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KWAME NKRUMAH UNIVERSITY OF SCIENCE AND TECHNOLOGY KUMASI

COLLEGE OF SCIENCE

DEPARTMENT OF THEORETICAL AND APPLIED BIOLOGY

**THE USE OF INDIGENOUS PLANT SPECIES IN THE RECLAMATION OF
DEGRADED MINED LAND- A CASE STUDY; NEWMONT GHANA GOLD
LIMITED AHAFO**

THIS DISSERTATION IS PRESENTED TO THE DEPARTMENT OF
THEORETICAL AND APPLIED BIOLOGY IN PARTIAL FULFILMENT OF THE
REQUIREMENTS OF MASTER OF SCIENCE DEGREE IN ENVIRONMENTAL
SCIENCE

BY

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OCTOBER, 2012.

DECLARATION

"I declare that I have wholly undertaken this study reported therein under the supervision of Dr. Philip K. Baidoo and that except portions where references have been duly cited, this dissertation is the outcome of my research."

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DEDICATION

I dedicate this work to my family, the Joseph Kwasi Amponsah family and my mentor,

Dr. P. K. Baidoo.

KNUST



ABSTRACT

Reclamation of degraded mined land is mandatory to mining industries with respect to Environmental Protection Agency Act (Act 490), 1994 which was amended in 1999. This involves the re-vegetation of the land back to original form. To obtain the natural succession of the degraded area, it is ideal to use plant species that are indigenous to the area. Although the idea of using indigenous species in vegetation of degraded lands is now widely accepted there is need for continued investigation into the establishment of indigenous species, which perform comparably to exotic species on Mine tailings sites. One of the specific methods for removal of metals from these environmental segments is the use of certain plant species which have the ability to accumulate heavy metals. In recent years, the ability of plants to accumulate pollutants has received significant attention and given rise to a new technology called phytoremediation. The ability of hyperaccumulation of three indigenous plant species *Chromolaena odorata*, *Pennisetum purpureum* and *Centrosema pubens* to hyperaccumulate heavy metals were assessed. Heavy metal content in waste rock and in each plant species was determined using AAS in dry samples of plants which were sampled. The concentration of Fe was the highest in both waste rock chip and the control soil. The concentration of Pb, however was the lowest in both the both waste rock chip and the control soil. The accumulation of all metals except Pb by the three plants was found to be higher as compared with concentration in waste rock. The plant species bioaccumulation factors showed their specific metal affinity and time limitations for their application as phytoremediants. All the three plants *Chromolaena odorata*, *Pennisetum purpureum* and *Centrosema pubens* showed significant accumulation for Copper (Cu), Arsenic (As) Iron (Fe) and Zinc (Zn). *Chromolaena odorata* was good candidate for hyperaccumulation of of Zinc (Zn), Copper (Cu) and Arsenic (As), while *Centrosema pubens* and *Pennisetum purpureum* were good candidates for hyperaccumulation Iron (Fe), Copper (Cu) and Arsenic (As). The adaptability of these three indigenous plants species to heavy metal stress thus provides useful information for their selective exploitation in phytoremediation of contaminated mine sites.

TABLE OF CONTENTS

TITLE PAGE.....	I
DECLARATION-----	ii
DEDICATION -----	iii
ABSTRACT -----	iv
TABLE OF CONTENTS -----	v
LIST OF TABLES -----	ix
LIST OF FIGURES-----	ix
LIST OF PLATE -----	ix
LIST OF ABBREVIATION AND ACRONYMS -----	x
ACKNOWLEDGEMENTS -----	xi
 CHAPTER 1 -----	 1
INTRODUCTION -----	1
1.0 Background-----	1
1.1 Statement of the problem-----	4
1.2 Justification-----	5
1.3 General objective -----	6
1.3.1 The specific objectives were to-----	6
1.4 Hypothesis-----	6
 CHAPTER 2 -----	 7
LITRETURE REVIEW -----	7
2.1 Mining-----	7
2.2 Types of mining techniques-----	8
2.3 Mining in Ghana-----	9
2.4 Heavy metals -----	10
2.5 Origin of waste rocks -----	11
2.6 Impact of mining-----	12
2.6.1 Economic impacts-----	12

2.6.2 Social impacts-----	13
2.6.3 Environmental impacts -----	16
2.6.3.1 Air Contamination -----	17
2.6.3.2. Water Contamination -----	19
2.6.3.3 Soil Contamination -----	23
2.6.3.4 Biodiversity -----	23
2.7 Mitigating Environmental and social impacts of mining.-----	25
2.7.1 Mitigation of social impact -----	26
2.7.1 .1 Mitigation to Air contamination-----	27
2.7.1 .2 Mitigation to impact on water -----	28
2.7.1 .3 Mitigation measures for impact on Land -----	29
2.8 Methods of reclamation-----	29
2.8.1 Practices of mine waste remediation/ reclamation-----	30
2.8.1.1 Challenges of waste rock remediation/ reclamation -----	30
2.8.1.2 Re- vegetation Strategies-----	31
2.8.2 Phytoremediation in reclamation -----	33
2.8.3 Mechanisms Used for Phytoremediation -----	34
2.8.3.1 Phytoextraction -----	34
2.8.3.2 Phytostabilization -----	36
2.8.3.3 Rhizofiltration -----	36
2.8.3.4 Phytovolatilization -----	37
2.8.3.5 Phytodegradation-----	37
2.8.3.6 Rhizodegradation-----	38
2.9 Indigenous plant species for reclamation -----	38
 CHAPTER 3 -----	 41
MATERIALS AND METHODS -----	41
3.1 Study Area-----	41
3.2.1 Climate condition of the area -----	41
3.2.2 Vegetation -----	42
3.2.3 Topography-----	42

3.2.3 Geology and soil-----	42
3.3 The Waste rock dump-----	43
3.4 The reclamation Plot -----	43
3.5 Experimental Design -----	44
3.6 Sample Collection, preparation, storage and analysis-----	44
3.7 Laboratory Analysis-----	45
3.8 Data Collection-----	45
3.8.1 Waste Rock chip and Soil analysis -----	45
3.8.2 Organic Carbon Determination-----	45
3.8.3 Available Phosphorous -----	46
3.8.4 Soil and Waste rock (milled) pH -----	47
3.8.5 Digestion of Soil Samples for Total Metal Content -----	47
3.8.6 About Atomic Absorption Spectrometry -----	48
3.8.7 Analysis of Total Metal Content -----	49
3.9 Plant analysis-----	50
3.9.1 Fresh and Dry Weights -----	50
3.9.2 Moisture Content -----	50
3.9.3 Ashing and Digestion of Plant Materials for Total Metal Analysis-----	50
3.9.4 Bioaccumulation Factor (ratio) (Bf) -----	51
3.10 Data Analysis -----	51
CHAPTER 4 -----	53
RESULT -----	53
4.1 Selected identified indigenous plant species at Apensu waste rock dump -----	53
4.2. Soil and waste rock chip physicochemical properties -----	53
4.2.1 The particle size distribution of waste rock chip-----	54
4.3 Metal concentration in milled waste rock and control soil -----	54
4.4 Metal concentration in plant -----	55
4.5 Metals concentration in waste rock chip compare to concentration plant species --	56
4.7 Bioconcentratrion Factor -----	58

4.7.1 Determination of Bioconcentration factor of <i>C. odorata</i> , <i>P. purpureum</i> and <i>C. pubensis</i> in waste rock chip. -----	58
--	----

CHAPTER FIVE -----	60
---------------------------	----

DISCUSSION -----	60
-------------------------	----

5.1 Soil and rock chip physicochemical parameters -----	60
---	----

5.2 Metal concentrations in waste rock chip and control soil-----	61
---	----

5.3 Metal concentration in plant species -----	63
--	----

5.4 Comparison concentration in waste rock, control soil and the plant species-----	64
---	----

5.5 Comparison of metal concentration in the plants -----	67
---	----

5.6 Bioaccumulation factor (ratio) – Bf-----	67
--	----

CHAPTER SIX -----	71
--------------------------	----

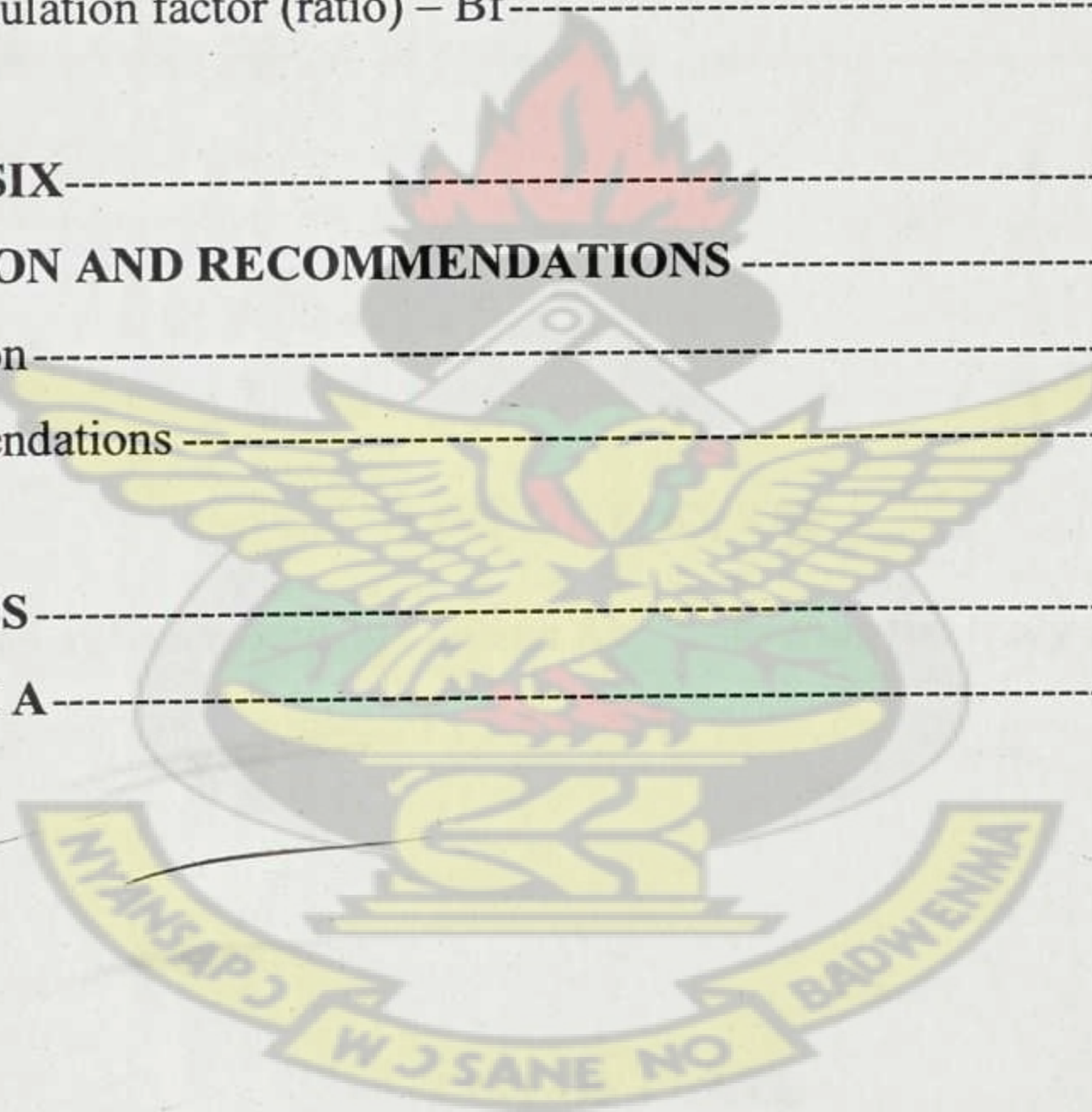
CONCLUSION AND RECOMMENDATIONS -----	71
---	----

6.1 Conclusion-----	71
---------------------	----

6.2 Recommendations -----	73
---------------------------	----

REFERENCES -----	74
-------------------------	----

APPENDEX A -----	88
-------------------------	----



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LIST OF TABLES

Table 1. Physicochemical characteristics of waste rock and control soil -----	53
Table 2. Metals concentration in waste rock chips and control soil (mg/kg) -----	55
Table 3. Metals uptake or concentration in selected plant species (mg/kg) -----	56
Table 4. Metals concentration in waste rock chips and in the plant species -----	57
Table 5. Metals concentration in control soil and in the plant species -----	57
Table 6. Bioconcentration factors of <i>Chromolaena. odorata</i> -----	58
Table 7. Bioconcentration factors of <i>Pennisetum. purpureum</i> -----	59
Table 8. Bioconcentration factors of <i>Centrosema. pubensis</i> -----	59

LIST OF FIGURES

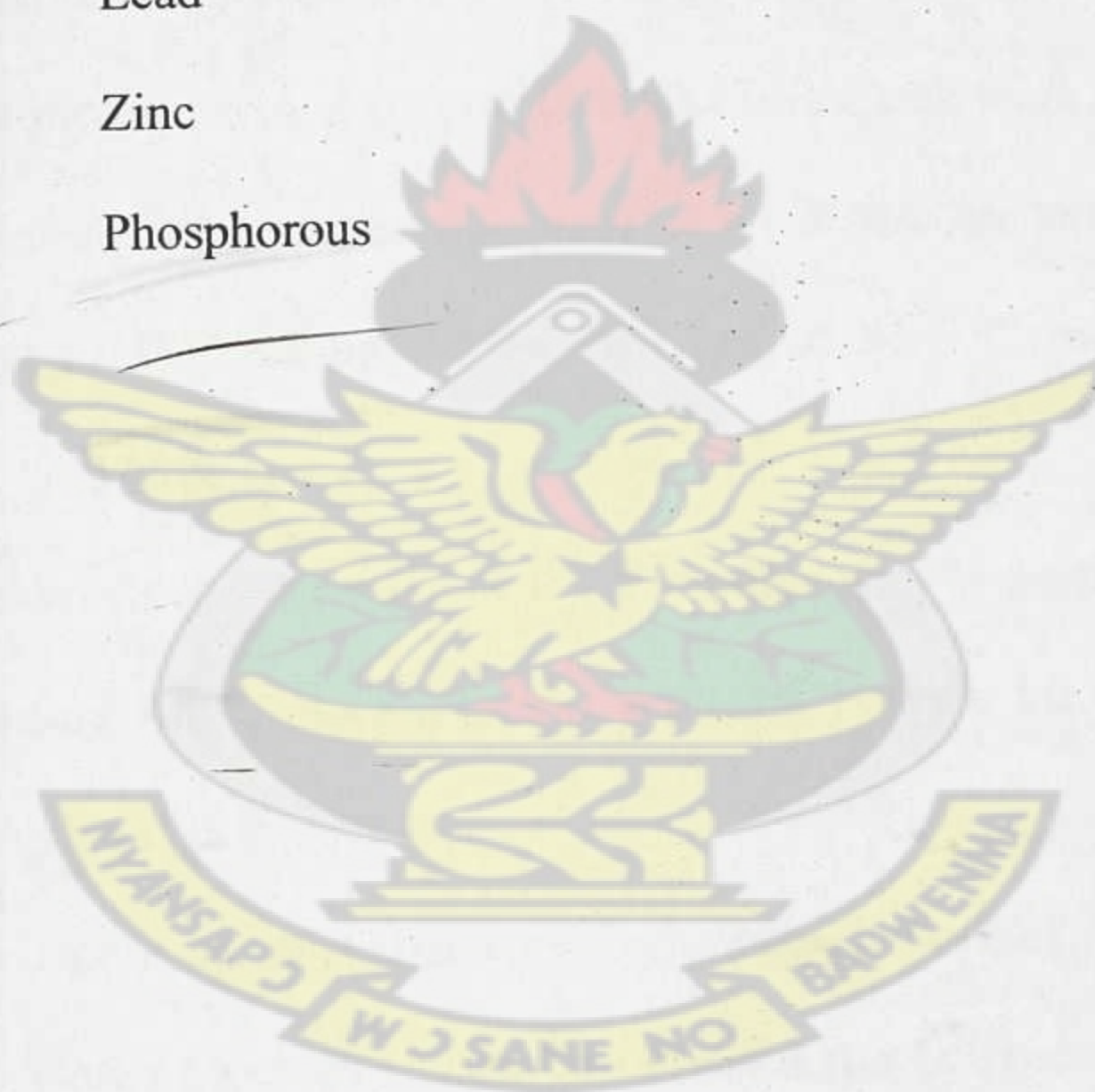
Figure 1. a map of Newmont Gold Ghana Ahafo showing the study site. -----	43
---	----

LIST OF PLATE

Plate 1. Atomic Absorption Spectrometer used to determine heavy metals in soil and plant samples. -----	48
---	----

LIST OF ABBREVIATION AND ACRONYMS

As	Arsenic
Cu	Copper
Fe	Iron
Pb	Lead
Zn	Zinc
P	Phosphorous



ACKNOWLEDGEMENTS

To almighty God be the glory and honour for seeing me through to the successful completion of this dissertation. I would not have been able to get this far without His grace and mercies.

