lead is usually slow, but some soil parameters, such as increased acidity, formation of Pb–OM complexes may increase its solubility. The lead levels of soils that are toxic to plants are not easy to evaluate, due particularly to the difficulty in assessing how much of soil Pb is available to plants (Kabata-Pendias, 2011). The study further showed that, an upper limit for lead content of an unpolluted soil could be established as 70 mg kg^{-1}. A similar value for lead concentration limit (60 mg kg^{-1}) was established in China for tea garden soils. The soil lead concentration of 450 mg kg^{-1} in a playground is also proposed. The maximum mean tolerance limit value for lead in soils of South Africa is accepted as 56 mg kg^{-1}, at its range of 2.99–65.8 mg kg^{-1} (Kabata-Pendias, 2011).

The solubility of lead is significantly lower than other trace metals in the environment, and is primarily controlled by soil pH. Lead minerals are very insoluble, and therefore lead is considered to be the least mobile of the heavy metals in natural environments; being 100 times less soluble than Cd in the pH range of 5-9. It is considered a major environmental pollutant and is phytotoxic to plants in the 100-1000 mg kg^{-1} range (Adriano, 2001; Kabata-Pendias, 2011). Low Pb concentrations in soils may inhibit some plant processes however, lead poisoning has rarely been observed under field conditions (Kabata-Pendias, 2011).

Arsenic

The average content of As in the Earth's crust is estimated as 1.8 mg kg^{-1} and it's highly associated with deposits of several metals and metalloids and therefore, known as a good indicator in geochemical prospecting surveys. Numerous oxide minerals of asenic are a result of the oxidation of sulfide deposits. The overall mean value of the total asenic for different soils is estimated as 6.83 mg kg^{-1} (Kabata-Pendias, 2011). Typical ranges for asenic in soils is 0.1 to 40 mg asenic kg^{-1} soil. Arsenic is primarily present as an anion in soils and its compounds are readily soluble, but have limited mobility due to strong sorption by clays, hydroxides, and organic matter (Kabata-Pendias & Pendias, 1992). The bioavailability of As is controlled by the oxidation state of the soil, the amount of phosphorus (P) in the soil, soil pH, and soil organic matter (Adriano, 2001; Kabata-Pendias, 2011). Arsenic and phosphorus have been found to react similarly in soils in terms of sorption capacities and bioavailability (Adriano, 2001). Arsenic is a constituent of most plants, and root growth stimulation has been observed in some species, yet arsenic is not considered an essential element for plant metabolism (Kabata Pendias and Pendias, 1992; Adriano, 2001). Phytotoxic levels of soil As have been documented as 15-315 mg kg^{-1} (Kabata-Pendias and Pendias, 1992). Total asenic is a relatively poor indicator of phytotoxicity. Multiple studies have shown higher correlation between plant growth and soluble asenic than total asenic (Adriano, 2001). Symptoms of As phytotoxicity in plants include wilted leaves, violet coloration, root discoloration, and growth reduction (Kabata-Pendias and Pendias, 1992; Adriano, 2001).

Jain et al. (1999) investigated the adsorption properties of asenic in soil and found that asenic was strongly adsorbed onto soil particles, was highly correlated with Fe-oxide and relatively immobile in soils. Wang et al. (2002) also reported that the transfer of As from soil to plant is low as a result of:

(i) the restricted uptake by plant roots due to low As bioavailability;
(ii) the limited translocation of As from root to shoot;
(iii) and As phytotoxicity even at low concentrations in plant tissues.

Sadiq et al. (1983) indicated increased As bioavailability by increasing soil pH with the addition of lime. They argued that because As is mostly present as an anion in soils, its bioavailability increases with increasing pH.

Iron

Kabata-Pendias (2011) explained that, iron is the most important metal and one of the major constituents of the lithosphere and plays a special role in the behavior of several trace elements. The colour of soils is largely associated with amounts and forms of iron compounds. As emphasized, the speciation and distribution of iron compounds in soils is very useful for the typology and diagnosis of soil properties. The general rules governing the mobilization and fixation of iron are that oxidizing and alkaline conditions promote the precipitation of iron, whereas acid and reducing conditions promote the solution of iron compounds. The released Fe readily precipitates as oxides and hydroxides, but it also substitutes for Mg and Al in other minerals and often complexes with organic ligands. Results presented by Kabata-Pendias (2011) indicated that amounts of metals bound to both amorphous Fe oxides and crystalline iron oxides in metal-polluted soils are fairly similar and governed mainly by soil properties. Many reactions are involved in the solubility of Fe in soil, but hydrolysis and formation of complexed species appear to be the most important. The solubility of Fe^{3+} and Fe^{2+} amorphous hydrous oxides especially control the mobility of soil Fe. Soils developed on acid sulfate soils have very poor agricultural productivity because of their acidity. This acidity is a result of the oxidation of pyrite. At high pH, pyrite is oxidized chemically, but at pH values below 4.5, it is mediated by microorganisms. These processes may result in a very low soil pH and greatly influence the release of metals (Kabata-Pendias 2011).

Plant injury due to Fe toxicity is most likely to occur on strongly acid soils (Ultisols, Oxisols), on acid sulphate soils, and on flooded soils. A high concentration of Fe in the soil solution is almost always related to Fe toxicity. The concentration of Fe^{2+} at about 500 mg kg^{-1} in paddy soils was reported to kill rice seedlings. Edible parts of vegetables contain Fe within mean values of 33 to 65 mg kg^{-1}, in carrot and soybean seeds, respectively. Various cereal grains do not differ much in their Fe concentrations. The average Fe contents of different cereals range from 31 to 98 mg kg^{-1}. Values above 100 mg kg^{-1} are reported only for some countries, for example, 218 and 133 mg kg^{-1} in barley from the United Kingdom and in oats from Canada respectively (Kabata-Pendias, 2011).

Cadmium

The average Cd content for the Earth’s crust is given as 0.1 mg kg^{-1}. The metal occurs rarely in nature in a pure form and has similar ionic structures, electronegativities, and chemical properties with Zn. Cadmium has a stronger affinity for S than Zn and thus, its mobility in an acidic environment is higher than that of Zn (Kabata-Pendias, 2011). Although Cd has the same valence and similar ionic radius as Ca, it does not substitute for Ca in minerals. Cadmium is considered as being one of the most ecotoxic metals that exhibits adverse effects on all biological processes of humans, animals and plants. This reveals its great adverse potential to affect the environment and the quality of food. The world average soil cadmium concentration is estimated as 0.41 mg kg^{-1}. The main factor determining cadmium contents of soils is parent material. The average contents of Cd in soils lie between 0.2 and 1.1 mg kg^{-1}. In uncontaminated soils, its contents are highly governed by soil texture and ranged from 0.01 to 0.3 mg kg^{-1} in sandy soils and from 0.2 to 0.8 mg kg^{-1} in loamy soils. Surface soils from major agricultural production areas of the United States contain Cd within the range of < 0.01 to 2.0 mg kg^{-1} (Kabata-Pendias, 2011).

Alloway (2011) reported Cd contents in reference soils from different countries to range from 0.06 to 4.3 mg kg^{-1}. Although Cd is considered to be a nonessential element for metabolic processes, it is effectively absorbed by both root and leaf systems and is also highly accumulated in soil organisms. There is evidence that an appreciable fraction of Cd is taken up passively by roots, but Cd is also absorbed metabolically. The Cd content of plants is of great concern as a pathway of Cd to man and animals. Thus, tolerance and adaptation of some plant species to higher Cd levels, although important from an environmental point of view, create a health risk. Cadmium concentration in food plants is of special concern and several international organizations have been collaborating to protect human health and the environment from the Cd stress. According to Kabata-Pendias (2011) many soil parameters control Cd uptake by plants, some plants species reveal an unusual ability for its uptake and transfer to upper parts. Some of these plants include *Alyssum murale*, *Zea mais*, *Salix viminalis*, *Helianthus annuus*, and *Viola baoshanensis*.

Manganese

Manganese is one of the most abundant among trace elements in the lithosphere. Its common occurrence in rocks ranges from 350 to 2000 mg kg^{-1} and higher concentrations are associated with mafic rocks. Manganese is a member of the iron family and both elements are closely associated in geochemical processes. Manganese contents of worldwide soils vary from 411 to 550 mg kg^{-1}. Its highest levels occur in loamy and calcareous soils. Manganese has not been considered to be a polluting metal in soils however, the Maximum Allowable Concentration (MAC) value for this metal in agricultural soils is estimated at the range of 1500–3000 mg kg^{-1}. Generally, the most readily available manganese is in acid and flooded soil. More than a tenfold increase in the manganese content of lucerne (alfalfa) was observed in plants grown on flooded soil, compared to background values. Indeed, manganese concentration in plants shows a negative relationship with increasing soil pH and a positive relationship with soil OM.

Soil-to-plant Transfer Factor

The transfer or translocation factor (TF) is formulated as TF = the Metal concentration in the edible part of the vegetable or crop/Metal concentration in the media sample (Khan et al., 2010; Saglam, 2013). The metal concentrations in the extracts of the soils and plants are mostly calculated on the basis of dry weight. If the ratios >1, the plants have accumulated elements, the ratio 1 indicate that the plants are not influenced by the elements, and ratios < 1 show that plants exclude the elements from the uptake (Olowoyo et al., 2010). If the plants have higher TF values, they can be used for phytoremediation (Saglam, 2013).

Heavy metal levels in Ghana soils and food crops

Hayford et al. (2008) investigated the presence of toxic elements and possible heavy metals in cassava and soil from the mining communities in and around Tarkwa. Their results showed that the maximum concentrations of the respective metals in cassava and soil samples were arsenic (As = 0.7 mg kg^{-1} and 14.35 mg kg^{-1}), zinc (Zn = 38.42 mg kg^{-1}, and 42.36 mg kg^{-1}), and copper (Cu = 66.39 mg kg^{-1}, and 53.46 mg kg^{-1}). They concluded that the heavy metals concentrations in cassava were higher than the values proposed by the FAO, the Expert Committee on Food Additives of the WHO and the Agency for Toxic Substance and Disease Registry (Hayford et al., 2008).

Zango et al. (2013) also studied the levels of Cd, Pb, Zn and Cu (mg/kg) in soils and food crops sampled from farms around surface mined areas at Wassa-Amenfi-West District of Ghana. They indicated that mean concentrations of Lead ranged from 1.03-5.5 mg kg^{-1}, cadmium 1.87 - 2.48 mg kg^{-1}, Zinc 7.93-17.46 mg kg^{-1}, and copper from 4.42 -12.87 mg kg^{-1} respectively. They concluded that heavy metal levels in soil samples were within the permissible limits of Indian and EU standards. However, heavy metals in cassava were higher than in soils, with Pb, Cd, Zn and Cu in cassava exceeding FAO/WHO recommended values. Therefore local inhabitants were not safe in consuming the cassava crops (Zango et al., 2013).

Factors affecting heavy metals mobility and bioavailability in plants

Plant uptake of trace elements is generally the first step of their entry into the agricultural food chain. Plant uptake is dependent on: movement of elements from the soil to the plant root, elements crossing the membrane of epidermal cells of the root, transport of elements from the epidermal cells to the xylem, in which a solution of elements is transported from roots to shoots, and possible mobilization, from leaves to storage tissues used as food (seeds, tubers, and fruit) in the phloem transport system. After plant uptake, metals are available to herbivores and humans both directly and through the food chain. The limiting step for elemental entry to the food chain is usually from the soil to the root (Amare, 2007). Plant species, relative abundance and availability of necessary elements also control metal uptake rates. Abundant bio available amounts of essential nutrients can decrease plant uptake of non-essential but chemically similar elements. Bioavailability may also be related to the availability of other elements. For example, copper toxicity is related to low abundances of zinc, iron, molybdenum and (or) sulphate (Amare, 2007).

According to Singh et al. (2012), different vegetable crops grown on heavy metal contaminated soil show marked difference in metal accumulation, their uptake and distribution pattern. Crop species also showed remarkable difference in metal concentration of various plant parts. Generally, based on metal accumulation in edible parts and whole plants, the root and leafy vegetables showed higher distribution of metals to the edible parts, whereas fruit type vegetables exhibited least transport of metals to fruits (Singh et al., 2012).

Table 1: Ranges of maximum allowable concentrations (MAC) or trigger action value (TAV) for trace metals in agricultural soils (mg kg^{-1})

Metal	MAC [a]	TAV [b]
As	15–20	10–65
Cd	1–5	2–20
Cu	60–150	60–500
Pb	20–300	50–300
Zn	100–300	200–1500

Source: Kabata- Pendias (2011).

Table 2: Heavy metal composition of typical uncontaminated soils and agricultural crops on dry weight basis (mg kg^{-1})

Heavy metals	Range in soil	Range in agricultural crops
Cd	0.01–0.7	0.2–0.8
Cu	2–100	4–15
Fe	7,000–55,000	–
Mn	100–4,000	15–100
Pb	2–200	0.1–10
Zn	10–300	15–200

Source: Alloway (2011).

A brief history of mine reclamation

Prior to the Sixteenth Century, mines and quarries, with some exceptions, were primitive, usually small, and dominated by manual labour, (Hoover and Hoover, 1950) marks the start of the era of modern mining. Mines typically disturbed only a few hectares and the loss of land use and off-site impacts were often small and unimportant to society and hence reclamation was not considered. In the following centuries, the average size of mines increased (Gordon, 2002) Starting in the 1940s, there was development of large earth moving equipment and modern accounting and investment methods that permitted an expansion into large mining activities. Most open pit mines today disturb a few thousand hectares, the largest disturb tens of thousands of hectares – the size of modest cities. And as the economics of moving vast amounts of the earth improved, lower grade ores are mined – typically less than five percent of the rock and soil moved is ore (and often much less than one percent), the rest is waste. More than ever, mining is an exercise in waste management (Hutchison & Ellison, 1992).