My sincere gratitude goes to Dr. P. K. Biadoo for his supervision, guidance and unflinching encouragement and support through the research work.

I cannot find words to express my gratitude to my family Mr. and Mrs. Joseph Kwasi Amponsah, Owusu Victoria my lovely wife and Audrey Worae DENTAA my dear daughter. May God bless you.

I gratefully acknowledge the staff of the Environmental Department of NEWMONT Ghana Gold Brong Ahafo especially Mr. Williams Todorh, Mr. Benn Abubakari, for their help during my data collection.

To Mr. Samuel Joe Acquah (Laboratory Technician, School of Agricultural Science – KNUST), Dr. Moses Mochiah (Research Officer Crop Science Institute, Entomology Section – Crop Research Agric Kumasi Kwadaso). I say a big thank you for your support and contribution to the work.

It is a pleasure to the appreciate support of my friends, Agyapong Daniel, Teku Emmanue, Amoako Kwasi Gilbert, Oduro Otabil Ankamah Charlse, Maamah Seth, Eric Damoah for being there for me when he need arose.

Finally, I would like to thank all and sundry who in diverse ways helped me to complete this work. God bless you all.

CHAPTER 1

INTRODUCTION

1.0 Background

Mining is a major contributor to the economy of a nation. Its operations, however, results in devastating damage to the environment. Cao, (2007) noted that mining activities or mineral exploration, despite being an economic activity cause substantial damage to the environment.

All types of mining worldwide involve stripping of vegetation, excavating of land and rocks to get to the ore body. These activities generate waste of all kinds including liquid, and solid (mostly rock waste) that are deposited on land that may also cause severe damage to the environment. Surface mining severely alters the landscape and causes damage to the natural environment and the surrounding land (Hamilton, 2005). As the state of the environment is altered as a result these mining operations, mining companies' management make a plan to reclaim the degraded lands for sustainable development., Modern mining processes involve prospecting for ore bodies, analysis of profit potential of a proposed mine, extracting the desire materials and finally reclamation of the land to prepare it for other uses once the mine is closed (Tandoh, 2011).

According to US Department of the Interior, (1997), surface mining of minerals completely eliminate existing vegetation, destroys the genetic soil profile, displaces or destroys wildlife and habitat, degrades air quality, alters current land uses and to some extent permanently changes, the general topography of the area mined. The community

of micro-organisms and nutrient recycle processes are upset by movement, storage and distribution of soil.

Primarily, mining, particularly surface mining involves the clearing of the existing vegetation, stripping of the top and subsoil (a habitat for most micro-organism) and even deeper. Consequently, diversity of living organisms ranging from plants, various animals and micro-organisms are lost. The biological community structure of the area could be disturbed by eliminating species in the food chain. The mined land could be losing its original landscape beauty; moreover the ability to support meaningful life will also be diminished. Mine waste rock and tailings dumps destroy wildlife habitats and leach out toxic chemical into streams and groundwater if not properly rehabilitated. In many instances, these dumps have been abandoned, with limited or no rehabilitation treatment (Gardner, 2001).

Generally, soil disturbance and associated compaction result in erosion. The soil removed from surface mined areas alters or destroys many natural soil characteristics and may reduce productivity of agriculture or biodiversity, even to the extent of the soil structure being disturbed by pulverization or aggregate breakdown. Additionally, the contamination resulting from leakage of chemicals can also affect the health of the local population if not properly controlled (Larmer, 2009).

Gold mining activities may generate environmental impacts in surrounding areas and even far from its origins. The tails and refuse piles of mining will contain minerals such as Arsenopyrite, the most common source of contamination from mining industry (Martey, 2010).

Mine waste rock are large piles of crushed rocks that are left over after the metals of interest like Lead, Zinc, Copper, Silver, Gold and others, have been extracted from the rocks that contained them (en.wikipedia.org/wiki/Mining). When the mine tailing and waste rock piles are exposed to atmospheric oxygen and water, potential toxic elements are released into the environment through acid mine drainage and other drains related to the mining activities that can affect the land and aquatic ecosystem. The common elements associated with mine tailings include Fe, S, Mn, Cu and As, with As being the one very highly toxic even at low concentrations. Improved techniques for reclamation of land disturbed by mining activities have become increasingly important as pressures for extracting minerals from the earth increase (Julie and Darsell, 2007).

The environment constitutes and contains the natural resources needed for the survival of mankind. However, the anthropogenic activities undertaken for the exploitation of these resources detriment the life that exist and depend on the environment. Such activities including mining must be monitored and controlled in accordance with Environmental Protection Agency Act 490, 1994. As stated by Appiah-Opoku, (2001), EIA was introduced in Ghana in 1995 and is mandatory for projects such as mining.

In the study of Cao, (2007), the uprising of public awareness and concern for the environment regulate the legislative requirement for mine site to be properly rehabilitated or reclaimed after the life of mining activity elapses. Effective reclamation is necessary and if properly done; it is able to restore the disturbed areas to its former state, thus supporting economic activities.

Ultimately, reclaiming mined site involves re-vegetation and phytoremediation techniques, both of which require plant species for such purposes. However, peculiar plant species have characteristics that are adapted to harsh soil and climatic conditions of mined site. Either exotic or indigenous species may be capable, but ideally the indigenous species are at an advantage.

Although the idea of using indigenous species in vegetation of slime dams is now widely accepted, there is the need for continuous investigation into the establishment of indigenous species, which perform comparably to exotic species on Mine waste rock sites (Mendez *et al.*, 2007).

1.1 Statement of the problem

Mining companies offer a number of short term benefits to rural communities and the nation as a whole. However, mining activities are associated with long term adverse impacts to the society and the environment. Without doubt most of these mining sites are located in rural areas, where the land is used for farming, (the paramount source of livelihood of those living in the area). Mining may in effect lead to the change of land use of the rural community.

Mine waste rock are sources of acid rock drainage and thus, both underground workings and open pit walls may generate acid rock drainage (ARD), which can contaminate ground and surface waters for decades or even hundreds of years after mine closure. This occurs where “significant” amounts of sulfide minerals are present in the ores and waste rock, and causes a lowering of water pH to about 1.0 to 4.0, which then mobilizes many other chemicals, such as toxic metals and non-metals (Borreguard *et al.*, 2000).

The acid rock drainage (ARD) and acid mine drainage from mine tailings which contain heavy metals such as Cu, Mn, Zn, Cd, Ni, Fe, As and Ag (Mulugisi, *et al.*, 2009), may drain onto farm lands to affect soil characteristics which will hinder agricultural productivity that affect the livelihood of the people.

Moreover, leachate from waste rocks may drain into surrounding water bodies causing contamination. Dispersion of waste rock particles in wind storm to surrounding communities may result in health problems to the people of the community. Consequently, the company may attract bonds and fines against unsatisfactory rehabilitation. According to Rodrigues *et al.*, (2004), the natural forest recovery on abandoned goldmine site is often problematic due to disturbed soil and hydrology. Thus, the landscape, soil and water are greatly disturbed which leaves a lot of problems to be dealt with. These scenarios are the evidences that post-mining reclamation is necessary, since a proper reclamation may ensure regaining land for short and long future use.

1.2 Justification

If the mining industry is to contribute effectively to sustainable development, it must develop and consistently apply sound environmental management practices to minimize on and off site environmental impacts. There are cases in which waste rock dumps have been effectively rehabilitated for agriculture, forestry, nature conservation, urban or industrial land uses (Moffat, 2001; Mulizane, 2005). This can be successful in cost effective manner by employing indigenous species due to their tolerance and adaptive characteristics to climatic conditions of mining area.

According to Julie and Darsell (2007), indigenous plant species are appropriate for reclaiming degraded mined land. Once the area is re-vegetated it requires no maintenance care as the species are well adapted to the climate and environmental conditions of the region. Actually, much work has to be done on indigenous plant species for mining reclamation in Ghana.

1.3 General objective

- The main objective for the study was to assess effectiveness of the indigenous plant species that could be used to reclaim degraded mined land at NEWMONT Ghana Gold Ahafo mines.

1.3.1 The specific objectives were to

- identify indigenous plant species that have been growing on the mining waste rock dump.
- determine the some metal composition of the rock waste.
- determine the physicochemical parameters of waste rock chips and soil of reclaimed site.
- determine the concentration of some metals in the plant species.

1.4 Hypothesis

H_0 ; there are indigenous plant species for reclamation at NEWMONT Ahafo.

H_1 there are no indigenous plant species for reclamation at NEWMONT Ahafo.

CHAPTER 2

LITRETURE REVIEW

2.1 Mining

Mining is the extraction of valuable minerals or other geological materials from the earth, from an ore body. The term also includes the removal of soil. Materials recovered by mining include base metals, precious metals, iron, uranium, coal, diamonds, limestone, oil shale, rock salt and potash (en.wikipedia.org/wiki/Mining). Any material that cannot be grown through agriculture, or created artificially in a laboratory or factory, is usually mined. Mining in a wider sense comprises extraction of any non-renewable resources such as petroleum, natural gas, or even water (en.wikipedia.org/wiki/Mining). Before any work is done in the mining of a mineral, exploration drilling is done to define a viable orebody to mine, after which geologists and engineers will work together to determine the most efficient method of mining (MIGSDAF, 2002).

The initial step of the mining and mineral processing is the actual removal of the mineral value in the ore from the host rock or matrix. Minerals may be extracted from the earth using a variety of techniques. Most extraction processes result in the removal of ore and associated rock or matrix in bulk from the deposits using blasting and various mechanical means to break the ore into pieces of manageable size or to separate the ore minerals from unwanted materials (USEPA, 2000).

2.2 Types of mining techniques

As reviewed by the United State Environmental Protection Agency (USEPA, 2000), mining can be categorized as surface mining, underground, and in-situ mining. MIGSDAF (2002) also indicated that there are two basic methods of mining which are opencut and underground that can remove any ore from the ground. Surface mining is used to excavate ores at or close to the earth's surface (USEPA, 2000). Methods under surface mining include; open-pit or opencut mining which is used when the ore body is near the surface and little overburden (waste rock) needs to be removed (MIGSDAF, 2002). Strip mining and Mountaintop removal (MTR) are other methods of surface mining. Opencut or open-pit and strip mining such as contour striping all may use some of the very large mobile machinery including bucket-wheel excavators which can move as much as 12,000 cubic meters of earth per hour to achieve high production rates. (en.wikipedia.org/wiki/Mining; MIGSDAF, 2002). The Mountaintop removal (MTR) method of surface mining uses explosives to blast "overburden" off the top of some Appalachian mountains. Excess mining waste or "overburden" is dumped by large trucks into fills in nearby holler or valley fills. MTR involves the mass restructuring of earth in order to reach the mineral seam as deep as 400 feet (120 m) below the surface, (en.wikipedia.org/wiki/Mining). A type of surface mining known as terrace mining is used where the overburden is too thick or the floor of the pit that is the ore inclination is too steeply dipping to allow waste dumping directly over the pit, it is necessary to use intermediate cyclic or continuous transport to transport the overburden to where it can be tipped back into the previously mined void (Bullivant, 1987).

Underground and in-situ mining are other mining methods where both remove mineral from deeper deposit, the underground mining is done by excavating under the surface and removing the ore and in-situ mining is done by sinking injection and extraction wells and leaching the ore in place (USEPA, 2000).

Underground mining is further classified into underground hard rock mining and soft rock mining. The hard rock mining refers to as the kind of underground mining techniques used to excavate hard minerals, mainly those containing metals such as ore containing gold, silver, iron, copper, zinc, nickel and lead, but also involves using the same techniques for excavating ores of gems such as diamonds. In contrast the soft rock mining refers to excavation of softer minerals such as salt, coal, or tar sands. Other form of underground mining is shaft mining or shaft sinking. This refers to the method of excavating a vertical or near-vertical tunnel from the top down, where there is initially no access to the bottom. When the top of the excavation is the ground surface, it is referred to as a shaft or portal; when the top of the excavation is underground, it is called a winze (en.wikipedia.org/wiki/Mining).

2.3 Mining in Ghana

The mining sector of Ghana is one of the largest contributors to government revenues through the payment of mineral royalties, employee income taxes, and corporate taxes. Mineral commodities produced in the country include gold, aluminum, bauxite, diamond and manganese (en.wikipedia.org/wiki/Mining-industry-of-Ghana).

Akosa *et al.* (2002), identified two basic types of mining found in Ghana; small scale mining and large scale mining. Processes or techniques of ore extraction include handpicking amalgamation, cyanidation, floatation electrowinning and roasting of the ore.

The techniques used in mining operations differ between large and small scale mining and varies depending on the type of deposit and its location (Ntibery *et al.*, 2003).

In Ghana and similar to what is done around the globe two forms of mining are employed open-pit and underground (Annan, 2010). There are 19 large mining companies in Ghana, 16 are operating gold mines, 1 bauxite, 1 diamond and 1 manganese mine. Currently, with the exception of Anglo Gold Ashanti (AGA) Obuasi mine and the Prestea Gold Resources Limited, all the other mines are surface operations (Akabzaa and Darimani, 2001; AGA, 2007).

2.4 Heavy metals

Heavy metals are significant inorganic pollutants, which are, as a rule, a part of many industrial wastewaters and whose negative impact is reflected primarily in numerous changes at different levels of biological system organizations. Heavy metals have different chemical and physical intrinsic properties and physiological effects. Their common property is that even in relatively low concentrations their effects are toxic and therefore they fall under very dangerous environmental pollutants (Kastori and Maksimovic 2006). Increased concentrations of heavy metals in water and sediments have harmful effects to plants and animals due to their toxic properties, and they also affect humans through food chain (Babovic, *et al.*, 2010). For example, metal like Lead and mercury have been listed as a potential carcinogen in the EPA Toxic Release

Inventory (TRI). When one is exposed by either inhalation or ingestion the effects from both are the same. Pb accumulates in the body organs like the brain, which may lead to poisoning (plumbism) or even death. The gastrointestinal tract, kidneys, and central nervous system are also affected by the presence of lead (AGC, 2001). Children exposed to lead are at risk for impaired development, lower IQ, shortened attention span, hyperactivity, and mental deterioration, with children under the age of six being at a more substantial risk. Adults usually experience decreased reaction time, loss of memory, nausea, insomnia, anorexia, and weakness of the joints when exposed to lead (Jeanna, 2000).

According to Jeanna, (2000) the **most common** heavy metals at hazardous waste sites are Cadmium (Cd), Chromium (Cr), Copper (Cu), Lead (Pb), Mercury (Hg), Nickel (Ni) and Zinc (Zn). Out of these, lead and mercury are two of the most significant contaminants, posing serious and sometimes life threatening health hazards. Lead, which contaminates more than 50% of sites found on the NPL, is one of the most prominent metal contaminants found in hazardous waste sites. Mercury also poses significant environmental and health concerns. The World Health Organization (WHO) has approximated that each year 10,000 tons of mercury is released globally from both natural and anthropogenic sources (AGC, 2001).

2.5 Origin of waste rocks

Waste rocks are large piles of crushed rocks that are left over after the metals of interest have been extracted from the rocks that contained them (en.wikipedia.org/wiki/Mining).

AGC, (2001), documented that Birimian rock system which contains mined mineral is

the source of certain heavy metal. Thus, reviewing waste rock may be rich in heavy metals such as Cu, Mn, Zn, Cd, Ni, Fe, As and Hg.

2.6 Impact of mining

The impact of mining can be categorized as socio-economic, and environmental. Borregaard *et al.*, (2000) noted that mining and metallurgic processes in all their phases have an impact on both the natural environment and the socio-economic and cultural ones. The nature of mining processes create a potential negative impacts on the environment both during mining operations and for years after the mine is closed. This impact has led to most nations adopting regulations to moderate the negative effects of mining operations (en.wikipedia.org/wiki/Mining).

Waste rock dumps destroy wildlife habitats and may leach out toxic minerals into streams and groundwater if not properly rehabilitated. In many instances, these dumps have been abandoned, with limited or no rehabilitation treatment (Gardner, 2001). (UNEP), 2002 has documented that several organizations had documented the resulting impact of the release of acid mine waters containing elevated metal and Cyanide concentration onto landscapes and waterways.

2.6.1 Economic impacts

Minerals and Mining Law, for instance, in Ghana scaled down the corporate income tax liability and provides more specific fiscal allowances that aim to reduce the general tax liability of mining sector operators. For example, corporate income tax stood at 50-55% in 1975 but was reduced to 45% in 1986 and further scaled down to about 35% in 1994.

The mining sector has been contributing significantly to the government revenue until the reform of mining laws got scrapped (Akabzaa and Darimani, 2001).

Mining companies are also a significant source of revenue for local and state governments. Mining industries through tax payments of all kinds and royalty payment to local government strengthen the economy of a community and the nation at large (Hughes and Diaz 2008).

A percentage of employment is given directly by the mining sector in a region. However, indirect employment generated by mining companies in a region is important. The importance of the mining sector as a regional employment source is reinforced when it is taken into consideration the policy of employing as many local people as possible (Borregaard *et al.*, 2000). Mining companies strengthen communities by employing residents from communities and purchasing supplies and services from hundreds of companies (Hughes and Diaz, 2008). Abandoned mines may result in large clean-up costs and closure liabilities for governments (World Bank, 2002).

2.6.2 Social impacts

Gold mining, particularly surface mining activities take place on indigenous lands that provide land-based livelihoods to many rural people. Mining takes place in areas of high illiteracy and poverty, thus, mining communities have weak capacity to face the challenge of negotiating on high technical issues with mining companies that can afford to employ the services of competent experts. Although, the mining law makes for fair,

adequate and prompt payment of compensations, low compensations are made which exacerbate the problem of poverty in mining communities (Owusu-Koranteng, 2005).

Borregaard *et al.* (2000), reviewed socio-cultural impacts of mining as follows; the mining sector activities may have important social and cultural impacts, especially in situations where the mining sector becomes the predominant productive activity in a region or local community. The influx of workers and their families into areas previously sparsely populated, or the predominance of mining activities over several hundreds of years obviously have significant impacts in socio-cultural terms. The main potential socio-cultural impacts beyond the company-town phenomenon include: a rise in alcoholism, an increase in the use of drugs, a rise in prostitution, homo-sexualism, and related diseases. The income differences between people also increases, improved educational or health services division in the community between those that “belong” directly or indirectly to the mining sector and those that do not.

Some of these different social impacts have been analyzed in Borregaard *et al.* (2000), for example “the contribution of the mining sector to local health services was analyzed to be substantial to the life of people of the community. These contributions often motivate good community and company relationship. However, problems can arise as the expectations of community are not fulfilled. For instance a profound frustration may result when the community noticed that several of the promises of the company were not complied with, basically the promises including local contracting services. Thus, very often these experiences that occur in the community disseminate situation of distrust amongst the different local or regional people.

According to Singh, (2008), despite the very substantial benefits mining bestow on society, there are great social challenge for the mining industry which is sustainable development and company acceptance of its role in society. The problem of mining-induced displacement and resettlement (MIDR) poses major risks to societal sustainability. Some of these social impacts are: (i) Landlessness: MIDR raises the significant risk of landlessness by removing the foundations upon which productive systems, commercial activities, and livelihoods are articulated. (ii) Joblessness: The local people living in the designated areas depend generally on the land for their livelihood. Since, in mining areas the land is taken for mining and associated activities these people lose their source of livelihood. Post-displacement unemployment or underemployment is often chronic following the dismantling of the local income-generating resource base. (iii) Homelessness: Defined as the "loss of house-plots, dwellings and shelter." For many people, homelessness only may be temporary, but indigent executed displacements, remains chronic.

(iv) Risk of Marginalization: The risk of marginalization threatens displaced individuals and entire communities as they slip into lower socio-economic situation relating to their local areas.

(v) Changes in population dynamics: All the manpower required for mining and associated activities comes from outside the country of operation as such trained manpower is usually not available in the local population. Thus, the population dynamics of the area undergoes a major change over the years resulting in dilution of the local population.

(vi) Cost of living: Increased industrial and economic activities generate more money and increase the purchasing power of the people directly and indirectly associated with these activities. This leads to an increase in the cost of living, which adversely affects the other people, including the indigents, who are not associated with these activities.

(vii) Health Risks: The already marginal health status of the place is worsened by the situation and manner of moving. Recurring problems are reported with resettled populations gaining access to safe potable water and safe sanitation; increased diarrhoea, dysentery and epidemic infections often result (Singh, 2008).

(viii) Disruption of Formal Educational Activities: Risk occurs in the disruption of education and routine socialization. Displacement and relocation often cause a significant interruption in the functioning of schools and in child access to education during the year of transfer or for longer periods of time. (ix) Addictions: Increased economic activities and affluence brings in more addictions to drugs in the society (Singh, 2008).

Social risks arising from badly conducted mine closure can cause severe distress in mining communities because of the threat of economic and social collapse (World Bank, 2002).

2.6.3 Environmental impacts

Mining activities cause significant impacts to all aspect of the environment, this has been documented in Borregaard *et al.*, (2000) that historically, the environmental problems of greatest repercussion is derived from mining activity and this include: Air

contamination by emission of gases and breathable dust particles imbued with heavy metals. Soil contamination by particle matter, dissolved metals and the presence of salts in liquid effluents; marine, underground and surface water contamination by emission of liquid effluents containing particulate matter, dissolved metals, acids and salts; contamination and destruction of renewable natural resources that absorb air, water and soil contaminants; Effects on landscape by open pit mining activity; loss of biodiversity.

There are environmental impact to surface land use and hydrology including streams, lakes, springs, and wetlands. Beyond the effects of subsidence to the land use and water, it may mar landscapes and diminish the aesthetic value of the natural landforms (Gill, 1971; Blodgett, *et al.*, 2002).

2.6.3.1 Air Contamination

The various mining and related construction activities mobilize tremendous quantities of dust particles. These particles may produce negative impacts due to the physical nature of the particles. Such impacts include: Reduction in visibility; aesthetic impacts, such as coating houses, cars, laundry with dust, health impacts, such as respiratory diseases and allergies, due to airborne contaminants; damage to vegetation—gardens, commercial crops, vineyards; and health problems that might result from consumption of contaminated foods grown on such contaminated soils (<http://en.wikipedia.org/wiki/Mining/impact/air/pollution>).

Some of these impacts may be more truly chemical in nature due to the chemical components in the dusts. Mineral processing, most especially, the smelting operations release massive quantities of potentially toxic airborne particles and gases. These

constituents include the various sulphur, carbon, and nitrogen commonly detected in air monitoring. In addition, they may include toxic concentrations of numerous metals such as arsenic, nickel, lead, cobalt and mercury. These components may result in the following impacts, especially if the operations are located near cities or towns: Silicosis, for example, has contributed to the death of thousands of miners throughout the world (<http://en.wikipedia.org/wiki/Mining/impact/air/pollution>).

Singh, (2008), documented that air pollution in mineral mines is mainly due to the emissions of particulate matter and gases including methane, sulphur dioxide, oxides of nitrogen and carbon monoxide. Major operations producing this dust include drilling, blasting, hauling, loading, transporting and crushing. Basically, dust sources in mines can be categorized as primary and secondary sources that generate the dust, which disperse the dust and carry it from place to place (Singh, 2008).

Ogola, *et al.* (2002), opencast mining is more severe with an air pollution problem as compared to underground mining. In underground mining men suffer from dust inside underground mine workings since there is little air to breathe, however, opencast mining creates much more air quality deterioration in respect of dust and gaseous pollutants. It creates air pollution problems not only within the mining premises but also in surrounding residential areas affecting abundant air quality. High levels of suspended particulate matter increase respiratory diseases such as chronic bronchitis and asthma, while gaseous emissions contribute towards global warming besides causing health hazards to the exposed population. The vehicular traffic on haul roads has been

identified as the most important cause of dust emissions and can contribute as much as 85% of the dust emitted from an opencast mine (Singh, 2008).

2.6.3.2. Water Contamination

Mineral processing produces numerous wastes and products that can cause water contamination; these include tailings, waste rock, laboratory wastes, chemical reagents and contaminated containers (solid wastes), blasting compounds, smelter slag / dusts, spent leached ores, ore stockpiles. In addition, the associated infrastructure that must be developed to support a large mining and processing operation generates sewage wastes, water treatment sludge, oils, petroleum, diesel fuels, etc. All can cause contamination of surface and ground waters (Borregaard *et al.*, 2000).

Many mines have an active programme to reduce the water table or divert major watercourses away from the mines. This exercise has disruptive outcomes for the quality and availability of surface and ground water (Yeboah, 2008).

The various exploration, test, and construction activities may increase sediment loads into water bodies (rivers, lakes, oceans), which can harm water quality and aquatic organisms. The water extracted and diverted from operations are impounded or re-injected into the subsurface. These operations often result in undesirable chemical interactions between the dewatering water and the rock or sediment it gets in contact with, which can release elements such as arsenic, contaminating surface and ground waters (Akabzaa and Darimani, 2001).

Mining (both underground and open pit) breaks and crushes rock, creating new pathways for oxygen, air and microbes to react with the rock. Thus, both underground

workings and open pit walls may generate acid rock drainage (ARD), which can contaminate ground and surface waters for decades or even hundreds of years after mine closure. This occurs where “significant” amounts of sulphide minerals are present in the ores and waste rock, which causes a lowering of water pH to between 1.0- 4.0. The lowering of the pH may then mobilize many other chemicals, such as toxic metals and non-metals. ARD draining from underground mine workings have negatively impacted thousands of miles of streams in the USA alone (Borregaard *et al.*, 2000).

Open pit mines that produce ARD may result in an acid, toxic lake forming within the excavated area after the dewatering wells have been shut off. Such conditions may also contaminate ground waters around the pit area. This process may be aggravated because open pits are often connected to older underground tunnels that provide preferred pathways for migration of the contaminated waters. Simple mining processes (without ARD generation) also contaminate waters by increasing the amounts of suspended sediments (and other soluble constituents) released, and by increasing concentrations of nitrates and ammonia due to the blasting compounds used. All these processes can result in eutrophication and contamination of water bodies. Indigenous artisanal mining techniques and placer mining often greatly disturb surface gravels, causing tremendous increases in suspended sediment loads and mercury contamination (Borregaard *et al.*, 2000). Waste rock, which may account for more than 80 percent of the rock mined at many mine sites, often contains high concentrations of sulphide and toxic metals and non-metals. This waste rock is normally disposed of in piles on the surface of the ground at the edges of pits. Many contaminants can be leached out of these waste piles, contaminating surface and ground waters. Undesirable chemical releases from waste

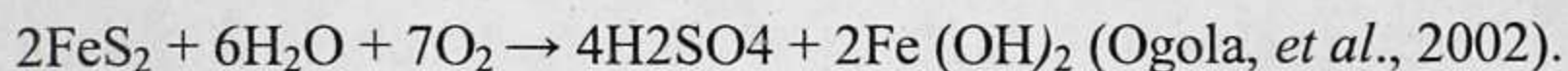
rock can occur even where acid is not added to the piles or where ARD fails to develop. Because of the increased surface area of the broken rock, chemical reaction rates are increased. Many contaminants may be mobile under high pH conditions (above approximately pH 8.5) for example sulphate, arsenic, manganese, iron, mercury, lead, nickel, selenium, molybdenum, vanadium, uranium etc. These constituents are often common in copper ores and in desert environments where alkaline waters predominate. Mineral processing activities sometimes involve grinding the ore, adding various chemicals, and possibly several physical separation processes. These processes result in wastes called tailings, which contain numerous metal and non-metal residues from the ore, but also contain high concentrations of the processed chemicals. These compounds may include kerosene and other petroleum-based or organic compounds, organic acids, cyanide and related compounds, various acids, lime. At older operation sites, tailings may be disposed of directly into stream channels or into the ocean. Where uncontrolled, these tailings wastes obviously can cause significant contamination of all water bodies. The water and soil contaminated with waste held in the tailing dams can affect people's health in the long term, since the absorption and metabolization of these toxic substances produce: teratogenic effects, which cause embryonic deformation; carcinogenics, which cause cancer; and mutagenics, which provoke chromosome alterations, in turn altering the genetic load of the cell (Sierra, 1998 quoted by Borregaard *et al.*, (2000)). On the other hand, the water-soluble toxic metals that are transferred through the soil to water courses by leaching, acidify and contaminate the underground water, and provoke filtering of the waste and sediments into the continental and marine water bodies. Acid water is produced by the contact of water with the metals

contained in the tailing dams in the presence of oxygen and carbon dioxide, aided by microorganisms that act as oxidation catalyst. The greater the amount of water and oxygen in contact with the mineral, the greater the amount of acid water produced. However, in the north of the country, where precipitation is low and geological faults facilitate the underground water percolation, there is no great risk of acidity. The metal sediments can induce changes in the composition of marine water, drastically affecting aquatic life (Sierra, 1998). It is estimated that 67% of fine sediments can cause changes in marine fauna by reducing light penetration with a consequent reduction of photosynthesis in plants that many invertebrates feed on, leading to the death of these and their vertebrate consumers. The sediments can interfere with the particle filter system of some organisms that fulfill a food and respiratory function, restricting their surface area and thus diminishing the aquatic species' disease resistance (Sierra, 1998).

River waters which receive catchments from metal mining areas exhibit high dissolved cadmium, lead and zinc concentrations, primarily due to inputs from the former metal mining areas, although industrial activity, sewage treatment works and atmospheric fallout also contribute (Robson and Neal, 1997; Rees *et al.*, 1998). An Arsenic value in excess of 0.05mg/l in drinking water is considered a hazard to human health. Both the World Health Organization (WHO) and the United State Environmental Protection Agencies (EPA) has adopted this value as standard for arsenic (Straskraba and Moran, 2006).