Coincident with the enlargement in mine disturbances over the past five decades, has been a change in focus from taming the environment to one of protecting the environment (Cronon, 1995). Perhaps ironically, the wealth created in part by mining, has led to increased concern for the environment by society. The impacts of primary industries (agriculture, forestry, fishing, mining) on the environment are under increasing scrutiny. Regulations requiring mine reclamation started in the 1960s. As increasing population exerts ever-increasing pressures upon the finite land base, society recognizes the need to balance the environmental disruption and landscape changes caused by mining (Gordon, 2002).

In return for being permitted to reap the mineral wealth from the land, the mining industry has, in essence, formed a social contract with society – that after mining, using some of the profits from mining, the land will be restored to a safe and useful state. Miners have adopted a mantra of "temporary use of the land" (Kumar, 2013).This social contract forms the basis of government legislation, policy, and regulations that attempt to form a more clearly defined obligation on the part of the mining companies.

Remediation of Mined Soils

Mining generates considerable waste materials and tailings, which are deposited on the surface as mine spoil dumps. Removal of fertile topsoil, formation of unstable slopes prone to sliding and erosion, and siltation of water bodies due to wash off of mineral overburden dumps are also major negative effects of mining. The metals released from mining, smelting, forging, and other sources would accumulate in the soil, altering its chemistry (Khan et al., 2009). Metal contamination is not restricted to the mining site only because, considerable release of metals occurs through acid mine drainage and erosion of waste dumps and tailing deposits (Kumar, 2013).

Current conventional methods to remediate heavy metal contaminated soil and water, such as ex situ excavation, landfill of the top contaminated soils detoxification and physico-chemical remediation are expensive, time consuming, labour exhaustive and increase the mobilization of contaminants, and destroy the biotic and structure of the soil (Danh et al., 2009). These remediation techniques therefore, are not technically or financially suitable for large contaminated areas. Bioremediation was developed as a technology to degrade pollutants into a low toxic level by using microorganisms. However, the use of this technology to remediate contaminated areas by applying living organism was less successful for extensive metal and organic pollutants. Plants are able to metabolize substances produced in natural ecosystems. Phytoremediation/re-vegetation is an approach in which plants are applied to detoxify contaminated areas (Ahmadpour et al., 2012).

Reclamation of mine dumps and abandoned mine lands (AML) is a complex multi-step process. The first step in transforming the mine contaminated lands into productive agricultural lands is restoring its ecological integrity (Sheoran & Sheoran, 2009; Juwarkar & Singh, 2010). Most AML sites and many active mining or re-mining sites however, lack any true topsoil and often consist primarily of mine spoils or overburden whose properties can range from loose, coarse textured material with many rock fragments, to highly compacted clay material. Broadly, two types of effects are plausible: excesses (supra-optimal levels of chemical elements including metal ions) and deficiencies (suboptimal concentrations of essential elements). Mine contaminated soils thus, represent a very harsh environment for crop production (e.g., phytotoxicity and high acid production potentials of waste materials, low fertility, and limited topsoil availability). The principal restoration options are therefore, ameliorative (improving the physical and chemical nature of the site) and adaptive (careful selection of species, cultivars, or ecotypes) (Kumar, 2013).

Re-vegetation and its importance

Throughout the world re-vegetation is carried out in an attempt to restore habitat, influence and improve water quality, preserve threatened flora and fauna, control and mitigate soil erosion, rehabilitate areas devastated by mining and provide shelter for livestock. The removal of woody vegetation has also indirectly led to emission of carbon and loss of nutrients in soils globally and re-vegetation may provide a pathway to restoring these vital ecological cycles. The potential for re-vegetation to restore habitat as well as provide nutrient recycling and carbon sequestration services is a highly relevant topic at this time in the global climate and agriculture (Yadav et al., 2009; Wang et al., 2011).

According to Liping et al. (2009) re-vegetation is the most widely accepted and useful practice for reduction of erosion and protection against soil degradation. Many countries have adopted re-vegetation as an important measure to restore and reconstruct natural ecosystems. Re-vegetation does not only increase the land cover with improved plant diversity, but can also strengthen the soil capacity by conserving water and increasing the soil fertility level. Various studies have demonstrated that re-vegetation has a great effect on soil nutrients and enzymatic activities (Liping et al., 2009).

Furthermore, the effects of re-vegetation on soil nutrient and biological properties are reflected in deeper soil layers as the re-vegetation time increases. As reported by Liping et al. (2009), after more than 20 years of restoration with artificial vegetation coupled with the cessation of grazing in the Zhifanggou Watershed, the plant communities had obvious effects on soil nutrient accumulation. They also found that the soil nutrient content was higher in the 0 - 20 cm than in 20 - 40 cm layer. The contents of soil organic matter, total nitrogen, available N and P, sucrase, urease and neutral phosphatase in an abandoned pasture all decreased as the soil depth increased, but that, all of these values increased with time (Liping et al., 2009).

Two important distinct processes when it comes to re-vegetation are restoration and rehabilitation. There are distinct and diverse differences between the aims and outcomes of rehabilitation and restoration which are quite often overlooked when the success or merits of re-vegetation efforts are assessed (Zedler, 2007). It is critical to outline by which standard definition the re-vegetation is part of and assessed, accordingly. The distinct differences in definitions will impact on whether the re-vegetation will be determined as a success or failure in the years after planting. Re-vegetation is the process carried out for a number of reasons including farm forestry and forestry operations, ecosystem and biodiversity enrichment, amenity, erosion and water movement control, stock comfort and naturally regenerating vegetation (Atyeo & Thackway, 2009).

Restoration

Broadly, the term ‘ecological restoration’ has been interpreted to mean the processes involved in returning an altered, dysfunctional and or degraded landscape to a functioning healthy ecosystem (Willis, 2012). According to Willis (2012), ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed. The overall aim is to restore a damaged ecosystem to its pre-disturbed condition however, the success of restoration is not measured on whether it has achieved its former state but rather that, it becomes a functioning, self-sustaining and environmentally resilient ecosystem (Society for Ecological Restoration International Science & Policy Working Group, 2004). Restoration is the activity that accelerates the recovery of the ecosystem in terms of the ecological connections and functions of the system, the structure and diversity of species, and the ability of the ecosystem to support biota (Willis, 2012).

Rehabilitation

The words rehabilitation and restoration are often, and incorrectly, interchanged when referring to re-vegetation. Rehabilitation is any act which seeks to improve a landscape from a degraded state but not necessarily to a functioning healthy ecosystem. Catalysts for rehabilitation include the lowering of water tables, providing shelter for livestock and crops, improving water quality, improving aesthetic conditions, providing habitat by re-establishing native vegetation to a previously denuded landscape producing timber (Fenton, 2010), carbon sequestration (Yadav et al., 2009) and land stabilization (Willis, 2012).

Gauging the success of Re-vegetation

Success is a subjective term interpreted in different ways by different people. It can be especially subject based when applied to analyzing individual aspects of a whole ecosystem (Willis, 2012). The term success is defined broadly by the Oxford dictionary as

- the accomplishment of an aim or purpose,
- the good or bad outcome of an undertaking

From this definition and when positioned within the context of assessment of re-vegetation, the re-vegetation should begin with clearly defined aims and or purposes for comparison along a timeline, and can also be assessed as either a success or failure on account of the outcomes which were defined at the beginning of the re-vegetation. The difficulty when gauging success in a multi-faceted system such as an ecosystem is the aspect on which you are gauging the success. For the purposes of this research, success of the re-vegetated mine areas will be assesses by the ability of the re-vegetated mine land to:

- provide soil nutrients for plant growth
- reduce the availability of heavy metals to acceptable level both in soil and crops grown

Plant species used for Re-vegetation and their impacts on soil

The re-vegetation of eroded ecosystems must be carried out with plants selected on the basis of their ability to survive and regenerate or reproduce under severe conditions provided both by the nature of the dump material, the exposed situation on the dump surface and on their ability to stabilize the soil structure (Madejon et al., 2006). The normal practice for re-vegetation is to choose drought-resistant, fast growing crops or fodder which can grow in nutrient deficient soils. Selected plants should be easy to establish, grow quickly, and have dense canopies and root systems. In certain areas, the main factor preventing vegetative growth is acidity. Plants must be tolerant of metal contaminants for such sites. Role of exotic or native species in reclamation needs careful consideration as newly introduced exotic species may become pests in other situations. Therefore, candidate species for vegetation should be screened carefully to avoid becoming problematic weeds in relation to local to regional floristic. For artificial introduction, selection of species that are well adapted to the local environment should be emphasized. Indigenous species are preferable to exotics because they are most likely to fit into fully functional ecosystem and are climatically adapted (Sheoran et al., 2010).

Grasses are considered as nurse crop for an early vegetation purpose. Grasses have both positive and negative effects on mine lands. They are frequently needed to stabilize soils but they may compete with woody regeneration. Grasses, particularly C4 ones, can offer superior tolerance to drought, low soil nutrients and other climatic stresses. Roots of grasses are fibrous that can reduce erosion and their soil forming tendencies eventually produce a layer of organic soil, stabilize soil, conserve soil moisture and may compete with weedy species. The initial cover must allow the development of diverse self-sustaining plant communities (Sheoran et al., 2010). Trees can potentially improve soils through numerous processes, including - maintenance or increase of soil organic matter, biological nitrogen fixation, uptake of nutrients from below and reach of roots of under storey herbaceous vegetation, increase water infiltration and storage, reduce loss of nutrients by erosion and leaching, improve soil physical properties, reduce soil acidity and improve physical properties, reduce soil acidity and improve soil biological activity. Also, new self-sustaining top soils are created by trees. Plant litter and root exudates provide nutrient-cycling to soil. On mine spoils, nitrogen is a major limiting nutrient and regular addition of fertilizer nitrogen may be required to maintain healthy growth and persistence of vegetation (Yang et al., 2003). An alternative approach might be to introduce legumes and other nitrogen-fixing species. Nitrogen fixing species have a dramatic effect on soil fertility through production of readily decomposable nutrient rich litter and turnover of fine roots and nodules. Mineralization of N-rich litter from these species allow substantial transfer to companion species and subsequent cycling, thus enabling the development of a self-sustaining ecosystem (Sheoran et al., 2010). Sheoran et al. (2010) further reported that native leguminous species show greater improvement in soil fertility parameters in comparison to native non-leguminous species. Also, native legumes are more efficient in bringing out differences in soil properties than exotic legumes in the short term.

Generally, a mix of plant species is required because it is impossible to predict the success of any one species in marginal environments where the vegetation is going to receive little or no maintenance. Monocultures are disadvantaged due to the vulnerability to pests and diseases, they do not satisfy fully the multiple use and conservation roles, do not provide a balanced ecosystem and also they fail in providing more balance stable system in the face of environmental variability and directional change than mixed cultures (Mensah, 2015).

Mixed forestry systems, on the other hand, might be more stable as they sequester carbon more securely in the long term. In short, mixed systems could offer more to mitigating climate change than monocultures. The species mix should include grasses, forbs and woody species, both bushes and trees, except where specific requirements make such a mix undesirable, as with some types of gully reclamation or along pipe line corridors (Mensah, 2015). Zhang et al. (2006) had found that the annual litter amount of 55-year old *Pinus massoniana* and *M. macclurei* mixed forest was 11.2% higher than that of a similar-aged *P. massoniana* stand. Species composition is important for litter production within the same climate range (Yang et al., 2004).

Role of Re-vegetation in Improving the Soil physical properties

Re-vegetation plays very important roles in improving the soil physical properties. Some of the soil properties that re-vegetation improves are elaborated below.

Maintenance or improvement of soil physical conditions

The superior soil structure, porosity, moisture characteristics and erosion resistance under forest is well documented, as is their decline on forest clearance. Porosity is key to many physical properties: pores of 5-50 µm in diameter determine available water holding capacity, while those of over 250 µm in diameter are necessary for root penetration (Mensah, 2015). Compaction occurred during degrading of overburden and topsoil, mining, and reclamation activities at the time of unfavourable moisture conditions, and because of insufficient time for the soil-forming processes to decrease bulk density. Mensah, (2015) explained that abrupt increase in bulk density at the 30 cm depth is due to overburden and spoil material being extensively graded before topsoil application and the presence of large amounts of rock fragments at depths below 30 cm and that water content of the soil is an important property that controls its behaviour.

Improvement of infiltration and soil moisture availability/content

Ground vegetation, such as grasses will protect the slope against erosion by rain drop impact and runoff, and also trap moving sediments, while shrubs and trees will increase the strength of the soil through root reinforcement. Vegetation increases the infiltration of water into the soil (Mensah, 2015). However, this can cause problems where rainfall amounts and intensities are very high. Whilst the resulting reduction of in runoff will help control surface erosion, the increased moisture content of the soil may exacerbate the rate of mass soil failure (Singh et al., 2002). They further reported that the reduction in runoff is, to a small degree, caused by canopy interception and direct transpiration, but the greater part of its results from higher soil infiltration capacity under trees.

Breaking of compact layers by roots

The improvement in surface protection causes a reduction in crusting and effect in compaction leading to increased infiltration; less leaching and prevention of erosion (Mensah, 2015).