Waste rock dumps, overburden and run-off mill stockpiles are sources of acid drainage. It is a consequent of the oxidation of sulphide minerals, mainly pyrite (FeS_2), pyrrhotite Ferrous sulphide (FeS), galena (PbS), sphalerite (ZnS), arsenopyrite (FeAsS), and

chalcopyrite (CuFeS_2). FeS_2 , for example, when exposed to oxygen and water oxidizes to sulphuric acid and ferrous hydroxide as follows:



It is the sulphuric acid of the above reaction that gives the strong acidic property, whereas ferrous hydroxide is responsible for the jelly-like yellowish orange colouration to stream water. The sulphuric acid attacks other sulphide minerals and thus breaks them down to release metals such as Pb, As, Cd, Cu, Zn, Ni. The stronger the acid solution, the more the metals become soluble in water and this lowers the pH (Ogola, et al., 2002).

2.6.3.3 Soil Contamination

Mining activities result in significant changes in land use, land stability (particularly with underground exploitation), and land quality, due to the disposal of solid or liquid residues (Borregaard et al., 2000).

2.6.3.4 Biodiversity

Mining activities, particularly site exploitation, can considerably alter the ecosystems present (Blanco, et al., 1998). Studies on the impact that any activity could have on a zone's flora and fauna tend to be more complex, since the effects can be direct or indirect, habitat deterioration, and repercussions on reproductive cycles. Furthermore the impact can occur far from the exploitation area; for example, in the marine ecosystems located at the mouth of a river that carries waste from a mine (Borregaard et al. 2000).

According to Akabzaa and Darimani (2001), extensive areas of land and vegetation in Tarkwa have been cleared to make way for surface mining activities. Currently, open pit mining concessions have taken over 70% of the total land area of Tarkwa. It is estimated that at the close of a mine company would have utilized 40-60% of its total concession space for activities such as siting of mines, heap leach facilities, tailings dump and open pits mine, camps, roads, and resettlement for displaced communities (Akabzaa and Darimani, 2001). Both the prospecting and exploration stages of the mining activities could have possible effects on biodiversity and the landscape (CONAMA 1994; Borregaard *et al.*, 2000).

Actually impact of mining on biodiversity is not concern only about losing a few plants and animals; mankind stands to lose much more. The destruction of the tropical forests, risk the quality of life of mankind. Disturbing the stability of climate and local weather, the existence of other species are threatened, and that undermines the valuable services provided by biological diversity (Groninger, 2007). While in most areas environmental degradation has yet to reach a crisis level where entire systems are collapsing, it is important to examine some of the effects of existing environmental impoverishment and to forecast some of the potential repercussions of forest loss. continuous loss of natural systems could make human activities increasingly vulnerable The most immediate impact of deforestation occurs at the local level with the loss of ecological services provided by tropical rainforests and related ecosystems (Groninger, 2007). Such habitats afford humans valuable services such as erosion prevention, flood control, water treatment, fisheries protection, and pollination—functions that are particularly important to the world's poorest people, who rely on natural resources for their everyday survival.

Forest loss also reduces the availability of renewable resources like timber, medicinal plants, nuts and fruit (Groninger, 2007).

Gold, copper, diamonds, and other precious metals and gemstones are important resources that are found in rainforests around the world. Extracting these natural resources is frequently a destructive activity that damages the rainforest ecosystem and causes problems for people living nearby and downstream from mining operations (Groninger, 2007).

2.7 Mitigating Environmental and social impacts of mining.

The nature of mining processes create potential negative impacts on the environment both during mining operations and for years after the mine is closed. This impact has led to many nations adopting regulations to moderate the negative effects of mining operations (en.wikipedia.org/wiki/Mining).

The most effective ways to mitigate the problem of gold mining contamination in Amazon was perhaps to use remedial procedure, implement education and training to the miners the local population and also to use some clean-up procedures in polluted areas (Daiz, 2000).

In modern operations, these tailings wastes are generally sent to engineered impoundments which are often lined with synthetic liners. If the mining industry is to contribute effectively to sustainable development, it must develop and consistently apply sound environmental management practices to minimize on and off site environmental impacts. There are cases in which tailings dumps have been effectively

rehabilitated for agriculture, forestry, nature conservation, urban or industrial land uses (Moffat, 2001).

Recently, this balance has begun to shift as ideas about corporate social responsibility and sustainable development have gathered strength within the mining industry. At the global and national levels, leading mining companies have now formally embraced the paradigms of sustainable development, corporate social responsibility and 'triple bottom line' reporting. Each of these paradigms defines the community obligations of companies quite broadly and stresses the need to improve social, as well as environmental performance. For example, the International Council of Mines and Metals' (ICMM) sustainable development framework includes an undertaking by signatories to 'contribute to the social, economic and institutional development of the communities in which we operate' (ICMM, 2003). In a similar vein, the World Coal Institute, in its submission to the 2002 'Rio Plus 10 Earth Summit' in Johannesburg, stated that a key 'action area' for the industry is to make a more effective contribution to the social and economic development of local communities' (World Coal Institute, 2001).

2.7.1 Mitigation of social impact

Often mining-induced removal and resettlement of populations threaten the rights to livelihoods. Therefore, projects are designed so that resettlement is minimized or avoided wherever possible. Treating resettlement issues as an integral component of the overall mining project and dedicating proper planning, implementation and financial

resources is necessary for the success of a project's ability to restore or even improve social and economic well being to the displaced population (Singh, 2008).

Mining companies must accept the right of land-owners to negotiate access to their land, to determine whether or not exploration or mining takes place there. If access is given, they must be able to negotiate conditions, such as the preservation of sacred sites, access to traditional hunting grounds, proper resettlement and rehabilitation of those who have to be moved and the determination of compensation packages. In conjunction with community representatives, companies should develop protocols for conducting negotiations over land. These should include procedures for identifying land-owners, conducting meetings, providing technical advice, handling differences between land-owners and determining when negotiations should cease (Singh, 2008).

Account should be taken of alternative plans proposed by the affected people, who must be allowed to identify suitable resettlement sites. Where those being displaced have agriculture as their primary source of income and livelihood, every effort must be made to replace land with the proper land reclamation exercise. If suitable land is not available, other strategies built around opportunities for employment or self-employment should be used. Relocated people must receive legal land titles for their resettlement plots, whether these are house plots or agricultural land (Singh, 2008).

2.7.1 .1 Mitigation to Air contamination

Singh, (2008) suggested that the following measures can be adopted to minimize the generation of dust in the mining operation and its associated activities. These measures include using dust extractors with drills, optimizing blast design for obtaining proper

fragmentation and then minimizing the dust generation, Controlling and managing the generation of dust on the haul roads, either by suitable spraying or by using appropriate chemicals, suitable designed water sprays at the transfer points to suppress the dust is necessary.

2.7.1 .2 Mitigation to impact on water

Singh, (2008) suggested that the following measures can be adopted for optimizing the impacts of mining on the water system in the environment. These measures include: planning a good surface layout of both opencast and underground mines so that the impacts on the surface water bodies as well as the surface drainage system are the minimum, waste water from mining operation is needed to be well treated before discharge in the environment, Plan the opencast and underground mines with provisions for the development of underground and surface water bodies so that in the post mining period water in sufficient quantity is available for various uses. The concentration of metals like Arsenic toxic to human and aquatic life has resulted from mining activities in some instances. Hence, regulatory agencies in U. S. require gold mining companies to comply with Arsenic restrictive standards (Straskraba and Moran, 2006).

Acid mine drainage is mitigated by either Active Treatment Methods including: Limestone, Hydrated Lime, Soda Ash, Caustic Soda, and Ammonia addition to acid drain catchment to water bodies. Passive Treatment Methods include: Constructed Wetlands, Pyrolusite Process, Microbial Reactor System, Bio-sorption System, Limestone Ponds, Open Limestone Channels, Limestone Sand Treatment, Diversion

Wells, Anoxic Limestone Drains, Vertical Flow Reactor (Singh, 2008; Maddocks, 2009).

AMD follows the same flow pathways as water; therefore AMD can best be controlled by controlling water entry into the site of acid formation by diversion of surface water away from the residue storage areas, prevention of groundwater infiltration into the mine workings, prevention of hydrological seepage into the affected areas and controlled placement of acid-generating waste (Akcil and Koldas, 2006).

2.7.1 .3 Mitigation measures for impact on Land

Mining and associated activities are designed for the minimum possible land requirement adopted for the optimization of the impacts on land. The subsidence movements are predicted due to underground mining and their impacts on the land and its uses. In the case of opencast mines, the mine is planed with decommissioning, closure, reclamation and rehabilitation so that the land after mining can be brought to economic uses. In as much as possible provisions are made in opencast mining to separate removal and handling of top and sub-soils so that these can be re-laid at the time of reclamation for developing the land uses of the reclaimed surface (Singh, 2008).

2.8 Methods of reclamation

Jeanna, (2000) indicated that, there are a number of conventional remediation technologies which are employed to remediate environmental contamination with heavy metals such as solidification, soil washing and permeable barriers. Majority of these technologies are costly to implement and cause further disturbances to the already

damaged environment. Of all the methods, re-vegetation is the most documented and widely used because it is most practical and economical though it is often slow due to the hostile conditions such as toxicity of heavy metals (Troung, 1999; paradis, *et al.*, 2007). In the process of vegetative rehabilitation, the normal ecological succession using low-cost techniques should be initiated, so that the environment can enrich itself naturally in terms of biodiversity (Murdoch-Eaton *et al.*, 1997).

2.8.1 Practices of mine waste remediation/ reclamation

Currently, remediation practices used at mine sites to deal with waste rock typically involve re-shaping to reduce waste pile angles to 15-25 % (Bellairs, 1999), capping with clay or topsoil (Bell, 2001), and the addition of soil or waste amendments (attenuation) to stimulate plant growth and reduce the translocation of contaminants into the environment (Barnhisel *et al.*, 1998; Brown *et al.*, 2000).

Covers or encapsulation is another practice of remediation that has its structural and chemistry integrity likely to fail with 10 – 20 years, reported in a paper of (Wilson *et al.*, 2003; Ward *et al.*, 2000).

2.8.1.1 Challenges of waste rock remediation/ reclamation

There is the view that waste rock and tailings constitute the largest volume of material that must be stabilized and remediated. The fact is that they are often the hardest to remediate because of their chemical and physical properties (Kuhn, 1999; Fourie and Tibbett, 2007). The problems associated with remediating and revegetating acid waste rock are well documented and include metal toxicity, inherent acidity, high salt

concentrations, poor nutrient content and poor physical structure (Kabata Pendias and Pendias, 1992; Dollhopf, 1998; Alloway, 1995; Brown *et al.*, 2000). While much attention is paid to the remediation of soil media for the establishment of vegetation there is often a failure to comprehend the importance of the geotechnical properties of the soil and rock material within waste rock dumps (Kuhn, 1999).

2.8.1.2 Re-vegetation Strategies

Ghose, (2004), suggested re-vegetation strategies that included strategic clearing which implies that the area cleared should always be as small as necessary for the safe operation of the mine. Where possible, seeds are kept for the cleared vegetation, which can often be used during restoration as source of seeds. Where the restoration is aimed to restore the original ecosystem, the optimum time for clearing may be determined by the season when the important plant species bear seeds.

The quality of topsoil of the area is preserved during successful restoration, particularly where the objective is to restore a native ecosystem. If the topsoil contains large numbers of seeds of undesirable species, then it may be better to use the subsoil as a substrate for restoration (Biswas and Mukherjee, 1989 quoted by Ghose, 2004). The topsoil from all areas being cleared should be retained for subsequent restoration since the topsoil contains the majority of the seeds and other plant propagates (rhizomes, lignotubers, roots, etc.), soil microorganisms, organic matter and much of the more labile (more readily cycled) plant nutrients. The term 'topsoil' generally refers to the A soil horizon which is usually darker than the underlying soil because of the accumulation of organic matter (Braddy, 1988 quoted by Ghose, 2004). Topsoil is

removed after clearing the vegetative cover from the areas to be disturbed, before any drilling, blasting, mining, or other surface disturbance. It is essential that stripping should be carried out when the soil is as dry as possible. This will reduce to a minimum the risk of compaction and damage to the soil structure by smearing and remoulding. The routing of scrapers during this operation must be planned to minimize machine movement, which causes compaction and damage to the soil structure (Ghose, 2004).

The timing of topsoil stripping is also important for subsequent restoration. Soil is not stripped or replaced when they are too wet or too dry, as this can lead to compaction, loss of structure, and a loss of viability of seeds and mycorrhizal inocula (a natural ecosystem component that increases uptake of plant nutrients from the soil). The ideal moisture content to enable soil to resist damage will vary with different soils, therefore knowledge and experience will be required to determine when soils can be handled without damage (Ghose, 2004).

The topsoil is better to replace immediately on an area where the landform construction is complete. Direct replacement of topsoil has several advantages compared with placing the topsoil in stockpiles and storing it for later restoration. First, it avoids double handling. Second, the need to create stockpiles may mean that extra land must be cleared. Third, and most importantly, stockpiling reduces the quality of the soil resource. Stockpiles become anaerobic, soil structure deteriorates, organic matter and nutrients may be lost, seeds deteriorate, other plant propagates die and populations of beneficial soil microorganisms are reduced significantly. Species which do not have hard seed coats are particularly susceptible to loss during stockpiling (Ghose, 2004).

2.8.2 Phytoremediation in reclamation

Jeanna, (2000) reported that Phytoremediation is the use of green plants to clean-up contaminated hazardous waste sites. The report also reviews the rationale behind phytoremediation, stating that a significant advantage of phytoremediation is that a variety of organic and inorganic compounds are amenable to the phytoremediation process. Phytoremediation can be used either as an in situ or ex situ application. In situ applications are frequently considered because it minimizes disturbances of the soil and surrounding environment and reduce the spread of contamination via air and waterborne wastes. Another advantage of phytoremediation is that it is a green technology and when properly implemented is both environmentally friendly and aesthetically pleasing to the public. Phytoremediation does not require expensive equipment or highly-specialized personnel, and it is relatively easy to implement. It is capable of permanently treating a wide range of contaminants in a wide range of environments. However, the greatest advantage of phytoremediation is its low cost compared to conventional clean-up technologies (Jeanna, 2000).

Phytoremediation is a remediation technology which refers to the use of green plants and their associated micro biota for the treatment of contaminated soil and ground water (Sadowsky, 1999). The idea of using metal accumulating plants to remove heavy metals and other compounds was first introduced in 1983, but the concept has actually been implemented for the past 300 years (Henry, 2000). This technology can be applied to

both organic and inorganic pollutants present in soil (solid substrate), water (liquid substrate) and the air (Salt *et al.*, 1998).

Phytoremediation is however limited and disadvantaged in a very little way as documented by Jeanna, 2000 that contrast to its many positive aspects,. It is restricted to the rooting depth of remediative plants. Remediation with plants is a lengthy process, thus it may take several years or longer to clean up a hazardous waste site, and the contamination may still not be fully remediated. The use of invasive, non native species can affect biodiversity. The consumption of contaminated plants by wildlife is also of concern. Harvested plant biomass produced from the process of phytoextraction may be classified as a Resource Conservation and Recovery Act (RCRA) of 1979 hazardous waste, hence must be subjected to a proper handling and disposal. Unfavourable climate is another important consideration because it can limit plant growth and phytomass production, thus decreasing process efficiency.

2.8.3 Mechanisms Used for Phytoremediation

Jeanna, (2000) reviewed four phytoremediation processes; phytoextraction, phytostabilization, rhizofiltration, and phytovolatilization.

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2.8.3.1 Phytoextraction

Jeanna, (2000) documented that Phytoextraction is primarily used for the treatment of contaminated soils. To remove contamination from the soil, this approach uses plants to absorb, concentrate, and precipitate toxic metals from contaminated soils into ground

biomass such as (shoots, leaves). This method of treatment is also referred to as phytoaccumulation (United States Protection Agency Reports, 2000) and is the best approach to removing contamination from soil and isolating it without destroying the soil structure and fertility (Gosh and Singh, 2005).