Modification of extremes of soil temperature

There is experimental evidence from studies of minimum tillage that a ground surface litter cover greatly reduces the extremely high ground temperatures sometimes over 50°C that is expected on bare soils in the tropics (Singh et al., 2004). Mensah, (2015) points out that bare soil absorbs heat and becomes very hot during dry season very quickly and becomes very cold during the rainy season, but this will not happen if the surface of the land is kept covered with vegetation which serves to insulate the soil. In this case, the soil will neither become too hot or too cold.

Improvement of soil texture and soil structure

According to Mensah (2015), significant variations in silt and clay suggest that plantations are capable of changing the soil texture after their establishment and growth in due course. Singh (1993) indicated that the textures of mine spoils are drastically disturbed due to irregular pilling of overburden materials. The naturally re-vegetating mine spoil of five years ago showed percentages of sand, silt and clay as 61, 25 and 14. Particle size distribution is a major soil physical factor governing a successful re-vegetation on reclaimed land as it influences water holding capacity, bulk density, soil moisture availability and nutrient contents as well as availability.

Role of re-vegetation in improving the soil chemical properties

Reduction of soil acidity

Liter from trees tend to motivate the effect of leaching through addition of bases to the soil surface. However, whether tree litter can be a significant means of raising pH of acid soils is doubtful, owing to the orders of magnitude involved. One of the reasons for the above is that the calcium which trees supply through litter fall is insufficient to reduce the acidity even by a one pH point (Singh et al., 2002).

According to Singh et al. (2004), there are even cases where, in the temperate zone, tress produce acid, mor-type humus, which can lead to appreciable increase in soil acidification. However, experiments have also proven that the bases released by litter decay can check acidification. The increase in pH due to plantations suggests that the organic matter input modifies the pH of the soil. Since most plant species used for re-vegetation are dicotyledonous, these may release more base cations like Ca^{2+} into the soil and thus increase the pH of the soil more than the fresh mine spoil (Mensah, 2015).

Reduction of salinity or sodicity

Afforestation has been successfully employed as a means of reclaiming saline and alkaline soils (Singh et al., 2004). For example, under *Acacia nilotica* and *Eucalyptus tereticornis* in Karnal, India, lowering of top soil pH from 10.5 to 9.5 in five years and of electrical conductivity from 4 to 2 dSm^{-1} have been reported, but with trees establishment assisted by additions of gypsum and manure (Singh et al., 2004).

Reduction of the rate of organic matter decomposition

It is known that the rate of loss of humified organic matter is lower in forest than agriculture. Reduction in temperature by the tree canopy and litter cover is implicated in this observation (Singh et al., 2002).

Availability of nitrogen

Nitrogen is believed to be absorbed from the atmosphere and stored in the soil, which is then taken up in the form of nitrates by plants for use. The root nodules found on especially leguminous species help to achieve this. Leguminous cover crops such as *Pueraria*, *Centrosema* and *Calapogonium* are more effective. Tree species such as *Leucaena leucocephala* and other leguminous tree species are also equally effective. The nitrogen absorbed from the atmosphere in turn contributes to improving the soil fertility. The litter fall which improves the organic matter status of the soil eventually provides a favourable soil environment for N-fixation (Mensah, 2015).

Availability of phosphorus

Mineralized P is found to be higher in soils treated with organic matter. However, the total P accumulation in stem and bark is far greater than in leaves alone. P in the surface litter only accounts for small percentage of the total accumulation in the forest. It has been suggested that phosphorus deficiency is the inhibitive factor resulting in slow growth of *Acacia mangium* (Mensah, 2015).

Methods used in Re-vegetation process

Agricultural Approach

Williams et al. (1998) expounded that the agricultural approach, although somewhat outdated, has been used most often for re-vegetation in the past and is still the approach dictated by The Surface Mining Control and Reclamation Act (SMCRA) of 1977 regulations for coal mining projects. The agricultural approach generally employs standard methods of replacing the topsoil, fertilizer application, and planting typical reclamation species to re-vegetate disturbed land. Unfortunately, topsoil was not routinely salvaged during most historic mining activities and is not always available for use. Since the soils in the area surrounding most mine sites are naturally mineralized, importing suitable soils typically requires long, uneconomic haul distances. Therefore, the ameliorative and adaptive approaches, which directly re-vegetate mine sites without top soil, are often desirable. Although developed and used in Europe since the 1960s, it is believed the first successful utilization of both ameliorative and adaptive approaches on the North American continent was at the California Gulch Superfund site located near Leadville, Colorado in 1992. The approach used for the re-vegetation of this site was documented in the May/June 1996 issue of Land and Water magazine.

Ameliorative Approach

Paul et al. *(*2012) indicated that the ameliorative approach involves chemically modifying the soils to correct whatever problems are faced. Using species that are not tolerant of elevated levels of heavy metals introduces the need for waste treatment and amelioration, to alleviate toxicity and acidity, until vegetation becomes self-sustaining. Then the plants themselves may control the hostile conditions in the waste.

Paul et al. (2012) further argued that the aim of the approach was to determine the most effective method of controlling acidity and the release of toxic metals in mine land, with a view to allowing the successful growth of non-tolerant plant species. Prediction of pollutant movements through and away from the waste, by the establishment of toxic element dispersion mechanisms and patterns, was originally suggested as a way of improving reclamation strategies. Each case of contamination is unique and there is a need for a flexible predictive model. By determining the form of toxic elements and the routes of dispersal, greater insight into their impact on planned re-vegetation strategies can be gained (Paul et al., 2012).

Along with predictive modeling, novel methods of amelioration have been developed, the success of which will derive from the correct prediction of the variable conditions normally found within the boundaries of a derelict site. These novel methods of reclamation must overcome the hostile chemical, physical and biological characteristics of the spoil. Acidic mine spoils are particularly hostile to vegetation due to direct damage inflicted on plant roots and uptake mechanisms by hydrogen ions and also by the presence of plant-available heavy metals. The use of various industrial minerals as in-situ amendments to the waste has been intensively investigated, and the initial results show that they have great promise in controlling acidity and release of metals into solution. Unfortunately previous research has indicated that the control of acidity can sometimes clash with control of metals in solution (Paul et al., 2012).

Adaptive Approach

Williams et al. (1999) further elucidated that the adaptive approach involves identifying, specifying and establishing plants which are ecotypically differentiated, or adapted and tolerant of the site conditions. The method involves utilizing tissue culture techniques and growth media adjusted chemically to emulate the specific site conditions. A plant's germination and initial root growth response are indicative of its response to actual site conditions. In previous work, approximately 1000 species and varieties of the genera *Agropyron* (Wheatgrass), *Festuca* (Fescue), *Agrostis* (Bentgrass), *Phleum* (Timothy), *Poa* (Bluegrass) and *Vicia* (Vetch) have been tested utilizing the in-vitro methodology. Several varieties of the species *Festuca rubra* (Red fescue), *Agrostis tenius* (Colonial bentgrass), *Phleum ptatense* (Timothy) and *Achillea mellefolium* (Yarrow) were identified to be tolerant to low pH and high metal concentrations which are common on mineralized sites. Varieties of some of these particular species were previously unknown to be tolerant of these typical site conditions (Williams et al., 1999).

In addition to low pH, high metal concentrations and the often erodible nature of the soils, reclamation at historic mine sites is often further complicated by steep slopes and severe exposure problems related to mountainous area and high altitude. The combination of poor soil physical and chemical conditions and severe exposure difficulties can make natural recovery impossible and human rehabilitation very difficult in disturbed areas. Disturbed areas which have remained barren since their initial disturbance over 100 years ago are common. Previous attempts to re-vegetate these areas have often produced results which were less than acceptable to the concerned parties (Williams et al., 1999).

Cultivation of Cassava (*Manihot esculenta*)

Cassava, which can grow well on marginal lands, is one of the most important staple foods in Ghana. Cassava production represents approximately 50% of all roots and tubers production in the country. The majority of cassava is grown by small-scale farmers with small landholdings. At that scale production, harvesting, and post-harvest handling are carried out with limited chemical and technical inputs. It is grown in all regions of Ghana but is particularly abundant in Central, Eastern, Brong Ahafo, Volta, and Ashanti regions (Ulrich et al., 2013). According to the statistics of MOFA production of cassava roots has increased by almost 40% from 2007 to 2011. In large part this is due to an increase in average yield per hectare of 26% over that period from 12.76 to 16.17 tonnes per hectare. The amount of land under cultivation has increased 11% in that time (Ulrich et al., 2013).

The main planting season for cassava is during the rainy season from May to September. Cassava is harvested approximately 12 months after planting so harvesting can take place any time from March to October (in an average year). The largest percentage of the cassava root harvest comes onto the market in the early part of the wet season (May to July) before planting begins. Harvesting labour accounts for 15-20% of cassava root production costs. Cassava can grow and produce reasonable yields on soils where many other crops would fail. It is highly tolerant of soils with low levels of phosphorus and can generally grow even with no application of P-fertilizer. That is because cassava has formed a mutually beneficial association with a group of soil fungi called “vesicular-arbuscular mycorrhizae”. Present in practically all natural soils, mycorrhizae penetrate the cassava root and feed on the sugars it produces. In exchange, the fungi’s long filaments transport phosphorus and micronutrients to the root from a greater volume of the surrounding soil than the root alone could reach. That symbiotic association allows cassava to absorb sufficient phosphorus for healthy growth. Cassava can also be grown on very acid and low-fertility soils because it tolerates low pH and the associated high levels of exchangeable aluminium. While the yields of crops such as maize and rice are usually affected strongly when the soil pH is below 5 and aluminium saturation is above 50 per cent, cassava yields are normally not affected until the soil pH is below 4.2 and aluminium saturation is above 80 per cent. For that reason, cassava may not require large amounts of lime in acid soils, where other crops would not grow without them (FAO, 2013).

Processing Cassava has traditionally been regarded as ‘poor man’s food’ but increasingly its utilities as a cash crop are being recognized. Opportunities exist for earning incomes from processed cassava products, but significant constraints also continue to place restrictions on some opportunities (Ulrich et al., 2013). Cassava is processed to control deterioration of roots and decrease toxicity. Due to its high perishability and potential high cyanide content fresh cassava roots should be processed within 1-2 days of harvesting. These factors, combined with high moisture content of approximately 70%, restrict the marketing and transportation options for cassava. During a recent study by both farmers and processors stated that sufficient roots are available and there have been some yield improvements with new varieties. Of the roots harvested in Ghana approximately 50% of cassava is either consumed or sold as fresh roots to produce (at household level) boiled or pounded cassava (Fufu) (Ulrich et al., 2013). The major pests of cassava in SSA are the cassava green mite and the variegated grasshopper. The main diseases affecting cassava are cassava mosaic disease (CMD), cassava bacterial blight, cassava anthracnose disease, and root rot.

Summary of Literature Review

The review has shown that re-vegetation plays important role in the reclamation of mined lands by improving soil fertility. Obviously, there is no documented research information on the use of reclaimed mined land for crops production in Ghana. In view of the growing food insecure populations of the world it is important to research in to the safe use of re-vegetated mined lands for agricultural productivity especially, in Ghana. Re-vegetated site must only be gauged a success or failure based on the motivations for which the re-vegetation was undertaken. A re-vegetated mined site may be successful at stabilizing soil structure and improving the organic content of the soil, and on that scale the re-vegetation is a success, it can also be sufficient ecological sustainable but not adequate for agricultural (crop) production. These therefore, informed the choice of the topic, formulation of hypothesis and objectives of the present study.

CHAPTER THREE

RESEARCH METHODS

This section presents a brief account as well as the description of the study area, and the methodological approaches carried out in order to achieve the set out objectives of the study as well as materials/tools used.

Description of Study Area

The research was carried out at Obuasi, located in the Ashanti Region of Ghana. It is about 64 km south of Kumasi, the regional capital and 300 km north-west of Accra, the capital of Ghana. It is situated on latitude 6° 12’ 00’’ North and longitude 1° 40’ 00’’ West. It is located in the tropical evergreen rain forest belt. It covers an area of about 162.4 km^2 and is bounded on the south by Upper Denkyira District of the Central Region, east by the Adansi South District, west by the Amansie Central District and north by the Adansi North District. The climate is of the semi-equatorial type with a double rainfall regime. Total annual rainfall is about 1700 mm. Mean average annual temperature is 25.5 ºC and relative humidity is 75-80 % in the wet season (Bempah, 2013).

The area is underlain by metavolcanic, pyroclastic and metasedimentary rocks. The metavolcanic rocks are of basaltic and gabbroic in composition whereas the metasedimentary rocks are mainly tuffaceous and carbonaceous phyllites, tuff, cherts and manganeferous sediments (Foli & Nude, 2012). These two units are contemporaneous and separated by a major shear zone. Intruding the metavolcanic and metasedimentary rocks are magmatic bodies and porphyritic granitoids consisting of hornblende-rich varieties that are closely associated with the volcanic rocks, and mica-rich varieties which are found in the metasediment units. The Birimian is overlain by the Tarkwaian rocks made up of sedimentary units and also recent alluvial deposits. Gold ores contain high sulphide minerals, made up mainly of arsenopyrite and pyrite. The top 30-80 m of the subsurface is made up of porous rocks that mark the water table in the area (Foli & Nude, 2012). Rock porosity ranges from 1.0-5.0×10G7 m/s. These rocks have undergone some degree of weathering and consist mainly of clay deposits which have subsequently been hardened and altered. A cross section of the hills along the roads reveals the presence of an uneven distribution of quartz veins injected into the phyllites which break up on weathering to give rise to pisolithic and gravely pebbles (Boateng et al., 2012). The topography of Obuasi and its environs varies from gently undulating to hilly. There are moderately high-elevated lands with lowlands and valleys between them. The highlands trend in different directions; they have flat and plateau tops and are generally amorphous in their shapes (Bempah et al., 2013). The FAO-UNESCO classification (1990) of the soils in the study area are Orthic-Ferric Acrisol (Adu, 1992) and belong to the order Ultisol (USDA Taxonomic System, 2005).