Discovery of metal hyperaccumulator species demonstrates that plants have the potential to remove metals from contaminated soils. A hyperaccumulator is a plant species capable of accumulating 100 times more metal than a common non-accumulating plant. Thus, a hyperaccumulator will concentrate more than 1000 fg/g (0.1%) of Co, Cu, Cr, Pb, or 1% of Zn and Ni in their leaf dry matter. Most hyperaccumulator species accumulate Ni while others have been shown to accumulate Cd, Co, Cu, Zn. Currently there are no known Pb hyperaccumulators (Jeanna, 2000). Certain plants can extract lead from contaminated soils, but only when certain soil amendments have been added. There are several factors limiting the extent of metal phytoextraction including: metal bioavailability within the rhizosphere (root zone), rate of metal uptake by roots, proportion of metal "fixed" within the roots, rate of xylem loading/translocation to shoots and cellular tolerance to toxic metals (Jeanna, 2000).

Plant species that are applicable for this clean-up method, must extract large concentrations of heavy metals into their roots, translocate the heavy metal into the surface biomass, and produce a large quantity of plant biomass. In addition, remediative plants must have mechanisms to detoxify and/or tolerate high metal concentrations accumulated in their shoots (Jeanna, 2000).

2.8.3.2 Phytostabilization

Phytostabilization, also referred to as in-place inactivation, is primarily used for the remediation of soil, sediment, and sludges. It is the use of plant roots to limit contaminant mobility and bioavailability in the soil (Jeanna, 2000). This is mostly used for the remediation of soil, sediment and sludges (United States Protection Agency Reports, 2000). The plants primary used for this purposes are to decrease the amount of water percolating through the soil matrix, which may result in the formation of a hazardous leachate, act as a barrier to prevent direct contact with the contaminated soil and prevent soil erosion and the distribution of the toxic metal to other areas. Phytostabilization can occur through the sorption, precipitation, complexation, or metal valence reduction. It is useful for the treatment of lead (Pb) as well as arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu) and zinc (Zn) (Jeanna, 2000).

2.8.3.3 Rhizofiltration

Rhizofiltration is primarily used to remediate extracted groundwater, surface water, and wastewater with low contaminant concentrations. It is defined as the use of plants, both terrestrial and aquatic, to absorb, concentrate, and precipitate contaminants from polluted aqueous sources in their roots. Rhizofiltration can be used for Pb, Cd, Cu, Ni, Zn, and Cr, which are primarily retained within the roots. Sunflower, Indian mustard, tobacco, rye, spinach, and corn have been studied for their ability to remove lead from water, with sunflower having the greatest ability. In one study, after only one hour of treatment, sunflowers reduced lead concentrations significantly. Indian mustard has a bioaccumulation coefficient of 563 for lead and has also proven to be effective in

removing a wide concentration range of lead (Jeanna, 2000). Rhizofiltration can partially treat industrial discharge, agricultural runoff, or acid mine drainage. It can be used for Lead, Cadmium, Copper, Nickel, Zinc and Chromium, which are primarily retained within the roots (Chaudhry *et al.*, 1998).

2.8.3.4 Phytovolatilization

Phytovolatilization involves the use of plants to take up contaminants from the soil, transforming them into volatile forms and transpiring them into the atmosphere (Gosh and Singh, 2005). Phytovolatilization occurs as growing trees and other plants take up water as well as organic and inorganic contaminants. Some of these contaminants can pass through the plants to the leaves and volatilise into the atmosphere at comparatively low concentrations (Mueller *et al.*, 1999).

2.8.3.5 Phytodegradation

Phytodegradation is the breakdown of organics, taken up by the plant into simpler molecules that are incorporated into the plant tissues (Chaudhry, 1998). Plants contain enzymes that can breakdown and convert ammunition wastes, chlorinated solvents such as trichloroethylene and other herbicides (Gosh and Singh, 2005). The enzymes are usually dehalogenases, oxygenases and reductases (Black, 1995).

2.8.3.6 Rhizodegradation

Rhizodegradation is the breakdown of organics in the soil through microbial activity of the root zone (rhizosphere) and is a much slower process than phytodegradation (Gosh and Singh, 2005).

2.9 Indigenous plant species for reclamation

Those involved in rehabilitation projects have recognized the importance of using indigenous species not only for environmental reasons, but also to meet the livelihood and cultural needs of local communities who may depend on forest products and services (Sarrai and Ayrault, 2001).

Although the idea of using indigenous species in vegetation of slime dams is now widely accepted there is the need for continuous investigation into the establishment of indigenous species, which perform comparably to exotic species on mine tailings sites (Bell *et al.*, 2001; Ogola *et al.*, 2002; Naicker *et al.*, 2003; Mendez *et al.*, 2007; Schuwirth *et al.*, 2007).

In the study of Mendez *et al.* (2007) they demonstrated the suitability of indigenous qualibush [*Atriplex lentiformis* (Torr.) S. Wats], a perennial halophytic subshrub, in colonizing lead-zinc mine tailings. The halophytic subshrub improved the pH of mine tailings due to compost formation. Also the subshrub shoot-accumulated some metals such as Mn (400 to 1000mg/kg), Pb (30 to 100 mg/kg) and Zn (100 to 1000 mg/kg). Indigenous plants can easily adapt to the local conditions (Anh *et al.*, 2010).

Plant communities that are tolerant to imposed stress conditions can fulfill the objectives of stabilization, pollution control, visual improvement and removal of threats to mankind (Pratas *et al.*, 2004).

The constraints related to plant establishment and amendment of the physical and chemical properties of the toxic metalliferous soils depend upon the choice of appropriate plant species that will be able to grow in such a hostile environment. Thus, the plant community tolerant to toxic trace elements play a major role in remediation of degraded mine soils. Plants tolerant to toxic levels of trace elements respond by exclusion, indication or accumulation of metals (Baker, 1981).

Thus, more information about plant communities that are growing on metal-contaminated soils is essential to determine their application for mine stabilization/remediation and biogeochemical exploration (Baker *et al.*, 1994). The use of plant species as indicators of metal contamination (natural and/or anthropogenic) is based on their response to the elements present in the substrate (Kabata and Pendias, 2001). Plants growing on abandoned mine sites usually indicate the mineral composition of the soil. These plant species are tolerant to metals, and they are able to accumulate or exclude toxic metals. Thus, trace-metal-accumulating plants are of immense use for biogeochemical prospecting and geochemical exploration (Badri and Springuel, 1994). Therefore, metal-tolerant perennials and plants within Plant communities that are tolerant to imposed stress conditions can fulfill the objectives of stabilization, pollution control, visual improvement and removal of threats to mankind (Aziz, 2011).

Most plants are able to accumulate heavy metals from water and soil, of which some are very important for their growth and development (Fe, Mn, Zn, Cu, Mo and Ni); certain plants can accumulate metals which do not have high biological significance (Cd, Cr, Pb, Co, Se, Hg), but excessive concentration of heavy metals may be toxic for most plants (Rascio, 1997; Lasat, 1996).

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CHAPTER 3

MATERIALS AND METHODS

3.1 Study Area

The study area Newmont Ghana Gold (NGGL) is located at Ahafo Kenyasi the capital of Asutifi district in the Brong Ahafo region. The NGGL concession shares boundaries with the Sunyani Municipal in the North, Tano District in the North East, Dormaa District in the North West, Asunafo District in the South West and Ahafo Ano District (Ashanti Region) in the South East. It is located between the latitudes of 6° - $40'$ and 7° - $15'$ North and longitudes 2° - $15'$ and 2° - $45'$ West. The mine site includes three communities, namely, Ntotroso, Kenyasi No.1 and Kenyasi No.2. (en.wikipedia.org/wiki/kenyasi).

3.2.1 Climate condition of the area

The district lies within the wet semi-equatorial zone marked by double rainfall maxima; June and October with a mean annual rainfall between 125cm and 200cm. The first rainy season is from August to October. Relative humidity is generally high ranging between 75% to 80% during the two rainy seasons and 70% to 80% during the rest of the year (www.ghanaweb.com/./artikel.php).

3.2.2 Vegetation

The vegetation is moist semi-deciduous forest trees are Wawa, Odum, Mahogany Otee, Ohenna etc. Grasses found are; Elephant and spear. There are forest reserves which is over a total of about 475.6 square kilometers and 30% of the entire land surface area of the district.

3.2.3 Topography

The area lies within the forest dissected plateau physiographic region with average height of 700 feet above sea level. The lowest part is about 650 feet above sea level. The lowest level found along the river basins whilst the highest point is found within a chain of mountains in the north east reaching a height of 1400 feet above sea level (www.ghanaweb.com/artikel.php).

3.2.3 Geology and soil

The physiographic of the concession of mining area is under laid by Precambrian rocks of Birimian and Dahomeyan formation. The Birimian formations are known to be the gold bearing rocks. The Birimian rocks also have a high potential for manganese and bauxite (www.ghanaweb.com/artikel.php).

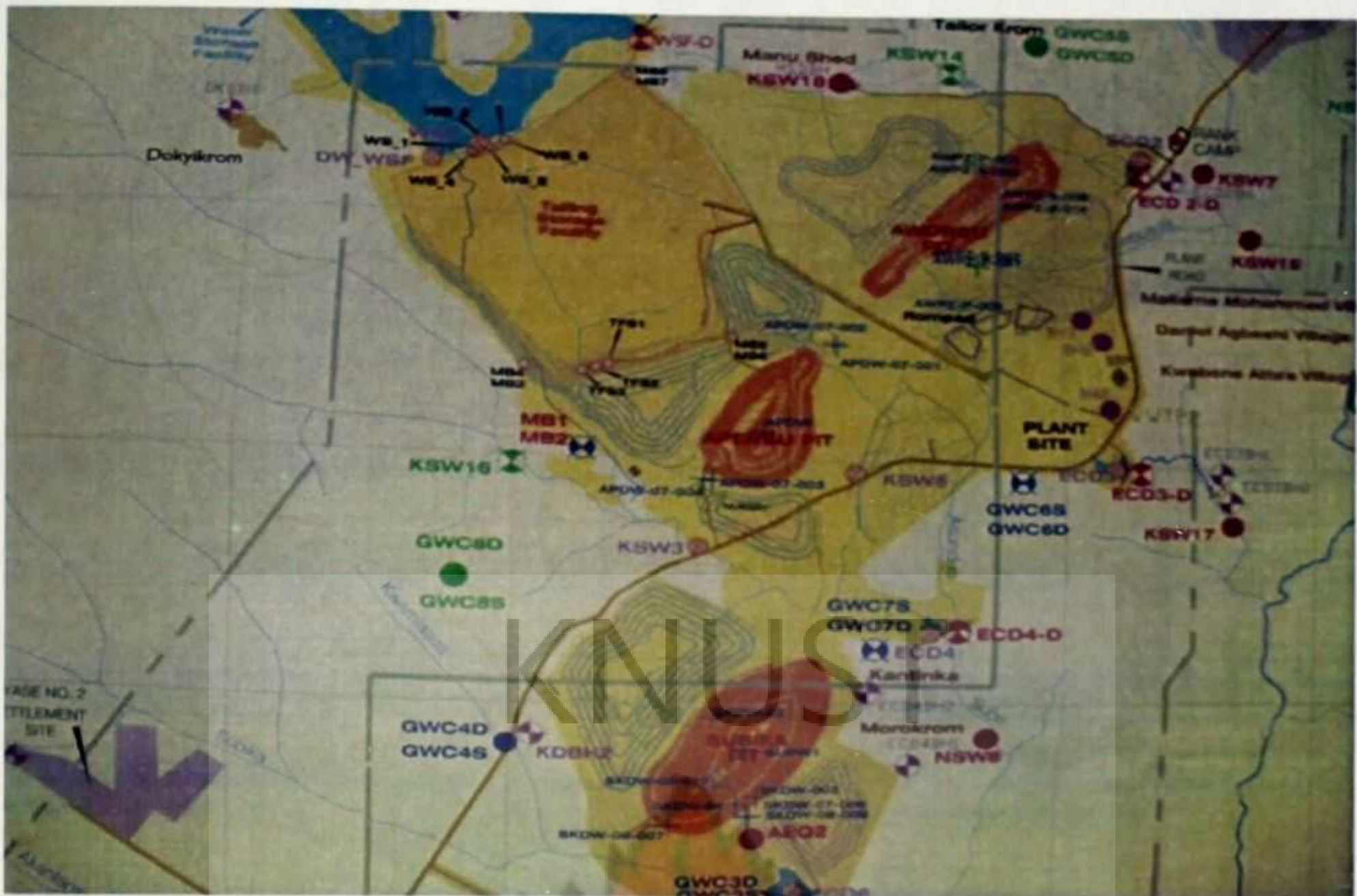


Figure 1. A map of Newmont Gold Ghana Ahafo showing the study site.

3.3 The Waste rock dump

Mining waste rocks are dumped at specified sites at the NGGL plant site. The waste has been dump to heap and form a hill. These waste rocks are from Birimian rock system as AGC, (2001) reported. (www.ghanaweb.com/artikel.php), also documented that, the Birimian formations of concession area are known to be gold bearing. On these bear waste rocks are different plant species of (herb, grass and shrub), growing.

3.4 The reclamation Plot

The study was conducted at Apensu reclamation plot located at NGGL plant site, Kenyasi. This site has been one of the waste rock dumps which is covered with soil to about 60cm level. The nature of the plot is sloppy with waste rock chips found at the

base. Even though the plot is filled with sub-soil and top-soil to 60cm level it still appears sloppy.

3.5 Experimental Design

The simple random technique was employed in selecting a waste rock dump for the study

Two bags of Waste rock chips with *Centrosema pubens* (legume) *Chromolaena odorata* (herb) and *Penissetum purpureum* (grass) growing on them were randomly sampled on the chosen waste rock dump. The plants species were sampled and placed in plastic bags and the corresponding waste rock chips among which they were growing were taken and placed and labeled in plastic bag. Soils from the Apensu reclamation plot were taken as a control to the experiment. At fortnightly intervals, two sampling were done within a month.

3.6 Sample Collection, preparation, storage and analysis

Waste rock chips from the waste rock dump was collected from two sampling points, Apensu (a) and Apensu (b), denoted as Aa and Ab. Ten kilogramme samples of waste rock chips each were taken from the two different point. Soil sample was taken from the Apensu reclamation plot to serve as control. Soil samples were taken at a depth of 30cm with soil auger. The samples were taken at both the top and the base of the plot. Soil samples were collected from about 30cm depth in the ground as most minerals of the soil are found within this depth. The plant species were collected were kept in a refrigerator at a temperature of 4°C. Samples of the selected plant species and waste

rocks were labeled in plastic. Samples were transported to the Soil Science and Chemistry Laboratory (Department of Crop and Soil Sciences) of Kwame Nkrumah University of Science and Technology (KNUST) for physicochemical and heavy metal analysis.

3.7 Laboratory Analysis

Organic carbon, organic matter determination, soil pH, moisture content, fresh and dry weights, digestion of milled rock chips, soil and plant samples were carried out at the laboratory of Department of Crop and Soil Sciences, KNUST. Analysis of soils waste rock chip for Phosphorous, total Arsenic, Copper, Lead, Iron, Manganese, and Zinc as well as analysis of plant species for these heavy metals were also carried out at the laboratory of Department of Crop and Soil Sciences, KNUST.

3.8 Data Collection

3.8.1 Waste Rock chip and Soil analysis

The waste rock chips were first crushed into smaller sizes as the coarse sand and milled into fine particles with a milling machine. This was done to determine and compare the levels of these parameters in the waste rock chips and control soil from reclamation plot.