Two re-vegetatively restored mined spoils at AngloGold Ashanti gold mine were selected for the purpose of this study. These included a 17 year restored Nhyiaso and a 15 year restored T_1 sites. An original unmined site cropped with cassava was used as a control for the study.

Nhyiaso Re-vegetatively Restored Mined Spoil

The site is about 180 m to the west of the Nhyiaso pit across the main Obuasi-Dunkwa highway. It is adjacent to the Nhyiaso community where surface mining was undertaken in 1993 and suspended due to social considerations of closeness to the community and disturbance from blasting. It stands out as an isolated site with the area covered predominantly with *Acacia mangium*. The total dump surface area is 5.5 hectares with a height of 2m. The waste material is mainly phyllite. After suspension of operations, the dump was subsequently leveled to enable re-vegetation activities to commence in 1997. The land has attained maturity and currently encroached with community farms and portions used as a meeting ground for a community church. The predominant crop found at the farms are cassava and pepper. The proposed end-use for this dump is forestry development and covering 4.59 hectares (Abeiku, 2014).

Plate 1: A farm located at the Nhyiaso Re-vegetatively restored mined spoil


[redacted]

Plate 2: A church located at the Nhyiaso Re-vegetatively restored mined spoil

T_1 Re-vegetatively Restored Mined Spoil

This site is located adjacent the T_1 pit and close to the South tailings storage facility covering a total surface area of 8.3 hectares. Re-vegetation was done in 1999 with a variety of plant species dominated by *Acacia mangium.* Tree heights are approximately 9-10 m and diameter of 0.5 m. The site as similar to Nhyiaso has been encroached with community farms by clearing portions of the dump. Similar crops are cultivated. The slope angle is <18° as recommended by mining regulators for waste dump closure. The proposed end used for this dump is forestry development and covering 5.50 hectares (Abeiku, 2014).

Plate 3: T_1 Waste dump re-vegetated with varied stand of *Acacia mangium*

The potential source of heavy metal at the re-vegetated sites is the mine tailings which is a constituent of the substrate on which the plants grow. Nitrogen fixing plant such as *Acacia mangium* is predominantly used for the re-vegetation exercise. Previous top soils excavated prior to mining were kept for reclamation or re vegetation purposes. The mine spoils comprising the waste rock and tailings were used for the re-vegetation of the study areas. Currently, the top soil will be heaped and kept where it can simply be accessed and used during reclamation or re-vegetation.

Control Farm

A farm cropped with cassava and located across the main Obuasi-Dunkwa highway was used as a control for the study. The farm size was about 1.5 hectares and had no previous history of mining activities.

Experimental Design

The Completely Randomized Design was used for the study because earlier to the re-vegetation exercise the spoils were comprehensively mixed to ensure homogeneity. Five farms of about 0.3 ha each were randomly sampled from two re-vegetatively restored mined spoils namely Nhyiaso, and T_1. Each farm at a particular restored mined spoil served as a replication. A total area of 1.5 ha for each re-vegetatively restored mined spoil and was used for the purpose of this study. Both Nhyiaso (N) and the T_1 sites were re-vegetated with *Acacia mangium* as a predominant plant species. The unmined control site of about 1.5 ha was planted to cassava (C). The control cassava farm was partitioned in to 5 plots with each plot size of 0.3 ha. Each 0.3 ha plot also served as a replication at the control farm. Therefore a total of 4.5 ha was used for the study.

Soil sampling and Sample preparation

Soil sampling was done during the months of June-July at a plough depths of (0 – 15) cm using the hand trowel and a worm auger. Five composite soil samples were taken at each of the re-vegetated mined land and the control farm. They were placed in a polyethylene bag and transported to the Soil Science Laboratory, University of Cape Coast and Ghana Atomic & Energy Commission in Accra. Approximately an area of one and half (1.5) hectare of land used for farming at each re-vegetated site was used for the sampling purposes**.** These samples were air dried in shallow wooden trays in a well-ventilated place protected from rain and contamination. The soil lumps were crushed to remove gravels and roots. All soil samples were passed through a 2 mm sieve by gently rubbing the crumbs through the mesh leaving the gravels and roots in the sieve. A sample of approximately 600 g was retained by cone and quartering for each sample.

Soil Physical Properties

Soil moisture content

Soil moisture content was determined by weighing 10 g of soil sample (M_1) using CP 224S Sartorius Analytical Balance into an aluminium dish covered with a lid and oven dried using the Towcen & Mercer oven at 105^0C till constant weight. The aluminium dish with the lid was removed from the oven and kept in a desiccator for 30 minutes to cool. The oven-dried soil with the can covered (M_3) was weighed (ISO, 1993) and the moisture content of the soil was calculated.

Particle size analysis

Soil particle size analysis was carried out using the pipette method as described by Rowell (1994). An amount of soil (10g + 0.01) was weighed into a 500 ml beaker. 20 mL of hydrogen peroxide was added and allowed to stand until frothing ceased. The suspension was then heated to complete the destruction of organic matter, and then allowed to cool. The peroxide-treated soil was quantitatively transferred into a 500 mL plastic bottle and 10 ml of dispersing agent added and the soil suspension was made up to 200 mL and shaken overnight. The contents were then transferred quantitatively into a 500 mL measuring cylinder and made up to 500 mL with distilled water. The suspension was then stirred using a plunger for thorough mixing. The suspension was allowed to settle for 40 seconds after which 25 mL of the suspension was drawn off from 10 cm below the surface into a weighed beaker. This gives the mass of silt and clay. The suspension was allowed to settle and 25 ml of the suspensions were drawn off at 10 cm depth after 5h. This gives the mass of clay. The pipetted suspensions were dried at 105^oC till constant weight. Most of the supernatant liquid was gently poured off and the sediment was quantitatively transferred into a beaker. The sediment was repeatedly washed through stirring, settling and decanting till a clear supernatant was obtained. The sand was transferred to a weighed beaker and dried at 105^{o}C till constant weight.

Calculation:

Percentage sand (m/m) = $\frac{Mass\ of\ sand}{Mass\ of\ oven\ dry\ soil} \times 100$

The total mass of silt in the soil sample = Mass in $\frac{25mL \times 500}{25}$

Percentage silt = $\frac{Total\ Silt}{Mass\ of\ oven\ dry\ soil} \times 100$

The total mass of clay in the soil sample = Mass in $\frac{25\ mL \times 500}{25}$

Percentage clay = $\frac{Total\ clay}{Mass\ of\ oven\ dry\ soil} \times 100$

The textural classes of the soil samples were determined using the USDA textural triangle (Rowell, 1994).

Determination of bulk density and pore volume by the graduated cylinder method

An oven-dry soil to be used was prepared by passing it through a 10mm mesh sieve. A 100 mL graduated cylinder was weighed and the weight recorded. Enough sieved soil was added to the cylinder to fill it to near the 25 mL mark. The soil was compacted by dropping the cylinder 5 times from a height of 2 inches on several thickness of paper towels spread on the desk (the soil was compacted this way so that it will fill approximately the same volume that it occupied in a natural profile. Successive compaction of small increment gave more uniform compaction than if large volume was compacted all at a time. The paper toweling provided enough cushion to prevent breakage of the cylinder). More of the soil was added to the cylinder until it was filled to a depth that was near the 75 mL mark and the soil compacted again. More soil was added to the cylinder until it was filled to a depth that was near the 100 mL mark. The volume occupied by the soil was read and the value recorded. (this was the volume of the soil). The cylinder plus the soil was weighed and the weight recorded. (this weight minus the weight of the cylinder only gives the weight of the soil). The bulk density of the soil was calculated by the equation below;

Bulk density g cm^{-3} = weight of oven dry soil (g) / Bulk volume of soil (c m^{3}). Using a spatula some of the soil from the cylinder was removed until the level of the remaining compacted soil is exactly at level with the 50 mL mark. The soil from the cylinder was transferred to a weighed vessel and weighed together with the soil. Tap water was added to the graduated cylinder to the 50 mL mark. This established the initial volume. The weighed soil was returned to the cylinder in small increments with constant stirring. The suspension was stirred with a glass rod to remove air bubbles adhering to the soil particles. The final volume of the soil-water suspension in the graduated cylinder was read and the volume recorded. The volume increment over 50 mL was the volume of the soil particles. The difference between the final volume of the soil-water suspension in the graduated cylinder and the 100 mL mark was observed (which represented the combined volume of water plus particles plus pore space) (Black, et al., 1965).

Pore volume was calculated by the equation below;

$$\text{Pore Volume (\%)} = \frac{Pore\ Volume}{Bulk\ Volume} \times 100$$

Soil Chemical Analysis

Soil pH

The pH of the soils was determined using RE357 (Tx) pH meter. All pH measurements were performed using a 1:2.5 (w/v) soil to distilled water ratio by weighing 20 g of the 2 mm air-dried soil into a 100 ml glass beaker and adding 50 mL distilled water. The soil was mixed with the distilled water and stirred intermittently using stirrer B211 for 30 minutes before pH was determined (Anderson and Ingram, 1993).

Total organic carbon

One gram of 0.5 mm air-dried soil sample was weighed into 500 ml Erlenmeyer flask. Ten millilitres of 0.1667 M $K_2Cr_2O_7$ solution was added from a burette into each flask and swirled gently to disperse the soil and 20 mL conc. H_2SO_4 was added using an automatic pipette directing the stream into the suspension. The flask was swirled gently until the soil and reagents were mixed. The flask was rotated and allowed to stand on porcelain for 30 minutes. Approximately 200 mL of distilled water was added after standing for 30 minutes. Three drops of diphenylamine sulphonate indicator was added and titrated with 1M $Fe(NH4)_2(SO4)_2$ solution. As the end point was approached, the solution took on a greenish cast and then changed to dark green. A blank titration was made in the same way. (Bray & Kurtz, 1945).

Total nitrogen

0.5g of the dried, finely grounded soil sample was weighed into a 50 mL round bottom Kjeldahl flask. 2 g H_2SO_4-HgO mixture was added. 3 mL of conc. H_2SO_4 was added running the acid slowly down the neck while rotating the flask. The flask was gently heat on a digestion rack until frothing subsides.

As the frothing subsides, a glass bubble was placed in the flask mouth and the heat increased (H_2SO_4 will now reflux down the flask neck). After the digestion became colourless or pale green heating was continued for about 15 minutes and subsequently increased to 1hour. On completion of the digestion, it was allowed to cool until just warm and then diluted to 100 mL with distilled water.

A steam distillation apparatus was set up and a steam passed through it for about 20 minutes. After flushing out the apparatus, a 100 mL conical flask containing 5 mL of boric acid indicator solution was placed under the condenser of the distillation apparatus. An aliquot of the sample digest was transferred to the reaction chamber through the trap funnel. 10 mL of alkali mixture was added and distillation immediately commenced. About 40 mL of distillate was collected. The distillate was titrated against M/140 HCl from green to the initial colour of the indicator (wine red). Digestion blanks were treated the same way and subtracted from the sample titre value. (Rowell, 1994).

Calculation:

$$\text{N (\%)} = \frac{(\text{S}-\text{B}) \times \text{solution volume}}{10^2 \times \text{aliquot} \times \text{sample weight}}$$

Where;

S = Sample titre value

B = Blank titre value

Available phosphorus

One gram of 2 mm sieved air-dried soil was weighed into 50 mL extraction tube and 30 mL of extraction solution Bray 1 added. The mixture obtained was placed on a shaker, shaken for 3 minutes and filtered through

Whatwan No. 42 filter paper. Ten millilitres of the clear supernatant was pipetted into 50 mL volumetric flask and 5 mL distilled water added. Eight millilitres of Ascorbic Acid was added to each sample and waited for the blue colour formation. It was topped up to the 50 mL mark with distilled water. After 5 minutes the concentration of P was measured electrophotometrically at 660 nm wavelength using Perkin Elmer Lamda 45 ultra violet Spectrophometer (Watanabe & Olsen, 1965).

Exchangeable bases

Five grams of air-dried 2mm sieved soil was weighed into an extraction bottle and 50 ml of 1.0 *M* NH4OAc solution at pH 7.0. The bottle with its content was placed in a shaking machine and shaken for one hour. At the end of the shaking, the suspension was centrifuged at 3000 rpm for 5 minutes. The supernatant solution was filtered through No. 42 whatman filter paper. Twenty five (25) millilitres aliquots of the extract were used for the determination of Ca, Mg, Na, and K (Rhoades, 1982). Calcium and Magnesium were determined on the 210 VGP Atomic Absorption Spectrophotometer; and Potassium and Sodium by Jenway PFP7 Flame Photometer (Williams ,1949).

Exchangeable acidity

Ten grams of 2 mm sieve soil was weighed into a 100 mL plastic extraction bottle and 50 ml of 1.0 M KCl solution was added. It was shaken for 30 minutes and the suspension was filtered into an empty clean bottle. Twenty five (25) millilitres aliquot of the filtrate was pipetted into a 100 mL conical flask. Two to three drops of phenolphthalein indicator was added and titrated to a permanent pink end point using 0.01 *M* NaOH. The titre for NaOH was recorded. This represented both H^+ and $Al^{3+\cdot}$ The solution was brought back to colourless by adding one drop of 0.01 *M* HCl. Ten millilitres of NaF solution was added to complex aluminium. This was titrated with 0.01 *M* HCl to pink. The titre obtained represented exchangeable H^+.

Calculation:

The exchangeable Al^{3+} was determined by the difference between exchangeable H^+ and Al^{3+} (exchangeable acidity) and exchangeable H^+.

Exchangeable Al^{3+} = Exchangeable acidity - Exchangeable H^+.

Effective Cation Exchange Capacity: This was estimated by summation of exchangeable bases and exchangeable acidity.

Heavy Metals Extraction from Soil Samples

About 1.5 g of soil sample was weighed into a previously acid washed labeled 100 mL polytetraflouroethylene (PTFE) Teflon bombs. 6ml of concentrated nitric acid (HNO_3, 65%), 3 mL of concentrated hydrochloric acid (HCl,35%) and 0.25 mL of hydrogen peroxide (H_2O_2,30%) was added to each sample in a fume chamber. The samples were then loaded on the microwave carousel. The vessel caps were secured tightly using a wrench. The complete assembly was microwave irradiated for 26 minutes using milestone microwave lab station ETHOS 900, INSTR: MLS-1200 MEGA using the microwave programme. After digestion, the Teflon bombs mounted on the microwave carousel were cooled in a water bath to reduce internal pressure and allow volatilized material to re- stabilize. The digestate was made up to 20 mL with double distilled water and assayed for the presence of Zinc, Copper, Iron, Arsenic, Lead and Cadmium using VARIAN AA 240FS- Atomic Absorption Spectrometer in an acetylene- air flame (Harmsen, 1997).

Reference standards used for the elements of interest, blanks and duplicates of samples were digested the same conditions as the samples. These served as internal positive controls. The following Quality Control and Quality Assurance techniques were used during the analysis:

Blanks: They were to check contamination during sample preparation.

Duplicates: To check the reproducibility of the method used.

Standards: To check the efficiency of the equipment being used (Sabienë, & Rimmer, 2004).

Potassium was determined on the Jenway PFP7 Flame Photometer. The concentrations of calcium, magnesium, copper, zinc, iron, arsenic and lead were read on the Perkin Elmer A Analyst 400 Atomic Absorption Spectrophotometer. Total phosphorus was determined on the Spectrophotometer by the Ascorbic acid indicator method and Total Nitrogen using Boric acid indicator by titrimetric method. Total Carbon was determined using Potassium dichromate as described by Walkley and Black (1934).

Crop sampling and Sample preparation

Matured cassava tubers were randomly sampled from five farmers' fields at the two re-vegetated mined sites during the month of July, 2014. Below was the list of crops sampled from the various farms.

Oc- Cassava tubers from the control farm

Nc- Cassava tubers from farms located at Nhyiaso re-vegetatively restored mined spoil/land

Tc-Cassava tubers from farms located at T_1 re-vegetatively restored mined spoil/land

Harvested edible parts were placed in separate paper bags and dried at 70 ^{0}C for 24 hours in a Memmert oven to minimize chemical and biological changes. The dried edible parts were milled to pass through 0.3 - 0.5 mm sieve using Brook Crompton, 2000 series grinding mill. The samples were re-dried at 70^{o}C for 2 hours before analyzing for Cu, Zn, Fe, Mn, Pb, As and Cd.

Heavy metals extraction from crops sampled

About 0.5 g of milled cassava sample was weighed into a previously acid washed labeled 100 mL polytetraflouroethylene (PTFE) Teflon bombs.6ml of concentrated nitric acid (HNO_3, 65%), and 1mL of hydrogen peroxide (H_2O_2, 30%) was added to each sample in a fume chamber. The samples were then loaded on the microwave carousel. The vessel caps were secured tightly using a wrench. The complete assembly was microwave irradiated for 25 minutes using milestone microwave lab station ETHOS 900, INSTR: MLS-1200 MEGA using the microwave programme. After digestion the Teflon bombs mounted on the microwave carousel were cooled in a water bath to reduce internal pressure and allow volatilized material to re- stabilize. The digestate was made up to 20 mL with double distilled water and assayed for the presence of Zinc, Lead and Copper using VARIAN AA 240FS-Atomic Absorption Spectrometer in an acetylene- air flame. Reference standards used for the elements of interest, blanks and duplicates of samples were digested the same conditions as the samples. These served as internal positive controls.

Data Analyses

The data collected were subjected to statistical analysis using GENSTAT Statistical Package (Discovery edition 4) .Significance among treatments was at $p \leq 0.05$ and means separation was done using Least Significant differences (LSD).

CHAPTER FOUR

RESULTS AND DISCUSSION

The purpose of the study was to determine the effectiveness of re-vegetation process used to restore the mined spoils at the concession of AngloGold Ashanti Mine at Obuasi using cassava as a test crop. Soil and crop sampling were done randomly. Physico-chemical soil properties, heavy metals concentration in both the soil and crop were determined. GENSTAT Statistical Package (Discovery edition 4) was used for statistical analysis of the data. This chapter therefore presents the results and discusses the following: Selected physico-chemical properties of the soil, concentration of heavy metals both in soils and cassava tubers, translocation factors of heavy metals and finally, the partial correlation between heavy metals in soil and the cassava tubers.

Results:

Table 3: Selected physical properties of soils for various treatments at AngloGold Ashanti Obuasi Concession.

Treatment	Sand	Silt	Clay %	Texture	GWC	Pore Vol.	BD gcm^{-3}
C	39.53	43.62	16.85	loam	20.99	32.00	1.37
N	43.90	42.63	13.87	loam	26.57	50.40	1.09
T	31.33	47.02	22.05	loam	23.30	40.00	1.33
l.s.d$_{0.05}$	8.38	7.17	6.72		4.57	0.71	0.11
s.e.	2.72	2.33	2.18		1.48	0.23	0.04

C= Original unmined area used for farming, N= Nhyiaso re-vegetatively restored mined spoil, T= T_1 re-vegetatively restored mined spoil, GWC= Gravimetric moisture content, BD= Dry bulk density

Selected Physical properties of the experimental soils

The results of the selected physical properties of the study soils are presented in Table 3. The particle size analysis showed per cent sand, silt and clay content of the study soils to range from 31.33 to 43.90, 42.63 to 47.02, and 13.87 to 22.05 respectively. The Nhyiaso soil had significantly ($p \leq 0.05$) higher sand content than T_1 and vice versa for clay. The latter soil had significantly ($p \leq 0.05$) clay content than C and N. in spite of this, the texture of all the study soils was loam.

The implication is that observed differences among soils of the study sites may be mainly due to the variations in the measured parameter rather than the texture. The loam texture has generally been observed to be ideal for cultivation of most crops (Goose, 2005). However, the relative performance of crop growth on these soils will depend, among other factors, on the fertility levels. In general, the bulk density of the top 15 cm depth of the soils at three sites, ranging from 1.09 to 1.37 g cm^{-3} was ideal for plant growth, with a trend of C > T > N. on a relative basis, the Nhyiaso soil, which had undergone 17 years of re-vegetation, recorded a significantly ($p \leq 0.05$) the lowest bulk density. Although the bulk density of C and T did not differ significantly ($p \leq 0.05$), the re-vegetated sites (N and T) recorded lower bulk densities. The vegetative cover of the *Acacia mangium* and its accompanying litter fall, cushioning the soil against the impact forces of raindrops, may account for the reduced bulk density alongside the positive impacts of its ramifying rooting system and presumed presence of macrofauna, such as earthworms on soil aggregation and porosity (Agodzo & Adama, 2003). These same factors account for the recorded greater porosity and gravimetric water content at the re-vegetatively restored mined soils. The absence of these factors, there is a tendency for bulk density to increase, as observed at site C. In spite of the observed differences, bulk density at all the sites fell within recommended ideal values of 1.1 to 1.5 g cm^{-3} for productive soils (Sheoran et al., 2010). Within this range favourable total porosity, aeration, soil infiltrability and moisture storage provide a conducive edaphic physical conditions for improved plant growth.

The results (Table 3), further showed that as bulk density increased, even within ideal range, there was a corresponding decrease in the total porosity of the soil. The values ranged from 32.0 to 50.4 % for C and N respectively with a decreasing trend of N > T > C, which was the inverse of the trend in bulk density. The differences in bulk density among the soils at the three sites were significant ($p \leq 0.05$). The observed inverse trend is not surprising since the results (appendix IV) showed a significant negative correlation ($r = -0.95$) between bulk density and total porosity. From a base value of 50.40% at a bulk density of 1.09 g cm-3 at Nhyiaso, total porosity at T and C was reduced by 20 – 63% and 36.51% respectively. It has been observed that as total porosity of the soil is reduced, pore size distribution shifts towards smaller sizes and pore space continuity decrease to adversely affect water movement and availability, nutritional uptake and aeration (Sands et al., 1977). This underscores the importance of re-vegetation in improving the physical characteristics of the soil for enhanced productivity. As total porosity increased, so also did gravimetric water content with values ranging from 20.99% at c to 26.67% at N in the order of N>T>C. Whilst the difference between N and C was significant ($p \leq 0.05$), that between the latter sites was not. From a base value of 26.57% at a bulk density of 1.09 gcm-3 at site N, gravimetric water content at C and T decreased by 21% and 12.3% respectively. It appears that seventeen years of re-vegetation by *Acacia mangium* presents a minimum threshold period for observable significant changes in the soil's physical parameters examined.

According to Campbell and Henshall (2001), gravimetric soil water contents of air-dry (25°C) mineral soil are often less than 2 per cent, but as the soil approaches saturation, the water content may increase to values between 25 and 60 per cent, depending on soil type. Soil sampling was done during the months of June–July, 2014 when there was rainfall. Therefore, much moisture might have been added to the soils at the study area through atmospheric precipitation at the time of sampling. This could be the most probable reason for the general high moisture contents of the study area.

Table 4: Selected chemical properties of soil for various sites at AngloGold Ashanti Obuasi Concession

Site	Exchangeable				Ex. Acidity	ECEC	pH	Total		BS	Av.P
	K^+	Ca^{2+}	Mg^{2+}	Na^+				N	OC		
	$cmol_c\ kg^{-1}$							(%)			
C	8.75	13.31	17.19	1.74	16.39	57.38	6.39	0.44	3.87	40.99	106.04
N	7.26	12.72	12.49	1.57	13.61	46.64	5.60	0.19	1.62	34.04	19.29
T	7.39	12.58	12.62	0.83	13.37	46.78	6.16	0.29	2.91	33.42	28.68
$Lsd_{0.05}$	0.15	0.42	0.47	0.19	0.34	0.82	0.68	0.12	0.59		2.28
s.e.	0.05	0.14	0.15	0.06	0.11	0.27	0.22	0.04	0.19		0.74

C= Unmined area used for farming, N= Nhyiaso re-vegetated mined site, T= T_1 re-vegetated mined site

N = Nitrogen, OC=organic carbon, BS= Base saturation, Av. P= available phosphorus,

Soil Chemical Properties

Selected soil chemical properties of soils from the study area were presented in Table 4. The data revealed that the highest concentration of exchangeable potassium was recorded at the unmined area used for farming with a mean value of 8.75 $cmol_c$ kg^{-1}. Nhyiaso re-vegetatively restored mined spoil recorded the lowest concentration of exchangeable potassium with a mean value of 7.26 $cmol_c$ kg^{-1}. The mean exchangeable potassium concentration (Table 4) between the unmined area used for farming and the two re-vegetatively restored mined spoils was significant ($p \leq 0.05$). The T_1 re-vegetatively restored mined spoil had significantly ($p \leq 0.05$) higher exchangeable potassium content than the Nhyiaso re-vegetatively restored mined spoil. According to soil test interpretation guide provided by Edward et al. (2012) exchangeable potassium values of; < 175, 175-280, and 280-800 mg kg^{-1} were described as low, adequate and high respectively. Based on this description, the concentration of exchangeable K can be said to be low. The low levels of exchangeable potassium observed at the re-vegetatively restored mined spoils could be attributed to high precipitation intensities which caused the leaching of the cation. The general low levels of organic carbon (Table 4) at the re-vegetatively restored mined spoils could have favoured the leaching of the potassium ion. Organic carbon content of the soil reflects the amount of organic matter present in the soil and also serves as a source for macro nutrients such as nitrogen, phosphorus, potassium and sulphur in addition to some micro nutrients.

The highest and lowest mean concentrations of exchangeable calcium were also recorded at unmined area used for farming and T_1 re-vegetatively restored mined spoil with values of 13.31 $cmol_c$ kg^{-1} and 12.58 $cmol_c$ kg^{-1} respectively. The mean concentration of exchangeable calcium observed at the unmined area used for farming was significantly ($p \leq 0.05$) higher than that of the re-vegetatively restored mined spoils. The differences in exchangeable Ca^{2+} can be attributed to variation of leaching losses, plant up take and the rate of organic matter decomposition at the study area. Edward et al. (2012) described exchangeable Ca^{2+} levels below 1000 mg kg^{-1} as low and therefore same can be said of the exchangeable Ca^{2+} levels at the study area.