3.8.2 Organic Carbon Determination

Organic carbon was determined by the Walkley-Black wet oxidation method (Schumacher, 2002). In this procedure 2.0 g of soil sample was weighed out into a 500 ml Erlenmeyer flask and 10 ml of 1.0 N Potassium dichromate solution added from a

burette. This was followed by the addition 20 ml conc. H_2SO_4 to generate heat to facilitate the reaction between Carbon and Cr_2O_7 . The mixture was swirled to ensure that the solution was in contact with all the particles of the soil. The flask and the content were allowed to cool on an asbestos sheet for 30 minutes. Two hundred (200) ml of distilled water was added, followed by 10 ml of orthophosphoric acid (to sharpen the colour change at the end point of titration). Diphenylamine indicator (2.0 ml) was added and the solution titrated with 1.0 ml ferrous sulphate solution until the colour changed to blue and then finally to a green end-point. The titre value was recorded and the blank solution corrected to (≥ 10.5). The organic carbon in the soil was then determined as follows:

$$\% \text{ organic C in soil} = \frac{(\text{m.e.K}_2\text{Cr}_2\text{O}_7 - \text{m.e.FeSO}_4) \times 0.003 \times \text{ferrous} \times 100\%}{\text{Weight of soil}}$$

3.8.3 Available Phosphorous

The Bray method (Bray and Kurtz, 1945) was used to determine available Phosphorous. Five grammes (5 g) of soil sample was weighed into a 100 ml flask and 35 ml of diethylene triamine pentaacetic acid (DTPA) added. This was shaken on a mechanical shaker for 10 minutes at room temperature and the solution filtered through a Whatman filter paper (Cat No 1001 110). Five (5) ml of the filtrate was poured into a 25 ml test tube and 10ml of cooling reagent added. A pinch of ascorbic acid was then introduced and the mixture stirred on a vortex mixture at 1500 rpm for 20 seconds. The solution was allowed to stand for 15 mins for colour development. An aliquot of this solution was put in a cuvette and placed in a Spectrum lab 23A spectrophotometer and the

phosphorous level recorded. Available Phosphorus was then calculated using the values obtained as follows:

Absorbance = $\frac{x}{0.0878}$, where x is the extracting factor (Olsen and sommers, 1982).

3.8.4 Soil and Waste rock (milled) pH

Soil and waste rock pH were measured with a pH meter (HI 8014) using a 1:2.5 of sample soil or rock chip (milled) to water ratio (Malik *et al.*, 2010). The pH meter was calibrated using standard buffer solutions (pH 4.01 and pH 7.01). The soil samples were first air-dried for 24 hours, ground and passed through a stainless steel mesh of 2 mm diameter. Five grammes (5g) of each soil and milled waste rock were weighed into a beaker. Distilled water was added (12.5 ml) and the mixture stirred vigorously for 15 seconds and left to stand for 30 minutes. The electrodes of the pH meter were placed in the slurry, swirled carefully, and the pH read and recorded.

3.8.5 Digestion of Soil Samples for Total Metal Content

Soil and milled rock samples were oven dried separately at 105°C for one hour and ground in a mortar into a fine powder. To 5g of each sample in a plastic bottle, 50ml of DTPA Extractant was added and shaken for 2 hours. Each of them was filtered using a Whatman 42 filter paper (Cat No 1001 43 110). The total concentrations of As, Fe, Cu, Pb and Zn in each of the filtrates was determined with the Atomic Absorption Spectrometer (AAS).



Plate 1. Atomic Absorption Spectrometer used to determine heavy metals in soil and plant samples.

3.8.6 About Atomic Absorption Spectrometry

The study will employ the atomic absorption spectrometry technique to analyze heavy metal concentration. A review about this technique is documented in (APHA – AWWA, 1998), Atomic Absorption Spectrophotometry is designed to determine the amount (concentration) of an object element in a sample, utilizing the phenomenon that the atoms in the ground state absorb the light of characteristic wavelength passing through an atomic vapor layer of the element.

Szalay, *et al.*, (2000) when using atomic absorption spectrophotometry (AAS) as an analytical technique the absorption of light of free atoms was measured. Therefore it is one of the branches of atomic spectroscopy, together with flame photometry (Standard base techniques: “Flame Photometry” that measures the intensity of light emitted by free atoms when their electrons return to ground state after the excitation by light). However - unlike flame photometry - AAS is based on the “first half” of the excitation process,

while atoms absorb light getting their electrons from the ground state to a higher energy level. Though is expensive technique but thanks to the fact that by AAS it is possible to determine about 70 elements (mainly metals) at very low concentrations.

3.8.7 Analysis of Total Metal Content

Filtrates obtained from the extraction were analyzed for total, Fe, Cu, Pb and Zn using a BUCK SCIENTIFIC Model 210 VGP AAS. Analysis was done by using Titration where 25ml of the extracted solution was measured into a 100ml volumetric flask. 1g ammonium chloride, 5ml of Ammonia solution and 25ml 0.1M magnesium sulphate were added to the content in the volumetric flask. The flask was filled to the 100ml mark with distilled water and was vigorously shaken. The set up was allowed to stand for 15minutes to settle. The solution was filtered with Whatman 42 filter paper. First 10 – 20ml filtrates were discarded. 50ml of remaining filtrate was heated to about 40 °C and titrated quickly after adding 1 drop of Eriochrome Black T (EBT) indicator with 0.1M Titriplex III solution until the colour changed to green. Calibration curves were prepared separately for all the metals by running different concentrations of standard solutions. The instrument was set to zero by running the respective reagent blanks. The digested solutions were aspirated individually and atomized in an air-acetylene flame. All samples were run in triplicates and average values taken for each determination. The detection limits for As, Fe, Cu, Pb and Zn were set at 0.001 mg^{-1}

3.9 Plant analysis

3.9.1 Fresh and Dry Weights

Total weights of plants were determined by taking the weights of the plants with a measuring scale immediately after harvesting. Dry weights were obtained by drying the plant materials in an oven at a temperature of 120°C for one hour. The weights were recorded and the samples returned into the oven for 30 minutes. This process was repeated until a constant weight was obtained after successive drying periods. The final weight was recorded as the dry weight.

3.9.2 Moisture Content

Moisture contents in the plants were obtained by taking the difference in the total and dry weights in each treatment. The percentage moisture content was obtained using the formula below:

$$\text{the percentage of moisture content} = \frac{\text{total weight of plant} - \text{dry weight of plant}}{\text{total weight of plant}} \times 100$$

3.9.3 Ashing and Digestion of Plant Materials for Total Metal Analysis

Samples of dry plants (whole plants) were broken into pieces and ground to fine particle. 1.00g of finely ground plant tissue of each plant species was weighed into porcelain crucibles and put in a muffle furnace and burnt at 500°C for four (4) hours. Samples were removed from furnace and cooled. The ignited residues were moistened with 2.0ml distilled water, and slowly and carefully 5.0ml of 8N HCl (that is 1: 2 water and conc. HCl) was added. The crucibles were covered and placed on water bath for 20 minutes. The content was filtered using Whatman No. 40 filter paper, obtaining the

filtrate in 100-ml Volumetric flask. The used crucibles were washed well with distilled water and the washing passed through the filter. The filter paper was washed several times with distilled water to make Volumetric to 100ml mark. The solution was then shook vigorously for complete mixing and was used for determination of the heavy metal concentration. Total As, Fe, Cu, Pb, Zn and Mn were determined with an AAS.

3.9.4 Bioaccumulation Factor (ratio) (Bf)

Bioaccumulation factor is the ratio of metal concentration in plant biomass to those in the soil. This factor was calculated for each of the plants in the different treatments by using the procedure described by Cai and Lena, (2003) this is also termed as Bioconcentration factor. The bioconcentration factor (BCF) of each plant species is determined by dividing the heavy metal content in plant shoots by the heavy metal concentration in the soil where the plant is growing as elaborated in (Anh et al. 2011) and expressed as; Bioconcentration Factor (ratio) (BCF) = $\frac{\text{metal concentration in plant}}{\text{Metal concentration in rock chip}}$ (Anh et al., 2010).

3.10 Data Analysis

The data for heavy metal concentrations of soil and plants (whole plants) under different soil treatments were analysed using the General Linear Model of SAS version 16 by analysis of variance (ANOVA) on ranks to compare the means of different treatments. The GLM procedure provide the 'P' value which if is less than 0.05 indicates significant difference while P value of greater than 0.05 indicates no significant difference. Where the difference was significant, the Student- Newman Keuls (SNK) Test was used to

separate the means. In the Student- Newman Keuls test treatments of the same letters have no significant difference while treatments of different letter indicate significant difference between treatment means.

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CHAPTER 4

RESULT

4.1 Selected identified indigenous plant species at Apensu waste rock dump

Among other plant species identified, these plant species were selected;

Chromolaena odorata

Centrosema pubens

Pennisetum purpureum

These plants were found on the waste rock dump where the particles size was small and the area was somewhat dry moist.

4.2. Soil and waste rock chip physicochemical properties

The physicochemical properties of the control soil and waste rock chip types are recorded in (Table 1). The pH for the waste rock chip was alkaline (pH of 9.6) whilst it was 6.7 acidic for the control soil. Available phosphorous (p) in the control soil was 0.14 fold higher than that of the waste rock.

Table 1. Physicochemical characteristics of waste rock and control soil

Properties	Waste rock chip (milled)	Control soil
pH	9.60±0.04	6.76±0.07
Moisture content (%)	0.15	1.5
Org Carbon (%)	0.51±0.01	0.87±0.04
Org Matter (%)	0.77±0.02	0.873±0.001
Available Phosphorous (ppm)	1.05	7.43

4.2.1 The particle size distribution of waste rock chip

The particle size distribution of waste rock chip was determined to be made up of 82.92% of sand, 12.2% of silt, and 4.96% of clay, hence with loamy sand texture. It also contained small rocks of size ranging from about 20 μ m -50 μ m.

4.3 Metal concentration in milled waste rock and control soil

From table 2, metal concentrations of Pb, Fe and Mn in waste rock were higher than their concentration in the control soil. However, concentrations of Zn and Cu in control soil were higher than their concentration in waste rock. The metal concentrations in both waste rock and control soil were below Maximum Allowable Concentrations (MAC), The Cu was below the Average Range (AR). The mean concentrations of heavy metal for Aa and Ab samples were not significantly different, however, were significantly different from the that of control soil. The largest metal concentration in waste rock was Fe with a value of 14.08mg/kg at sampling point Aa, whilst it was 14.211mg/kg at sampling point Ab. The least metal concentration recorded in waste rock was Pb with concentration of 0.133mg/kg. The largest metal concentration in control soil was Fe of 9.05mg/kg, however, Pb was the least metal concentration recorded to be 0.060mg/kg. The waste rock recorded Fe concentration which was significantly different from Fe concentration recorded in control soil, with a P value of 0.0003. Mn concentration in waste rock was significantly different from control soil with a P value of 0.0001. Zn concentration in waste rock was significantly different from control soil with a p value 0.0001. Cu concentration in waste rock was significantly different from control soil with

a p value 0.0003. However, As concentration recorded was not significantly different for waste rock and control soil.

Table 2. Metals concentration in waste rock chips and control soil (mg/kg)

Samples	Zn	Cu	Pb	Fe	Mn	As
Waste rock (milled) from Aa	0.600 ^b ±0.15	0.372 ^a ±0.03	0.123 ^a ±0.02	14.031 ^a ±0.01	0.707 ^b ±0.15	0.487 ^a ±0.034
Waste rock (milled) from Ab	0.59 ^b ±1.37	0.371 ^a ±0.17	0.122 ^a ±1.17	14.011 ^a ±0.01	0.708 ^b ±1.01	0.487 ^a ±0.034
Control soil	0.702 ^a ±0.02	0.204 ^b ±0.01	0.060 ^b ±0.01	9.05 ^b ±0.01	0.802 ^a ±0.13	0.467 ^a ±0.023

Within columns mean with the same letters are not significantly different (P > 0.05).

4.4 Metal concentration in plant

The heavy metal concentration; Zn, and Cu were about the same level, Fe recorded as the highest level while Pb is the lowest in *C. odorata*, *P. purpureum*. However, Cu level is about 39.48% more than Zn level in *C. pubensis*. The level of Mn is almost the same in all the three species.

Table 3: Metals uptake or concentration in selected plant species (mg/kg)

Samples	Zn	Cu	Pb	Fe	Mn	As
<i>Chromolaena odorata</i>	0.400 ^a ±0.003	0.460 ^a ±0.03	0.053 ^b ±0.003	13.4 ^a ±0.87	0.103 ^a ±0.003	0.597 ^a ±0.049
<i>Pennisetum purpureum</i>	0.300 ^b ±0.060	0.403 ^a ±0.003	0.043 ^b ±0.03	11.0 ^b ±1.15	0.107 ^a ±0.07	0.480 ^a ±0.021
<i>Centrosema pubens</i>	0.250 ^b ±0.020	0.430 ^a ±0.030	0.077 ^a ±0.003	15.73 ^a ±0.64	0.103 ^a ±0.003	0.480 ^a ±0.035

Within the columns of table 3 mean with the same letters are not significantly different (P > 0.05).

4.5 Metals concentration in waste rock chip compare to concentration plant species

The concentration of Zn in waste rock was 35.3%, 30.3% and 20.3% higher than the concentration in *C. pubensis*, *P. purpureum* and *C. odorata* respectively. Cu concentration in *C. odorata*, *P. purpureum* and *C. pubensis* are 9%, 6% and 3.3% respectively higher than the concentration in waste rock. Fe concentration in waste rock was 63% higher than *C. odorata* and 30.3% higher than in *P. purpureum* but was 170% higher in *C. pubensis* than in waste rock. However, there is a remarkable higher concentration of Pb and Mn in waste rock than all the plant species. These are recorded in Table 4.

Table 4. Metals concentration in waste rock chips and in the plant species

Mg/kg						
Samples	Zn	Cu	Pb	Fe	Mn	As
Waste rock (milled)	0.603 ^a ±0.003	0.370 ^a ±0.04	0.120 ^a ±0.006	14.03 ^b ±0.03	0.707 ^b ±0.007	0.487 ^a ±0.018
<i>Chromolaena odorata</i>	0.400 ^a ±0.003	0.460 ^a ±0.03	0.053 ^b ±0.003	13.40 ^b ±0.87	0.103 ^a ±0.003	0.597 ^a ±0.049
<i>Pennisetum purpureum</i>	0.300 ^b ±0.060	0.403 ^a ±0.003	0.043 ^b ±0.03	11.00 ^c ±1.15	0.107 ^a ±0.07	0.480 ^a ±0.021
<i>Centrosema pubensis</i>	0.250 ^b ±0.020	0.430 ^a ±0.030	0.077 ^a ±0.003	15.73 ^a ±0.64	0.103 ^a ±0.003	0.480 ^a ±0.035

Within the columns of table 4 mean with the same letters are not significantly different (P > 0.05).

Table 5. Metals concentration in control soil and in the plant species

Mg/kg						
Samples	Zn	Cu	Pb	Fe	Mn	As
Control soil	0.702 ^a ±0.02	0.204 ^b ±0.01	0.600 ^a ±0.01	9.05 ^a ±0.01	0.702 ^a ±0.13	0.467 ^c ±0.023
<i>Chromolaena odorata</i>	0.410 ^b ±0.002	0.480 ^a ±0.02	0.063 ^c ±0.006	12.60 ^b ±1.33	0.150 ^b ±0.003	0.460 ^c ±0.01
<i>Pennisetum purpureum</i>	0.410 ^b ±0.05	0.420 ^a ±0.001	0.043 ^d ±0.03	11.00 ^c ±1.15	0.250 ^b ±0.02	0.510 ^a ±0.021
<i>Centrosema pubensis</i>	0.290 ^c ±0.020	0.440 ^a ±0.029	0.070 ^b ±0.003	13.80 ^b ±1.27	0.103 ^b ±0.003	0.480 ^b ±0.035

Within the columns of table 4 mean with the same letters are not significantly different (P > 0.05).