Whilst the unmined area used for farming showed 17.19 $cmol_c$ kg^{-1} as the highest mean concentration of exchangeable magnesium, the Nhyiaso re-vegetatively restored mined spoil recorded the lowest concentration of 12.49 $cmol_c$ kg^{-1}. It can be inferred (Table 4) that the exchangeable magnesium at the unmined area used for farming was significantly higher ($p \leq 0.05$) than that of the re-vegetatively restored mined spoils. Surprisingly, the concentration of exchangeable magnesium followed the same trend as exchangeable potassium. Level of exchangeable Mg^{2+} can also be described as very low as compared to the <60 mg kg^{-1} as postulated by Edward et al. (2012).

The highest mean exchangeable sodium concentration of 1.74 $cmol_c$ kg^{-1} was observed at the unmined area used for farming. The lowest concentration value of 0.83 $cmol_c$ kg^{-1} was observed at T_1 re-vegetatively restored mined spoil. The mean exchangeable sodium concentration (Table 4) of the unmined area used for farming was significantly ($p \leq 0.05$) higher than that of the T_1 re-vegetatively restored mined spoil. There is however no significant difference between the mean concentration of exchangeable sodium for both Nhyiaso and T_1 re-vegetatively restored mined spoils. The acidic nature of the study area could be a major contributing factor for the low exchangeable sodium recorded. It was interesting to note that (Table 4) exchangeable sodium levels in the soil at the study area was the least. General low levels of the basic cations (K^+, Ca^{2+} Mg^{2+} Na^+) at the study area can be attributed to both acidification and leaching losses as reported by Bailey et al. (2005) who indicated that acidification usually leads to depletion of base cations because of the exchange between H^+ and the bases. Owusu-Bennoah et al. (2000) also stated that low basic cation in most soil were due to plant uptake of exchangeable bases and leaching.

Meanwhile, the highest mean exchangeable acidity concentration was observed at unmined area used for farming and the lowest concentration was witnessed at the T_1 re-vegetatively restored mined spoil with values of 16.39 and 13.37 $cmol_c\ kg^{-1}$ respectively. The exchangeable acidity at unmined area used for farming was significantly ($p \leq 0.05$) higher than the re-vegetatively restored mined spoil. That of T_1 and Nhyiaso re-vegetatively restored mined spoil were however the same. The exchangeable acidity (Al^{3+} and H^+) followed a similar trend as pH (Table 4). This conforms to the observation made by Oyedele et al. (2009) who also found that soil exchangeable acidity (Al^{3+} and H^+) followed a pattern similar to pH showing that Al^{3+} contributed most to the acidity of the soil. There was a direct relationship between pH and the exchangeable bases.

The concentration of ECEC ranged from 57.38 $cmol_c\ kg^{-1}$ to 46.64 $cmol_c\ kg^{-1}$. The results (table 4) showed that, the ECEC of the unmined area used for farming was significantly ($p \leq 0.05$) higher than that of the re-vegetatively restored mined spoils. The high ECEC values recorded at the study area could easily be attributed to the combine low effect of the basic cations (Table 4) and also the high values of the exchangeable acidity (Al^{3+} and H^+) of the study area.

The pH of the study area was generally acidic. However, using the descriptive ranges of soil pH as stated in Soil Survey Division staff (1993), except for the Nhyiaso re-vegetatively restored mined spoil which was moderately acidic, the rest of the sites were slightly acidic. Typical of mined soils, the re-vegetatively restored mined spoils had low pH values as compared to the unmined area used for farming (Table 4). The un-mined area therefore registered a relatively higher pH values than the re-vegetatively restored mined spoils. The low levels of both exchangeable cation and organic carbon might have accounted for this observation. More importantly was the accumulation of aluminium from the mine spoil due to mining of the Orthic-Ferric Acrisol at the re-vegetatively restored mined spoils. Furthermore, disturbance of the soil through mining usually increases soil aeration which could enhance oxidation of minerals such as arsenopyrites to sulphuric acid to reduce soil pH.

Data from Table 4 show that, the maximum mean total nitrogen was observed at unmined area used for farming with 0.44%. The lowest was observed at the Nhyiaso re-vegetatively restored mined spoil with a value of 0.19%. The mean total nitrogen observed at the unmined area used for farming was significantly higher ($p \leq 0.05$) than that of the re-vegetatively restored mined spoils. This observation is not surprising due to the fact that *Acacia mangium* trees that had the potential to fix nitrogen in the soil were felled to allow for farming. There was however no significant difference between the re-vegetatively restored mined spoils with regards to the mean total nitrogen contents. Edward et al. (2012) described soil total nitrogen per cent ranges of 0.05-0.13% as low, 0.13-0.23% adequate and 0.23-0.30% as high. It can therefore be inferred that, the mean total nitrogen per cent of the soils from the study area were high except for Nhyiaso re-vegetatively restored mined site which was adequate.

The mean total organic carbon of the study area showed the highest value of 3.87% at the unmined area used for farming. The lowest occurred at the Nhyiaso re-vegetatively restored mined spoil with a value 1.62%. The mean total organic carbon observed at the unmined area used for farming was significantly higher ($p \leq 0.05$) than that of the re-vegetatively restored mined spoils. This could be due to the high accumulation and slow decomposition rate of plant residue at the unmined area used for farming as compared to the restored mined spoils. The low content of soil organic carbon observed at the restored mined spoils clearly reflects the impact of mining on degradation with attendant loss of soil carbon. Another reason might be due to rapid decomposition rate at the sites and the indiscriminate rate of felling of *Acacia mangium* tree species used for the re-vegetation to create space for farming. This will intern reduce the amount of litter fall on the soil surface.

The mean organic carbon content at the Nhyiaso (1.6%) comparative to the T_1 (2.91%) restored mined site is noteworthy. It should be noted that preferred results of accumulated carbon will be achieved only when re-vegetated trees are kept continuously growing. Although re-vegetation works were implemented 17 and 15 years ago at both Nhyiaso and T_1 sites, the trees on the T_1 site were maintained while those at Nhyiaso were cut early and the soil cultivated to food crops using traditional methods with no external nutrients inputs. In this respect, the gap in carbon content between T_1 and Nhyiaso re-vegetatively restored mined spoils could be attributed to nutrient mining by cultivated crops. It is therefore necessary that for the desired effect of re-vegetation to be achieved clear guide lines must be established on the minimum lifespan of re-vegetated trees that must pass before some trees are cut to allow for farming. This conforms to the observation made by Witmann et al. (2012).

The mean total organic carbon observed at the T_1 re-vegetatively restored mined spoil was also significantly higher ($p \leq 0.05$) than that of the Nhyiaso re-vegetatively restored mined spoil. This observed difference could be due to the early felling of the tree species at the Nhyiaso restored mined spoil to allow for farming. Organic carbon content of the soil reflects the amount of organic matter present in the soil and also serves as a source for macronutrients such as nitrogen, phosphorus, potassium and sulphur in addition to some micronutrients. Bonsu et al. (1999) and Arun (2008) stated that in the tropics organic matter makes up 3-5 % of the total soil mass and is located within 15 cm of the top soil. According to Sheoran et al. (2010) a level of organic carbon greater than 0.75% indicates good fertility. John et al. (2009) revealed that whilst there no soil organic carbon level that affects soil function, observation of the functionality of the soils below 2% soil organic carbon, in terms of crop production would be a useful indicator of unsustainable farming practices and prompt to adjust management or tillage practices. It can be inferred that, the soils at the re-vegetatively restored mined sites might have good fertility, but due to the closeness of the organic carbon contents to 2%, soil amendment practices might be required to make the soils productive.

Base saturation is a measurement that indicates the relative amounts of base cations in the soil. By definition, it is the percentage of calcium, magnesium, potassium and sodium cations that make up the total cation exchange capacity (Rayment & Higginson, 1992). From Table 4 above, the highest value of 40% was observed for the unmined area used for farming and lowest (33.42%) from the Nhyiaso re-vegetatively restored mined spoil. The general low level of base saturation at the re-vegetatively restored mined spoil suggests that more of the cation exchange capacity might have been occupied by acid cations, such as hydrogen and aluminum. The pH of soil increases as base saturation increases (Brady & Weil, 2002)

Phosphorus accumulation was highest (106.04 $\mu g\ g^{-1}$) in the unmined area used for farming. This was significantly higher ($p \leq 0.05$) than that of the re-vegetatively restored mined spoils. The available phosphorus observed at the Nhyiaso re-vegetatively restored mined spoil was also significantly lower ($p \leq 0.05$) as compared to the T_1 re-vegetatively restored mined spoil. The relatively low phosphorus values obtained at the Nhyiaso re-vegetatively restored mined spoil gives an indication of the formation of insoluble phosphates compounds.

Another important factor may be attributed to the low pH value of 5.6 recorded at the re-vegetatively restored mined spoil. Aluminium goes through hydrolysis under acidic condition which then reacts with orthophosphate ions to form a precipitate known as aluminium–hydroxy phosphate which does not allow the release of phosphorus in the soil (Kwakye, 2013). In furtherance, the lowest total organic carbon value of 1.62% observed at the spoil could also be a contributing factor. Organic carbon content of the soil reflects the amount of organic matter present in the soil. Therefore soils with low organic matter will not be able to produce enough organic acids which solubilizes the inorganic phosphate compound in the soil and thereby not making phosphorus available. Ultimately, soil with low organic matter cannot also coat the surfaces of the soil minerals to prevent the sorption of phosphorus and therefore making phosphorus unavailable.

The relationship between selected physico-chemical properties of the experimental soils

Correlation studies was undertaken to investigate the relationship between selected physico-chemical soil properties and presented in appendix V. The results of the correlation studies showed that (appendix V), generally, there exist a moderate significant negative relationship between gravimetric soil moisture content and nitrogen, potassium, magnesium, manganese as well as soil organic carbon. This inverse relationship meant that increase in soil moisture content will result in a slightly decrease in the ions of the elements of concern. The negative relationship between soil moisture and nitrogen could be explained by the combine effects of de-nitrification of nitrate-N and leaching. Certain soil bacteria that thrive in saturated (anaerobic) soil conditions will convert nitrate-N to oxygen and nitrogen gases. Volatilization of the nitrogen gas can result in N losses of as much as 5% of the available nitrate-N per day. Soils at greatest risk to de-nitrification N loss are those that are naturally heavy and poorly drained, plus fields with significant levels of soil compaction that restricts natural drainage. The nitrate ion is very mobile and therefore can easily be leached by increased soil moisture (FAO, 2005).

The inverse relationship between soil moisture and potassium (appendix V) might be explained by the mobility parameter as described by Kwakye (2013). The mobility factor of an element is the speed with which the ion of the said element travels from the point of discharge to the absorbing roots. Therefore significant increase in soil moisture content at full saturation point might increase its mobility down the soil profile through percolating water. This observation is in contrast with the findings of Kuchenbuch et al. (1986), who saw increased K mobility in soil with soil moisture. Another important reason to explain the inverse correlation between soil moisture and potassium, magnesium as well as manganese is the fact that the ions of these elements are considered basic cations. Under high moisture conditions basic cations such as potassium, magnesium and manganese are highly leached. When there is sufficient moisture in the soil for leaching to occur, the percolating water can carry dissolved nutrients which will be subsequently lost from the soil profile. The nutrients which are easily leached are usually those nutrients that are less strongly held by soil particles. For instance, in a soil with a high cation exchange capacity and low anion exchange capacity, nitrate (an anion) will leach much more readily than calcium (a cation) potassium (a monovalent cation) will leach more readily than calcium (divalent cation) since calcium is more strongly held to the soil particles than potassium. According to Laboski and Peters (2012), oxygen is necessary for root respiration and potassium uptake. Therefore, factors such as soil compaction and excess moisture that reduce oxygen levels in the root zone decreases potassium uptake. The relationship between gravimetric soil moisture content and soil organic carbon showed a strongly significant inverse correlation (r= -0.800). This result conforms to the study of Jun-Hua et al. (2007) who stated that increased in soil drainage was a major factor in reducing soil organic matter decomposition. Similarly, Alberti et al. (2011) found that carbon losses in wetter sites of their study was as a result of decreased soil nitrogen stocks due to leaching. The inverse relationship between soil moisture and soil organic carbon however was in contrast with the findings of the FAO, (2005).

Effect of clay content on soil bulk density was found to be higher than that of sand and silt contents. Whilst a weak positive correlation of bulk density was observed with clay (r = 0.419) and silt contents(r = 0.220), a weak inversely correlation was observed with sand (r= -0.468). It can be inferred that the impact of the texture (clay) on bulk density was not too much, because of the low organic carbon. Wagner et al. (1994) estimated soil bulk density using soil texture parameters along with organic carbon content values. Bernoux et al. (1998) found a correlation between texture and bulk density. Dinesh et al (2009) indicated that soil texture specific tests would be required to determine the correct organic matter level to achieve a target bulk density to avoid the problem of compaction.

The bulk density indirectly provides a measure of the soil porosity. Soil porosity is the ratio of the volume of soil pores to the total soil volume. Thus the bulk density of a soil is inversely related to the porosity. Unsurprisingly, Appendix V revealed a strong significant negative correlation (r = - 0.951) between porosity and bulk density of soil samples. Bulk density can be changed by management practices that affect soil cover, organic matter, soil structure, compaction, and porosity. Excessive tillage destroys soil organic matter and weakens the natural stability of soil aggregates making them susceptible to erosion caused by water and wind. When eroded soil particles fill pore space, porosity is reduced and bulk density increases. Tillage prior to planting temporarily decreases bulk density on the surface but increases at the depth of tillage. Subsequent trips across the field by farm equipment, rainfall events, animals, and other disturbance activities will also compact soil. A soil’s available water capacity is also affected by organic matter and compaction. Organic matter increases a soil’s ability to hold water, both directly and indirectly. Compaction increases bulk density and reduces total pore volume, consequently reducing available water holding capacity (USDA).