4.7 Bioconcentration Factor

4.7.1 Determination of Bioconcentration factor of *C. odorata*, *P. purpureum* and *C. pubensis* in waste rock chip.

Table 6, 7 and 8, show the results of heavy metals concentrated by *C. odorata*, *P. purpureum* and *C. pubensis* with respect to the metal concentration in the waste rock chips (i.e. the ratio of the concentration of a metal in the plant to that of the same metal in the waste rock). The Bioconcentration factor (ratio) for Cu, As and Fe were the highest for all the plants, where the ratios of only Cu the three plants were all > 1 . The ratio of Zn were 0.672 0.685 and 0.55 for *C. odorata*, *P. purpureum* and *C. pubensis* respectively. The least bioconcentration factor (ratio) of Mn was recorded for the three plants as 0.133, 0.143 and 0.126 for *C. odorata*, *P. purpureum* and *C. pubensis* respectively.

Table 6. Bioconcentration factors of *C. odorata*

Samples	Zn	Cu	Pb	Fe	Mn	As
<i>C. odorata</i>	0.403	0.370	0.050	14.03	0.107	0.597
Waste rock	0.600	0.460	0.132	13.04	0.807	0.487
Bioconcentration Factor (BCF)	0.672	1.24	0.380	0.955	0.133	1.226

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Table 7. Bioconcentration factors of *P. purpureum*

Samples	Zn	Cu	Pb	Fe	Mn	As
<i>P. purpureum</i>	0.411	0.403	0.040	11.0	0.115	0.480
Waste rock	0.600	0.370	0.132	14.03	0.807	0.487
Bioconcentration Factor (BCF)	0.685	1.080	0.303	0.784	0.143	0.986

Table 8. Bioconcentration factors of *C. pubensis*

Samples	Zn	Cu	Pb	Fe	Mn	As
<i>C. pubensis</i>	0.332	0.430	0.080	15.73	0.102	0.480
Waste rock	0.600	0.370	0.132	14.03	0.807	0.487
Bioconcentration Factor (BCF)	0.55	1.16	0.600	1.120	0.126	0.986

CHAPTER FIVE

DISCUSSION

5.1 soil and rock chip physicochemical parameters

The physicochemical parameters of the waste rock chips and control soil (Table 1) showed that the waste rock chip has less amounts of Phosphorous (P) as well as lower organic matter and organic carbon content than that in the control soil. The waste rock chip was therefore of poor quality and inferior nutrient source for plant growth and development. The pH of the waste rock was basic, recorded as 9.60 whilst the pH of the control soil was slightly acidic 6.76. The percentage of organic matter, percentage of organic carbon and phosphorous of the waste rock were 0.77%, 0.51% and 1.05ppm respectively, whilst the percentage of organic matter, percentage of organic carbon and phosphorous recorded in control soil were 0.87%, 0.87% and 7.43ppm respectively. The basic or the alkaline pH of waste rock is a factor that affect heavy metal uptake by plant so as the slightly acidic pH of the control soil. These parameters significantly affect the available metal concentration in both soil and waste rock chip; the solubility and bioavailability are also significantly affected by soil pH. For instance, Bioavailability of Mn, Cu, and Zn decreases a 100 fold for each unit increases in soil pH (Lindsay, 1979).

The amount of exchangeable metal was also affected by the soil pH (Frageria *et al.*, 2000), therefore the high pH of waste rock could cause a great hindrances to metal uptake by these plant. Low organic matter and high pH level of 7 may lead to difficult metal uptake by plant (Aret, 2006).this implies that the low organic matter content of waste rock could have contributed to the least uptake of some metal by the plants. Whilst the high organic matter content of the control soil could have made it easy for the

plants to uptake the metal in it. The low level of phosphorus (1.05) in waste rock may also have hindered metal uptake by the plant and vice versa. However, (Murdock and Howe, 2001) reported that increase level of phosphorus and pH is strongly associated with reduced level of Zn in plant.

5.2 Metal concentrations in waste rock chip and control soil

Total concentrations of Cu, Zn and Mn in both the waste rock chip and control soils were within the medium critical values of DTPA extractable concentrations in soils, which are 0.3–0.5mg/kg, 0.4–1.5 mg/kg and 0.6–1.0mg/kg respectively (Jonson and Fixen, 1990). The total concentration of Fe in both waste rock and control soil were above the high critical values of DTPA extractable concentrations in soils. Arsenic was below the Maximum Allowable Concentrations (M.A.C) of heavy metals in both the waste rock chip and control soil (Lăcătușu *et al.*, 2009). These levels indicated that the metals concentrations were not harmful to the plants, except Fe concentration that appeared to be above the tolerant level by plants. Zn concentration in waste rock at both sampling points; Aa, and Ab were significantly different from the Zn concentration in the control soil, with a P value < 0.05 ($< .0001$). The separated means of Zn in waste rock and control soil were 0.60mg/kg and 0.71mg/kg respectively. The concentration of Zn was greater in control soil; however, the bioavailable Zn in the control soil could have been higher indicated by the Zn concentration in plants that had grown on the control soil. This could have been resulted from the pH difference of waste rock that was alkaline whilst that of the control soil was acidic. Cu concentration in waste rock chip at sampling points Aa and Ab were significantly higher than that of the control soil

indicated by the P value < 0.05 (< 0.0003). The separated means of waste rock and the control soil were 0.3733mg/kg and 0.2033mg/kg respectively. Cu in the control soil showed a greater bioavailability than in waste rock due to the weak acidic pH. Generally, Pb concentration was recorded as the lowest metal concentration in waste rock chip and was significantly higher than the concentration in control soil with a P value of 0.0001. The separated means were 0.1200mg/kg and 0.567mg/kg respectively. Thus, the concentration of Pb taken into plants was low. Lead (Pb) generally shows relatively little mobility from soils into plants. On normal soils, plants are typically found with < 10 mg/kg Pb (Reeves, 2005). The concentrations of Pb in both waste rock chip and control soil treatments were below 10 mg/kg. Zn, Cu and Pb were far below the Average Range, which are 25–200 mg/kg, 2–60 mg/kg and 10–150mg/kg respectively. (Lepp, 1981; Adriano, 2001). Fe metal had the highest concentration in the waste rock as well as the control soil. There was a significant different between the Fe concentration in these substrates with a P value of 0.0003. Their separated means were 14.033mg/kg and 8.400mg/kg respectively. Mn concentration in waste in rock at both sampling points was significantly different from the control soil. The P value was 0.0001. The separated means were 0.7066mg/kg and 0.80573mg/kg respectively. The Mn in soil was higher than in the waste rock may be due to the much bioavailable Mn in the control soil.

As concentration of both sampling points on the waste rock dump were not significantly different from the control soil with P value of 0.0656. The non significant different of As concentration in the waste rock chip could have been the fact that the dry formation of Gold crystallized out with arsenopyrovate (Asomaning, 2000), and this might have

caused the decrease in As concentration in waste rock. However, An Arsenic value in excess of 0.05mg/l in drinking water is considered a hazard to human health (Straskraba and Moran, 2006).

5.3 Metal concentration in plant species

C. odorata showed the highest hyperaccumulation for Fe with a concentration of 13.4mg/kg. The level of concentration of Zn, Cu, Mn, in *C. odorata* were close to concentration recorded in the waste rock. These and the control soil indicated that *C. odorata* was able to uptake the least bioavailability metal in soil. The concentration of As in *C. odorata* on both waste rock and control soil were above the level recorded in both substrates.

Little mobility of lead from soil to plant (Reeves 2005) was indicated in the study as *C. odorata* recorded the lowest lead concentration uptake. Similarly, the two other plant species indicated the same low uptake. *Pennisetum purpureum* showed a high concentration of Fe of 11.00mg/kg. For both waste rock and control soil. However, *P. perpureum* shows a lowest concentration of Pb. The concentration of Zn, Cu, Mn and As in *P. perpureum* on waste rock and control soil were closed to the levels of concentration in the substrates. This implied that Zn, Cu, Mn, and As were somewhat easily taken up by the plants. *Centrosema pubensis* showed a high Fe concentration of 15.7mg/kg and 13.8mg/kg on waste rock and control soil respectively. *C. Pubensis* also had the lowest concentration for Pb. *C. Pubensis* had Zn concentration of about half of the concentration in waste rock and in the control soil. In *C. pubensis*, Cu concentration was more than in control soil which were recorded as 0.43mg/kg and 0.44mg/kg

respectively. Mn and lead concentration in *C. pubensis* were low as compared to *C. odorata* and *P. purpureum*.

The three plant species selected from the same waste rock dump had total metal concentration that corresponded to the metal concentration in the waste rock chip with lowest concentration recorded for Pb in all the plant species. The heavy metal concentrations of; Zn, and Cu were about the same level in the three plants, Fe recorded as the highest level while Pb was the lowest in *C. odorata*, *P. purpureum*. However, Cu level was about 39.48% more than Zn level in *C. pubensis*. The level of Mn was almost the same in all the three species. Fe concentration in all plant species were above the high critical values of metal concentration, >10mg/kg (Lindsay and Norvell) and >5.0mg/kg (Jonson and Fixen). The clear indication was that all the three plant species have ability to accumulate Fe metal in their tissues. However, *C. pubensis* recorded the highest Fe concentration of 15.73mg/kg, as concentration in *C. odorata* was 13.41mg/kg and Fe concentration in *P. purpureum* was about 11.01mg/kg. The concentration of As in the plant tissues of both *P. purpureum* and *C. pubensis* were about the same concentration in the waste rock chip, except *C. odorata* whose As concentration slightly exceeded the concentration in the waste rock chip.

5.4 Comparison concentration in waste rock, control soil and the plant species

Generally the higher the concentrations of the metal in the waste rock and control soil, the higher the metal concentration in the plant species, and vice versa. The exception cases were; the concentration of Zn in waste rock which was higher than the

concentration in the plant species, Cu concentration in *C. odorata* and *P. purpureum* were lower than the concentration in the waste rock, except *C. pubensis* that had Cu concentration slightly above that in the waste rock. Pb had the least concentration recorded in waste rock with the corresponding lowest concentration in all the three plant species. The concentration of Fe was high in the waste rock and corresponding high concentrations were found in the tissues of both *C. odorata* and *P. purpureum*. However, *C. pubensis* tissues had a higher concentration of Fe than the waste rock. The concentration of Mn in waste rock was 89% higher than concentration of Mn found in the tissues of the three plants. Arsenic concentration in waste rock was about the same as the concentrations in the tissues of both *P. purpureum* and *C. pubensis*. Arsenic concentration in *C. odorata*, slightly exceeded the As concentration in the waste rock. *C. odorata* had a concentration of Zn that was significantly different from the Zn concentration in waste rock as well as that of Zn concentration in control soil. The separated means were 0.403mg/kg in *C. odorata* whilst 0.603mg/kg in waste rock and 0.7022mg/kg in control soil. There was also a significant difference in concentration of Pb between both the waste rock and the control soil. The separated means recorded were 0.1200mg/kg for waste rock 0.553mg/kg and 0.0776mg/kg for waste rock for *C. odorata* and control soil respectively.

Mn concentration in *C. odorata* was significantly different from both the waste rock chip and the control soil. The separated means were 0.706mg/kg for waste rock chip 0.103mg/kg for the control soil and 0.8033mg/kg for the control soil. The concentration of Fe and Cu for *C. odorata* were not significantly different from waste rock, however, Cu, Fe concentration in *C. odorata* were significantly different from control soil. The

separated means of Cu in *C. odorata* and control soil were 0.460mg/kg and 0.203mg/kg respectively. The means for Fe for *C. odorata* and control soil were 13.400mg/kg and 8.40mg/kg respectively. The Zn concentration in *P. perpureum* was significantly different from Zn concentration in waste rock chip and control soil. The means were 0.33mg/kg in *P. perpureum*; 0.603mg/kg and 0.710mg/kg in waste rock and control soil respectively. Pb concentration in *P. perpureum* was significantly different from waste rock chip; however, the Pb concentration was not significantly different from the Pb concentration in control soil. Mn concentration in *P. perpureum* was significantly different from the waste rock chip and control soil. The separated mean concentrations were 0.106mg/kg for *P. perpureum*, 0.706mg/kg in waste rock and 0.803mg/kg in control soil. *P. perpureum* showed a significant different concentration of Fe between both waste rock chip and control soil. Cu concentration in *P. perpureum* was significantly different from control soil; however, it was not significantly different from the concentration in waste rock chip. Zn concentration in *C. Pubensis* was also significantly different from concentration in both waste rock chips and control soil. The separated means recorded were 0.2500mg/kg in *C. Pubensis*, 0.6037mg/kg in waste rock and 0.7100mg/kg in control soil. Pb concentration in *C. Pubensis* was also significantly different from Pb concentration in waste rock chip and control soil. The mean concentration recorded was 0.076mg/kg comparing to 0.120mg/kg in waste rock and 0.050mg/kg in control soil respectively. *C. Pubensis* showed a concentration of Mn which was significantly in control soil different from Mn concentration in both waste rock chip and control chip. Both Cu and Fe concentration in *C. Pubensis* were not significantly different from concentration in waste rock. However, Cu and Fe

concentration were significantly different from the concentration in control soil. Arsenic concentration in *C. Pubensis* was not significantly different from both the waste rock and the control soil, since dry formation of Gold crystallizes out with Arsenopyrovate (Asomaning, 2000), could have led to low As concentration level in the waste rock chip.

5.5 Comparison of metal concentration in the plants

Concentration of Zn in *C. odorata* was significantly different from the Zn concentration in *P. purpureum* and *C. pubensis*. This indicated the different capability of the three different plant species to Zn uptake. The Pb concentration in *C. odorata* was significantly different from *C. pubensis*, but was not significantly different from *P. purpureum*. There was significant difference in Fe concentration for the three plants. *C. pubensis*, *C. odorata*, *P. purpureum*. Generally, the three plants had almost the same concentration of Mn and Cu since the difference in concentration was not significant on plants on both the waste rock and control soil.

5.6 Bioaccumulation factor (ratio) – Bf

The performance/ability of *Chromolaena odorata*, *Centrosema pubensis* and *Pennisetum purpureum*, as phytoextractors and potential hyperaccumulators was assessed by the bioaccumulation ratio of heavy metal concentration in plants to heavy metal concentration in the waste rock. Each plant species grown among waste rock of 0.603mg/kg concentration of Zn accumulated Zn of 0.40mg/kg, 0.30mg/kg and 0.25mg/kg in the whole plant of *C odorata*, *P purpureum* and *C. pubensis* respectively.

The bioaccumulations of Zn in waste rock were 0.672, 0.85, and 0.55 for the plant species respectively. However, each plant species grown in control soil of 0.702mg/kg, Zn concentrations accumulated were 0.41mg/kg, 0.41mg/kg and 0.290mg/kg in whole plant of *C. odorata*, *P. perpureum*, and *C. pubensis* respectively. The bioaccumulation ratios recorded for Zn in the plants were 0.58, 0.58 and 0.413 respectively.

The plants on waste rock chip showed a highest bioaccumulation ratio of Zn than those on control soil, this could have been due to the highest phosphorus of the control soil. The increasing level of phosphorus and pH are strongly associated with reduced level of Zn in plant (Murdock and How, 2001).

On waste rock chip of Cu concentration of 0.37mg/kg, there was accumulation of 0.460mg/kg, 0.403mg/kg and 0.43mg/kg for *C. odorata*, *P. perpureum* and *C. pubensis* respectively. The bioaccumulation ratios for Cu in waste rock were 1.24, 1.08, and 1.16 for *C. odorata*, *P. perpureum* and *C. pubensis* respectively.