Many researchers (Sakin et al. 2011; Catherine and Ouimet, 2007; Morisada et al. 2004 and Leifield et al. 2005) observed the relationship between organic matter and bulk density of soils and showed strong correlation between them. However, Curtis and Post (1964) observed a negative correlation between organic matter and bulk density. This study interestingly showed strong positive correlation (r = 0.878) between organic matter and bulk density of soil samples. Thus the study indicate that as the organic matter increases the bulk density of soil also increases.

The relationship between bulk density, available primary (N, P, K) and secondary (Ca, Mg) macronutrients in the soil was observed. It was found that the bulk density increases as the primary macronutrient contents in the soil increases (appendix V). Statistical correlation studies showed a moderate significant positive correlation of bulk density with exchangeable potassium (r= 0.664), available N (r=0.688) and P (r=0.685). Whilst the secondary macronutrient (Mg) showed a moderate positive correlation (r= 0.623) with bulk density, calcium revealed a weak positive relationship (r= 0.329).

Similar to that of the macronutrients, the available micronutrients (Fe, Mn, Zn, Cu) in the soil also varied with soil bulk density. It was found that the bulk density increased as Fe, Mn and Cu contents in the soil increased. It is only Zn that showed a weak inverse correlation with bulk density. This result is at variance with Laiho et al. (2004) who reported that the concentration of Fe and Zn decreased in the same magnitude with increase in bulk density. Correlation between bulk density and pH (r = 0.363) was positive but weak. However, Shaffer (1998) observed highest correlation between pH and bulk density at 0 to 15 cm, but he did not indicate the reasons.

The relationship between pH and available primary (N, P, K), secondary (Ca, Mg) macro and micro (Fe, Mn, Zn, Cu,) nutrients in the soil were observed. pH showed a moderate positive relation with nitrogen (r=0.611) and phosphorus (r= 0.508). It however indicated a weak positive correlation with potassium (r= 0.490). These observations might be due to the moderate to slightly acidic nature of the study area. Most nutrients (macronutrients especially) tend to be available when soil pH approaches neutral. The relationship between pH and secondary macronutrients (Ca Mg) was weak. This weak relationship might be due to the low levels of organic carbon recorded at the study area. The moderate inverse relationship between pH and Cu (r=0.564) shown in appendix V may be partly due to high levels of iron and zinc. This conforms to the assertion of Landon, (1991). Silver and

Uchida (2000) also postulated that copper availability decreases as pH increases, primarily due to decreased solubility of copper minerals. A moderate positive correlation between pH, Fe and Mn was also observed.

The strong significant positive correlation between organic carbon and macronutrients; nitrogen (r=0.771), phosphorus (r= 0.811), potassium (r= 0.818) was not surprising. This observation is consistent with the general assumption that the decay and decomposition of organic matter releases nutrients to the soil, and probably the main source of the macronutrients at the re-vegetated mined spoil was soil organic carbon.

Table 5: Heavy metals in soil (0-15 cm) for various sites at AngloGold Ashanti Obuasi concession

Site	As	Cd	Cu	Fe	Mn	Pb	Zn
				mg kg^{-1}			
C	1.15	ND	33.32	210.68	52.93	4.06	44.88
N	0.98	ND	31.56	206.64	11.62	3.35	42.13
T	0.36	ND	34.50	213.45	56.11	3.13	43.04
l.s.d. $_{0.05}$	0.30	*	0.75	2.40	1.94	0.09	3.05
s.e.	0.09	0.00	0.25	0.80	0.65	0.03	1.02

C= Unmined area used for farming, N= Nhyiaso re-vegetatively restored mined spoil, T= T_1 re-vegetatively restored mined spoil

Heavy Metals in Soils

Arsenic, cadmium, copper, iron, magnesium, lead and zinc

The results of the extractable heavy metal determination revealed that iron accumulation was the highest with mean values ranging from 213.45 to 206.64 mg kg^{-1}. The T_1 re-vegetatively restored mined spoil, was significantly ($p \leq 0.05$) higher than the Nhyiaso re-vegetatively restored mined spoil. This may be attributed to the 17 years of re-vegetation at Nhyiaso as compared to the 15 years at T_1. It can be inferred that iron uptake by the *Acacia mangium* trees were higher at Nhyiaso as compared to that of T_1. Therefore the more the years of re-vegetation the more the uptake and accumulation of iron by plants above the concentration in soil. The high iron levels at the study site conforms to the findings of Golow et al. (1996) who indicated that the gold ore which occurs at Obuasi is made up of iron and the combined forms of minerals such as pyrites (FeS) and arseno-pyrites (FeAsS). The accumulated iron levels at the study sites were however within the critical range (20-2500) mg kg^{-1} for rain fed low land rice cultivation as stated by Abah et al. (2012).

The extractable zinc followed iron in magnitude and the mean values ranged from 48.19 to 42.13 mg kg^{-1}. Higher concentration of this element was found in the unmined area used for farming. There is however no significant ($p \leq 0.05$) difference between unmined area used for farming and the re-vegetatively restored mined sites. The zinc concentration at the study area however, was relatively lower than the phytotoxicity level (100 – 300) mg kg^{-1} as stated by Kabata- Pendias, (2011); Alloway (2011). It can be inferred from Table 5 that high concentration of zinc in all the treatment corresponded to lower concentrations of copper. This observed relationship could be explained by the fact that higher concentrations of zinc in the soil solution, relative to copper, can reduce the availability of copper to a plant (and *vice versa*) due to competition for the same sites for absorption into the plant root. This could occur after the application of a copper fertilizer (Alloway, 2008).

The highest extractable manganese was found at T_1 re-vegetated mined spoil with a concentration of 56.11 mg kg^{-1}. Significant difference exists between the sites. It can therefore be inferred (Table 5) that concentration of Mn at the study area was far below the maximum concentration levels (100–4,000) mg kg^{-1} for a typical uncontaminated soil. Kabata- Pendias (2011), pointed out that manganese contents of worldwide soils vary from 411 to 550 mg kg^{-1} and its highest levels occur mostly in calcareous soils.

Meanwhile, the highest and lowest copper accumulation were in the range of 34.50 and 31.56 mg kg^{-1}. These were observed at T_1 and Nhyiaso re-vegetatively restored mined spoils respectively. Similar to manganese, there exist a significant ($p \leq 0.05$) difference between all the sites. It can therefore be inferred from the Table 5 that concentration of copper at the study area was below the maximum concentration levels (60–150 mg kg^{-1}) for a typical uncontaminated soil as depicted by Table 1. The copper concentrations at the study area were below the phytotoxicity range as noted by (Schulte and kelling, 1999). However, Cu concentration was above the required amount for plant nutrition. This was explained by Adriano (2001) who indicated that Cu is an essential micronutrient for plant nutrition; however it is only required in small amounts (5 to 20 mg kg^{-1}).

Lead concentration was highest at the unmined area used for farming with a mean concentration of 4.06 mg kg^{-1}. The lowest concentration of 3.13 mg kg^{-1} was however recorded at the T_1 re-vegetatively restored mined spoil. The lower concentration of the lead at the re-vegetatively restored mined spoils as compared to the unmined area used for farming may be due to the uptake of lead of the *Acacia mangium* trees from the spoil. Mean lead concentration was however significantly different among the treatment. This difference may be partly due to the different levels of lead minerals in the parent material from which the soil was formed. It can also be inferred (Table 5) that mean lead concentrations at the study area were far below the phytotoxic level (100 -1000) mg kg^{-1} for plants (Adriano, 2001; Kabata-Pendias, 2011).

The results of the extractable heavy metals concentration revealed that arsenic accumulation was the lowest with mean values ranging from 1.15 to 0.36 mg kg^{-1} respectively. The Nhyiaso re-vegetatively restored mined site was significantly ($p \leq 0.05$) higher than the T_1 re-vegetatively restored mined sites. Generally, the arsenic concentration at each treatment sites were very low and below the recommended soil arsenic maximum allowable concentration range as noted by Kabata- Pendias, 2011) who explained that soil arsenic concentration above 20 mg kg^{-1} may cause toxicological risks. Surprisingly, the lower concentrations of arsenic recorded at the study area does not support the research results of Aubynn (1997) who pointed out that arsenic emissions at (AngloGold Ashanti) Obuasi, are 1000 times higher than any world standards. Cadmium contents at the study area were below the minimum readable limits of the equipment used.

Table 6: Level of Heavy metals in cassava tubers at the study area (mg kg^{-1})

Site	As	Cu	Fe	Mn	Zn
C	ND	0.71	5.32	2.02	8.26
N	0.13	0.75	13.01	1.19	8.46
T	0.15	0.67	5.86	1.25	9.67
l.s.d $_{0.05}$	0.02	0.07	0.28	0.11	0.48
s.e	0.01	0.02	0.09	0.04	0.17

C= Unmined area used for farming, N= Nhyiaso re-vegetatively restored mined spoil, T= T_1 re-vegetatively restored mined spoil, ND= No detection

Extractable Heavy Metal from Cassava tubers Sampled

Arsenic, cadmium, copper, iron, magnesium, lead and zinc

Table 6 shows the mean concentration levels of arsenic found in the cassava tubers sampled from the study area. Accumulation of arsenic in cassava tubers from the T_1 re-vegetatively restored mined spoil was the highest with a mean concentration value of 0.15. The lowest concentration of 0.13 was found at Nhyiaso re-vegetatively restored mined spoil. Mean arsenic level found in cassava tubers from the T_1 re-vegetatively restored mined spoil was significantly higher than that of Nhyiaso re-vegetatively restored mined spoil. This observed difference might be due to the differences in the number of years used in the restoration process. Although *Acacia mangium* was used at the predominant plant species at both re-vegetatively restored mined spoil Nhyiaso restored spoil was 2 years older as compared to that of the T_1 restored spoil. There is therefore the likelihood of low arsenic concentrations at the Nhyiaso re-vegetatively restored mined spoil as compared to the T_1 restored spoil. This might have reduced the subsequent transfer of the arsenic to the cassava tubers. However, arsenic levels in the cassava tubers from the original unmined area used for farming indicated no detectable concentration.

It can be inferred (Table 6) that cassava tubers from both the Nhyiaso and T_1 re-vegetatively restored mined spoils accumulated low arsenic as compared to that of 1.03 mg kg^{-1} provided by Amono-Neizer and Amekor (1993). Although, concentration of arsenic in the cassava tubers were very low as compared to the values proposed by Amono-Neizer and Amekor (1993), continuous consumption of food crops contaminated with this element is lethal when accumulated in the body system.

The mean copper concentration in cassava tubers from T_1 re-vegetated mined spoil was significantly ($p \leq 0.05$) than that of both Nhyiaso re-vegetatively restored mined spoil and unmined area used for farming. Generally, considering the re-vegetated mined spoils, it can be inferred from (Table 6) that copper accumulation was highest (0.75mg kg^{-1}) in cassava tubers from the Nhyiaso re-vegetatively restored mined spoil. The lowest mean copper accumulation occurred in cassava from the T_1 re-vegetatively restored mined site. This difference might be due to the differences in cassava variety cultivated in the area. The mean copper concentration values observed (Table 6) were far below the threshold of 4 -15 as posited by Alloway (2011).

Iron in Table 6 recorded the highest accumulation (13.01 mg kg^{-1}) in cassava tubers from the Nhyiaso re-vegetatively restored mined spoil. The lowest accumulation (5.32 mg kg^{-1}) was however observed in cassava tubers from the unmined area used for farming. Iron accumulation in cassava tubers was significantly different in all the sites. This observation might be due to the fact that, cassava planted at each of the re-vegetated mined spoil was from different varieties or cultivars. A strange phenomenon was observed between the concentration of iron in soil (Table 6) and the accumulation of iron in cassava tubers. Iron content in soil was highest at the T_1 re-vegetated mined spoil but surprisingly was the least accumulated by cassava tubers from the same site. This was partly due to the low presence of iron in solution which made it too difficult for the roots of the cassava to extract from the soil. The mean iron concentration in the cassava tubers were extremely below the mean concentration value of 48 mg kg^{-1} proposed by Kabata-Pendias, (2011).

The accumulation of manganese in Table 6 was highest (2.02 mg kg^{-1}) in cassava tubers from the unmined site and lowest (1.19 mg kg^{-1}) from the Nhyiaso re-vegetatively restored mined spoil. Mean manganese concentration in cassava tubers differ significantly ($p \leq 0.05$) between the sitess. It can therefore be inferred (Table 6) that the mean concentration of manganese in all the crops were below the range (15-100) mg kg^{-1} stated by Alloway (2011).

From Table 6, accumulation of Zn in cassava tubers from the T_1 re-vegetatively restored mined spoil was the highest (9.67 mg kg^{-1}). The lowest accumulation was found in cassava tubers from the unmined area used for farming. Concentration of Zn in cassava tubers from the T_1 re-vegetatively restored mined spoil was significantly higher than the concentration found in the one from the Nhyiaso re-vegetatively restored mined spoil. Zn concentration reported for the crops were within the permissible limits (10-15) mg kg^{-1} for human consumption (Samara et al., 1992). Unfortunately both cadmium and lead contents in the edible parts of crops were below detection.

Transfer of Heavy metal from soil to cassava tubers

Table 7: Translocation factor values for Iron in cassava

H.M	Treatment	T.F
	C	0.025
Fe	N	0.063
	T_1	0.027

H.M= Heavy metal, T.F =Translocation factor, C= Original unmined area used for farming, N= Nhyiaso re-vegetatively restored mined site, Fe= Iron $T_=$ T_1 re-vegetatively restored mined site

The translocation factor, or uptake factor, is an index for evaluating the transfer potential of a metal from soil to plant. From Table 7 above, one can observe that translocation factor values for iron in cassava tubers from the

Nhyiaso re-vegetatively restored mined spoil (0.063) was highest, followed by that of T_1 re-vegetatively restored mined spoil (0.027) and unmined area used for farming (0.025) . This observation might be due to fact that, there was more iron presence in the soil solution of the re-vegetatively restored mined spoil as a result of previous mining activities undertaken than that of the unmined area used for farming. The translocation factor values for arsenic, copper, iron, manganese and zinc of cassava tubers are listed in appendix VI. According to Olowoyo et al. (2010), if the ratios >1, the plants have accumulated elements, the ratios around 1 indicate that the plants are not influenced by the elements, and ratios < 1 show that plants exclude the elements from the uptake. If the plants have higher TF values, they can be used for phytoremediation (Saglam, 2013).

It can be observed (appendix VI) that all the heavy metals had translocation factor ratio < 1. One can therefore infer that, the cassava crops/plants excluded the elements from the uptake and therefore cannot be used for phytoremediation. It is also vital to note that the immobility nature of iron, copper, manganese, and zinc in plants may have contributed to their low levels in the cassava tubers grown at the studied area as stated by Silva & Uchida, (2000).

Table 8: Correlation coefficient between heavy metals in soils and cassava tubers from the study area

	Fe2	Zn2	Cu2	Mn2	AS2
Fe	0.744**	0.171	0.042	-0.823**	0.678**
Zn	-0.071	0.442	0.084	-0.303	0.519*
Cu	0.765**	0.253	0.305	-0.802**	0.770**
Mn	0.604*	0.510	0.001	-0.984**	0.929**
As	-0.902**	0.570*	-0.263	0.361	-0.055

Heavy metals with numbers were from Cassava tubers and vice versa

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

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

Correlation between iron, zinc, copper, magnesium and arsenic in soil and in cassava tubers

From Table 8 above, there exist a strong significant positive correlation between iron in soils and that in the cassava tubers. This relationship observed could be due to presence of soluble iron in the soil solution and subsequent translocation in to the cassava tubers. Therefore, increase concentration of iron in the soil might correspond to increase accumulation in cassava tubers. This could occur only when the soil iron is not translocated much to other parts of the crop. Zinc and copper in soils also correlated positively but weakly with their corresponding levels in the cassava tubers. Therefore increase in soil zinc, and copper might not cause the same increase in magnitude of the elements in the cassava tubers. This could be due to the translocation of the elements to other parts of the crops other than to the edible parts. This assertion conforms to that of Saglam, (2013), who stated that some root crops grown on contaminated soils accumulate high concentrations of heavy metals in their various parts. Once these metals are taken by the plants, they then accumulate in different tissues of the plants. This continuous uptake and translocation can increase the concentrations of metals in plant tissues.

However, there was a strong significant negative correlation between manganese in soils and that in the cassava tubers and therefore any incremental increase in soil manganese might result in the decrease of manganese in the cassava tubers. This inverse relationship might be due to the insoluble form of manganese in solution and the inability of the crops to absorb the soil manganese. There was also a weak inverse relationship between arsenic in soil and that accumulated in the cassava tubers from the study area. Saglam (2013), indicated that if there is a strong correlation between elements abundant in the soil and elements existing in plants, one may concludes that those elements found in plants may come from the soil and vice versa. From the correlation coefficients (Table 8), it can be inferred that, manganese in soils and manganese in cassava tubers correlated stronger, followed by iron, zinc, copper, and arsenic in that order.

Summary of the Key findings

From the results and discussion the following key findings were deduced:

1. Except for the presence of low basic cations, soil physico-chemical properties were adequate to support crop production.
2. The concentration of heavy metals in the re-vegetatively restored mined soils were within phytotoxic limit.
3. Heavy metal concentration in the cassava tubers were within their threshold levels.

4. Cassava crop excluded the elements from its uptake and therefore cannot be used for phytoremediation.

CHAPTER FIVE

SUMMARY, CONCLUSIONS AND RECOMMENDATION

Summary and Conclusions

A study was conducted to assess the physico-chemical properties of soil, heavy metal concentration in soils and in cassava tubers cultivated at re-vegetatively restored mined spoil at AngloGold concession Obuasi. A farm away from the mining area and cropped with cassava served as a control. Physical and chemical soil properties were estimated. Concentration of heavy metals both in re-vegetatively restored mined spoil and in the cassava tubers were also evaluated to determine their toxicity levels.

1. Soil physical properties of the re-vegetatively restored mined spoil showed that the soils were loamy textured, well aerated with a total porosity of 40% and also with ideal bulk density ≤ 1.4 g cm^{-3}. This was adequate to support crop production.
2. Low pH values (5.6 - 6.16) coupled with low organic carbon (1.62% - 2.91%) accumulation might have enhanced leaching losses which may have contributed to the general low level of basic cations (K^+, Ca^{2+} Mg^{2+} Na^+) and the availability of total nitrogen and available phosphorus contents of soil from the re-vegetatively restored mined spoils.
3. The maximum concentrations of As, Cu, Fe, Mn, Pb and Zn in the re-vegetatively restored mined spoils (1.15, 35.41, 213.45, 56.11, 4.06, 48.19) mg kg^{-1} were below their corresponding threshold levels of (15 - 20, 60 - 150, 20 - 2500, 100 - 4000, 2 - 200, and 10 - 300 mg kg^{-1}).

Therefore, *Acacia mangium* plants might have absorbed some amounts of the heavy metals from the mined spoil.

4. The concentration of extractable heavy metals at the re-vegetatively restored mined spoil followed a decreasing order of Fe > Zn > Mn > Cu > Pb > As > Cd
5. The maximum accumulation of heavy metals (As, Cu, Fe, Mn, and Zn) in cassava tubers occurred as follows: (As: 0.15, Cu: 0.75, Fe: 13.01, Mn: 1.25, Zn: 9.67 mg kg^{-1}). These concentrations were below the phytotoxic limits of (As: 1.1, Cu: 4 - 15, Fe: 48, Mn: 15 - 100, and Zn: 15 - 200) mg kg^{-1}.
6. Based on the results of the study, both hypothesis are accepted. However, constant consumption of the cassava tubers from the re-vegetated restored sites could lead to bio-accumulation of heavy metals in the body system and this might be lethal. Therefore, constant consumption of the crop must be regulated.

Recommendations

1. Soil amendments could be applied out in order to improve the physico-chemical properties of the soil before any large scale crop production could be done.
2. Studies on heavy metal accumulation in the cassava grown at the re-vegetatively restored mined spoils could be made at different seasons in order to help advise farmers on the consumption pattern of the crop.

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APPENDIX I

Map of Ghana showing Obuasi Township

Source: http://web2.ges.gla.ac.uk/upima/Fieldwork_Ghana.html. 08/06/2015

APPENDIX II

The descriptive ranges of soil pH

Ultra acid	<3.5
Extremely acid	3.5-4.4
Very strong acid	4.5-5.0
Strong acid	5.1-5.5
Moderately acid	5.6-6.0
Slightly acidic	6.1-6.5
Neutral	6.6-7.3
Slightly alkaline	7.4-7.8
Moderately alkaline	7.9-8.4
Strongly alkaline	8.5-9.5
Very strongly alkaline	>9.5

Soil Survey Division Staff (1993).

APPENDIX III

Soil Test Interpretation Guide

A: Macronutrients (N P K)

Element:	Nitrogen		Phosphorus		Potassium
Extraction					Ammonium Acetate
Method:	Kjedahl	2NKCl	Bray	Olsen	Available
Unit	%	ppm	ppm	ppm	ppm
Levels:					
Low	0.05-0.13	<20	<20	<15	<175
Adequate	0.13-0.23	20-41	20-40	15-25	175-280
High	0.23-0.30	41-75	40-100	>25	280-800

B: Macronutrients (Ca, Mg)

Element	Calcium		Magnesium	
Extraction				
Method:	Ammonium Acetate		Ammonium Acetate	
Unit:	meq/100g	ppm	meq/100g	ppm
Levels:				
Low	<5	<1000	<0.5	<60
Adequate	5-10	1000-2000	0.5-1.5	60-180
High	>10	>2000	>1.5	>180

Edward, et al. (2012).

C: Micronutrients

Element	Iron	Manganese	Zinc	Copper
Extraction				
Method:	DTPA	DTPA	DTPA	DTPA
Units:	ppm	ppm	ppm	ppm
Levels:				
Low	<2.5	<0.6	<1.0	<0.6
Adequate	2.5-5.0	0.6-2.0	1.0-1.5	0.6-2.0
High	>5.0	>2.0	>1.5	>2.0

Edward, et al. (2012).

APPENDIX IV

Correlation of physico-chemical properties of soil at the re-vegetatively restored mined soil

	N	P	Ca	Mg/	K	Na	OC	silt	clay	sand	BD	Pore Vol	CEC	Exc Acidity	ECEC	pH	Fe	Zn	Cu	Mn	Pb	As
MC	-.620*	-.541*	-.365	-.520*	-.591*	-.093	-.800**	.143	.098	-.160	-.577*	.604*	-.502	-.502	-.502	-.495	-.226	.283	.353	-.553*	-.193	-.307
N		.764**	.673**	.735**	.783**	.245	.771**	.063	.079	-.100	.688**	-.775**	.730**	.730**	.730**	.611*	.280	-.505	-.720**	.609*	.334	.529*
P			.733**	.988**	.991**	.538*	.811**	-.074	-.035	.050	.685**	-.876**	.974**	.974**	.974**	.508	.090	-.185	-.798**	.520*	.677**	.832**
Ca				.756**	.722**	.710**	.490	-.016	-.004	.009	.329	-.513	.826**	.826**	.826**	.540*	-.033	-.119	-.658**	.219	.665**	.495
Mg					.976**	.607*	.769**	-.129	-.043	.097	.623*	-.832**	.990**	.990**	.990**	.479	.012	-.137	-.794**	.448	.732**	.794**
K						.533*	.818**	-.082	-.066	.077	.664**	-.858**	.966**	.966**	.966**	.490	.094	-.188	-.753**	.497	.683**	.832**
Na							.093	-.361	-.465	.571*	-.211	-.085	.694**	.694**	.694**	.088	-.631*	.423	-.248	-.378	.953**	.477
OC								-.039	.204	-.143	.878**	-.922**	.719**	.719**	.719**	.586*	.481	-.320	-.642**	.816**	.234	.563*
silt									.062	-.657**	.220	-.107	-.143	-.143	-.143	.528*	.497	-.257	-.128	.285	-.354	-.028
clay										-.790**	.419	-.263	-.096	-.096	-.096	.124	.705**	-.459	-.294	.490	-.479	-.254
sand											-.468	.283	.146	.146	.146	-.410	-.846**	.515*	.309	-.558*	.570*	.176
BD												-.951**	.541*	.541*	.541*	.619*	.667**	.598*	-.755**	.974**	-.068	.449
poreVol													-.768**	-.768**	-.768**	-.614*	-.473	.468	.837**	-.861**	-.243	-.652**
CEC														1.000**	1.000**	.476	-.053	-.088	-.760**	.364	.792**	.784**
ExcAcidity															1.000**	.476	-.053	-.088	-.760**	.364	.792**	.784**
ECEC																.476	-.053	-.088	-.760**	.364	.792**	.784**
pH																	.437	-.406	-.564*	.625*	.068	.204
Fe																		-.465	-.272	.764**	-.571*	-.037
Zn																			.527*	-.627*	.355	-.039
Cu																				-.654**	-.334	-.561*
Mn																					-.269	.268
Pb																						.678**

Where; MC – Gravimetric Moisture Content, N – Nitrogen, P – Phosphorous, Ca – Calcium, Mg – Magnesium, K- Potassium, Na- Sodium, Oc-Organic Carbon, Bd-Bulk density, CEC – Cation Exchange Capacity, Fe – iron, Zn- zinc, Cu- Copper, Mn- Manganese, Pb - Lead

APPENDIX V

A: Cadmium content of cassava tubers from the study area

Heavy metal	Treatment	cassava
	C	0.08
Cd	N	0.08
	T	0.08
l.s.d.	*	
s.e.	0.00	

B: Lead content of cassava tubers mature from the study area

Heavy metal	Site	cassava
	C	0.04
Pd	N	0.04
	T	0.04

l.s.d. *

s.e.d.0.000

APPENDIX VI

A: Translocation factor values for Arsenic

H.M	Treatment	Crop	T.F
As	N	Cassava	0.132
	T_1	Cassava	0.417

B: Translocation factor values for Copper

H.M	Treatment	Crop	T.F
Cu	C	Cassava	0.021
	N	Cassava	0.024
	T_1	Cassava	0.019

C: Translocation factor values for Iron

H.M	Treatment	Crop	T.F
Fe	C	Cassava	0.025
	N	Cassava	0.063
	T_1	Cassava	0.027

D: Translocation factor values for manganese

H.M	Treatment	Crop	T.F
Mn	C	Cassava	0.038
	N	Cassava	0.102
	T_1	Cassava	0.022

E: Translocation factor values for Zinc

H.M	Treatment	Crop	T.F
	C	Cassava	0.184
	N	Cassava	0.201
Zn	T_1	Cassava	0.225