The plants growing on control soil of 0.204 mg/kg there was accumulated Cu of 0.480mg/kg, 0.42mg/kg and 0.44mg/kg in *C. odorata*, *P. perpureum*, and *C. Pubensis* respectively. The bioaccumulation ratios of Cu in *C. odorata*, *P. perpureum*, *C. pubensis* on the control soil were 2.352, 2.059 and 2.15 respectively.

On controls soil of 0.60mg/kg pH concentration of *C. odorata*, *P. perpureum*, *C. pubensis* accumulated pH of 0.063mg/kg, 0.04mg/kg and 0.07mg/kg respectively. The corresponding bioaccumulation ratios were 0.1005, 0.07 and 0.11. On waste rock of 0.12mg/kg Pb concentration; *C. odorata* *P. perpureum* and *C. pubensis* accumulated

0.053mg/kg, 0.043mg/kg and 0.077mg/kg. The corresponding bioaccumulation ratios were 0.440, 0.358 and 0.642 of *C. odorata*, *P. perpureum* and *C. pubensis* respectively.

Generally, the plant showed the least hyperaccumulation capacity toward Pb. Pb is least mobile heavy metal in substrate, hence least bioavailable and exchangeable in the rhizosphere.

On the waste rock of 14.03mg/kg Fe concentration, *C. odorata*, *P. perpureum*, *C. pubensis* accumulated Fe concentration of 13.4mg/kg, 11.0mg/kg and 75073mg/kg. The corresponding bioaccumulations of these plants were 0.955, 0.784, and 1.120 respectively. In control soil of 9.05mg/kg of Fe; *C. odorata*, *C. pubensis* and *P. perpureum* accumulated 12.6mg/kg, 11.0mg/kg and 13.8mg/kg of Fe respectively. The bioaccumulation ratios showed by the plants were 1.392, 1.215, and 1.525 respectively. Fe was highly hyper accumulation by *C. pubensis* followed by *C. odorata* in both the waste rock and control soil. However the conditions or factors like the pH, phosphorus and organic matter content enhance the Fe metal uptake by the plant.

Waste rock chip of 0.707mg/kg concentration of Mn had *C. odorata*, *P. perpureum*, and *C. Pubensis* accumulating 0.107mg/kg and 0.103mg/kg respectively. The corresponding bioaccumulation ratio was 0.146, 0.151, and 0.146 for each plant species respectively.

The control soil with 0.702mg/kg of Mn, *C. odorata*, *C. Pubensis* and *P. Perpureum* accumulated 0.15mg/kg, 0.025mg/kg and 0.103mg/kg of the Mn respectively. The corresponding bioaccumulation ratios 0.214, 0.356, 0.147 respectively to each plant species. This is indicating that *C. Pubensis* had the highest capacity to hyper accumulate Mn in any metal contaminated a soil. The *C. odorata*, *P. perpureum*, and *C. pubensis* on

the waste rock chip of 0.487mg/kg accumulated 0.597mg/kg, 0.480mg/kg and 0.480 mg/kg respectively. The corresponding bioaccumulation ratios were 1.015, 1.092, and 1.228 in each plant respectively.

This indicated that *C. odorata* had a higher hyperaccumulation of As. However, all the three plants showed a good capacity to accumulate As, therefore they all could be used to phytoremediate heavy metal contaminated substrates. This was an indication that *C. odorata* has the greatest ability to hyperaccumulate Cu as compared to the other two plant species. *C. pubensis* also had the greatest ability to hyperaccumulate Fe as compared to other plant species. *C. odorata* shows a hyperaccumulation performance in the descending order as; Cu > As > Fe > Zn > Pb > Mn. The descending order of hyperaccumulation performance of *P. purpureum* is indicated as; Cu > As > Fe > Zn > Pb > Mn. The descending order of hyperaccumulation performance of *C. pubensis* is indicated as; Cu > Fe > As > Pb > Zn > Mn.

According to Gosh and Singh (2005), the bioavailability of metals can be increased in soil through several means. One way plants achieve this is by secreting phytosidophores into the rhizosphere to chelate and solublise metals that are soil bound (Kinnerseely, 1993). Both acidification of the rhizosphere and exudation of carboxylates are considered potential targets for enhancing metal accumulation, reported by (Aziz, 2011). Even with the high pH value ($pH\ 9.60 \pm 0.04$) of the waste rock chip substrate these plant species had the potential to take up the least bioavailable metal present in the waste rock substrate.

CHAPTER SIX

CONCLUSION AND RECOMMENDATIONS

6.1 Conclusion

Degradation of land by waste rock dump from mining industries can be ideally reclaimed by re-vegetation. Of all the methods, re-vegetation is the most documented and widely used because it is most practical and economical though it is often slow due to the hostile conditions such as toxicity of heavy metals. The plant species to use in the re-vegetation exercise are mainly indigenous to the area. The potential of three indigenous plant species; *Chromolaena odorata*, *Centrosema pubens* and *Pennisetum purpureum*, found growing on waste rock were analyzed.

The metal constituent and mineral constituent of the substrate (waste rock) on which the plant species were found was determined. The basis to use plant species as indicators of metal contamination (natural and/or anthropogenic) is the response to the elements present in the substrate. The Zn, Cu, As, and Mn concentrations in the waste rock indicated a level within the critical medium value in a soil. As these concentrations were not near harmful concentration to plants in soil, therefore, the waste rock may not contribute to metal contamination in soil. However, may contribute to contamination of water, for instance, since 0.05mg/L of Arsenic may be hazardous to human health. However, the Fe concentration in the waste rock exceeded the high critical value in soil indicating that leachate from the waste rock dump may be contributing to Iron contamination of surrounding soil and water if such drainage finds its way into the environment.

For plants to survive and thrive in soils with high concentrations of heavy metals, plants can either stabilize metal contaminants in the soil through avoidance or they can take up the contaminants into their cellular structure by tolerating them. There were significant difference in concentration of Zinc, Copper, Manganese Lead and Iron. in the waste rock as compared with the corresponding metal concentration in *Chromolaena odorata*, *Pennisetum purpureum* and *Centrosema pubensis*. Arsenic concentration was not significantly different. This indicated that the plant species were tolerant to Zn, Cu, Mn, Pb, As and Fe, hence able to uptake and accumulate them in spite of the least bioavailability.

The three plants had a potential to uptake any of the metal in their tissues, however, *Chromolaena odorata* greatly accumulate Cu followed by As, Fe, Zn, Pb with least accumulation ability for Mn. *Pennisetum purpureum* greatly accumulated Cu followed by As, Fe, Zn, Pb and with a least accumulation ability for Mn. *Centrosema pubensis* greatly accumulate Cu followed by Fe, As, Pb, Zn and also showing least accumulation ability for Mn. Neither of the plant species was able to show exceptional accumulation ability for Lead and Manganese with respect to the concentration present in the waste rock.

Chromolaena odorata, demonstrated the potential to highly uptake Copper and Arsenic than *Centrosema pubensis*. *Pennisetum purpureum* and *Centrosema pubensis* also demonstrated the potential to highly uptake Iron than *Chromolaena odorata* and *Pennisetum purpureum*. Both, *P. purpureum* and *C. pubensis* reviewed the potency to accumulate As, since the concentration of arsenic in waste rock was about the same to the concentration in these two plants. On the other hand As, concentration in *C. odorata*,

slightly exceeded the As concentration in the waste rock which may indicate its high potency to accumulate Arsenic. The three plants had a requirement that was of great significance for the accumulation of heavy metal or toxic metals. That was the ability of plants to tolerate the metals extracted from the substrate, that lead to the due survival of the plants on the waste rock dumps.

6.2 Recommendations

Heavy metals in soils and water continue to receive increasing attention due to the growing scientific and public awareness of environmental issues. Re-vegetation and Phytoremediation, the use of plants to ameliorate degraded or polluted substrates, is a technology with considerable promise for remediating and restoring contaminated sites. The continuous urgency for contaminated site clean up in a developing country like Ghana, demands that Re-vegetation phytoremediation should be given a careful and immediate consideration.

Continuous research in this field is necessary and future experiments could look at the metal constituent and concentration in leachate from waste rock compared to constituent and concentration in waste rock. The performance of *C. odorata*, *P. purpureum*, and *C. pubensis* on substrate degraded with heavy metals could be assessed.

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APPENDIX A

Cu (mg/kg)				
Reclamation plot soil (control soil)	Waste rock chip	<i>Chromolaena odorata</i>	<i>Punnistumpurpureum</i>	<i>Centrosema pubens</i>
0.20	0.40	0.40	0.40	0.50
0.20	0.30	0.50	0.41	0.40
0.21	0.41	0.48	0.40	0.40
0.204±0.01	0.37±0.04	0.46±0.04	0.403±0.03	0.43±0.07

Fe (mg/kg)				
Reclamation plot soil (control soil)	Waste rock chip	<i>Chromolaena odorata</i>	<i>Punnistumpurpureum</i>	<i>Centrosema pubens</i>
9.00	14.00	15.0	9.00	15.0
8.00	14.00	12.0	13.0	15.2
8.21	14.11	13.2	11.0	17.0
8.40±0.59	14.03±0.08	13.4±1.60	11.0±2.0	15.73±1.27

Mn (mg/kg)				
Reclamation plot soil (control soil)	Waste rock chip	<i>Chromolaena odorata</i>	<i>Punnistumpurpureum</i>	<i>Centrosema pubens</i>
0.80	0.70	0.10	0.10	0.10
0.81	0.70	0.10	0.10	0.10
0.80	0.72	0.11	0.12	0.11
0.81±0.01	0.72±0.02	0.11±0.01	0.12±0.02	0.11±0.01

Pb (mg/kg)				
Reclamation plot soil (control soil)	Waste rock chip	<i>Chromolaena odorata</i>	<i>Punnistumpurpure um</i>	<i>Centrosema puberulum</i>
0.060	0.13	0.050	0.040	0.080
0.060	0.11	0.050	0.050	0.080
0.050	0.12	0.060	0.040	0.070
0.057±0.03	0.12±0.01	0.053±0.07	0.043±0.07	0.073±0.07

Zn (mg/kg)				
Reclamation plot soil (control soil)	Waste rock chip	<i>Chromolaena odorata</i>	<i>Punnistumpurpure um</i>	<i>Centrosema puberulum</i>
0.70	0.61	0.40	0.20	0.30
0.70	0.60	0.40	0.30	0.20
0.73	0.61	0.41	0.40	0.25
0.71±0.02	0.61±0.01	0.40±0.01	0.30±0.10	0.25±0.05

As (mg/kg)				
Reclamation plot soil (control soil)	Waste rock chip	<i>Chromolaena odorata</i>	<i>Punnistumpurpure um</i>	<i>Centrosema puberulum</i>
0.49	0.48	0.60	0.47	0.42
0.47	0.46	0.68	0.45	0.54
0.44	0.52	0.51	0.52	0.48
0.47±0.02	0.49±0.03	0.60±0.08	0.48±0.04	0.48±0.06

Critical limits for DTPA – extractable micronutrients

Micronutrients (µg/g soil)

	Zn	Cu	Fe	Mn
Very low	0.05-0.5	0.0-0.1	0.0-2.0	0.0-0.5
Low	0.5-1.0	0.1-0.3	2.0-4.0	0.2-1.2
Medium	1.0-3.0	0.3-0.8	4.0-6.0	1.2-3.5
High	3.0-5.0	0.8-3.0	6.0-10.0	3.5-6.0
Very high	>5.0	>3.0	>10.0	>6.0

Source: Lindsay and Norvell, 1978.

**Critical soil test values of DTPA extractable
Cu, Zn, Fe, and Mn**

Table 1.1

Nutrient content (mg kg⁻¹)

Nutrients	Low	Medium	High
Zn	<0.9	0.4-1.5	>1.5
Cu	<0.3	0.3-0.5	>0.5
Fe	<3.0	3.0-5.0	>5.0
Mn	<0.6	0.6-1.0	>1.0

Source: Jonson and Fixen, 1990.

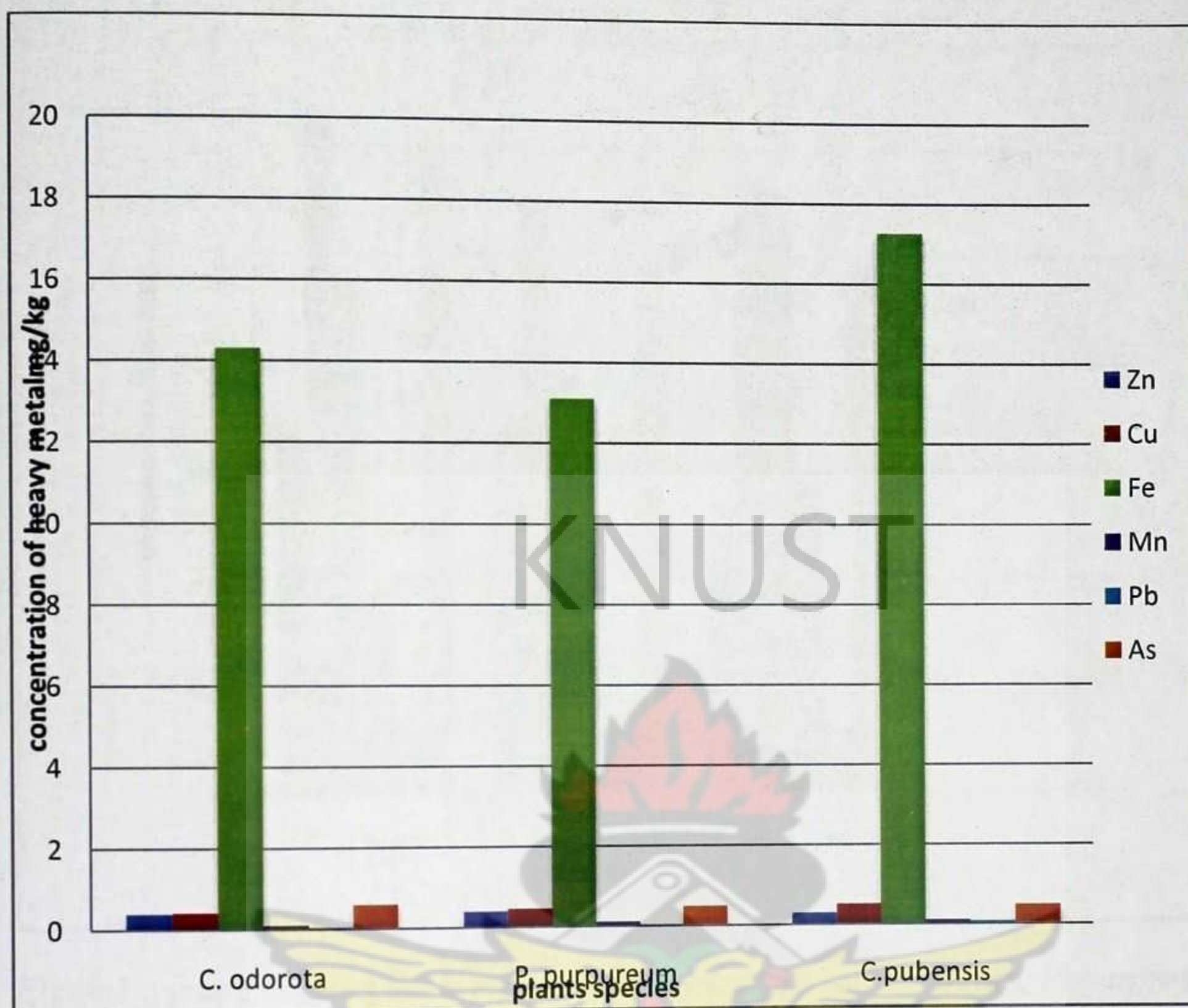


Fig2 a graph comparing the metal concentration in *C. odorata* *P. purpureum* *C. pubensis*.

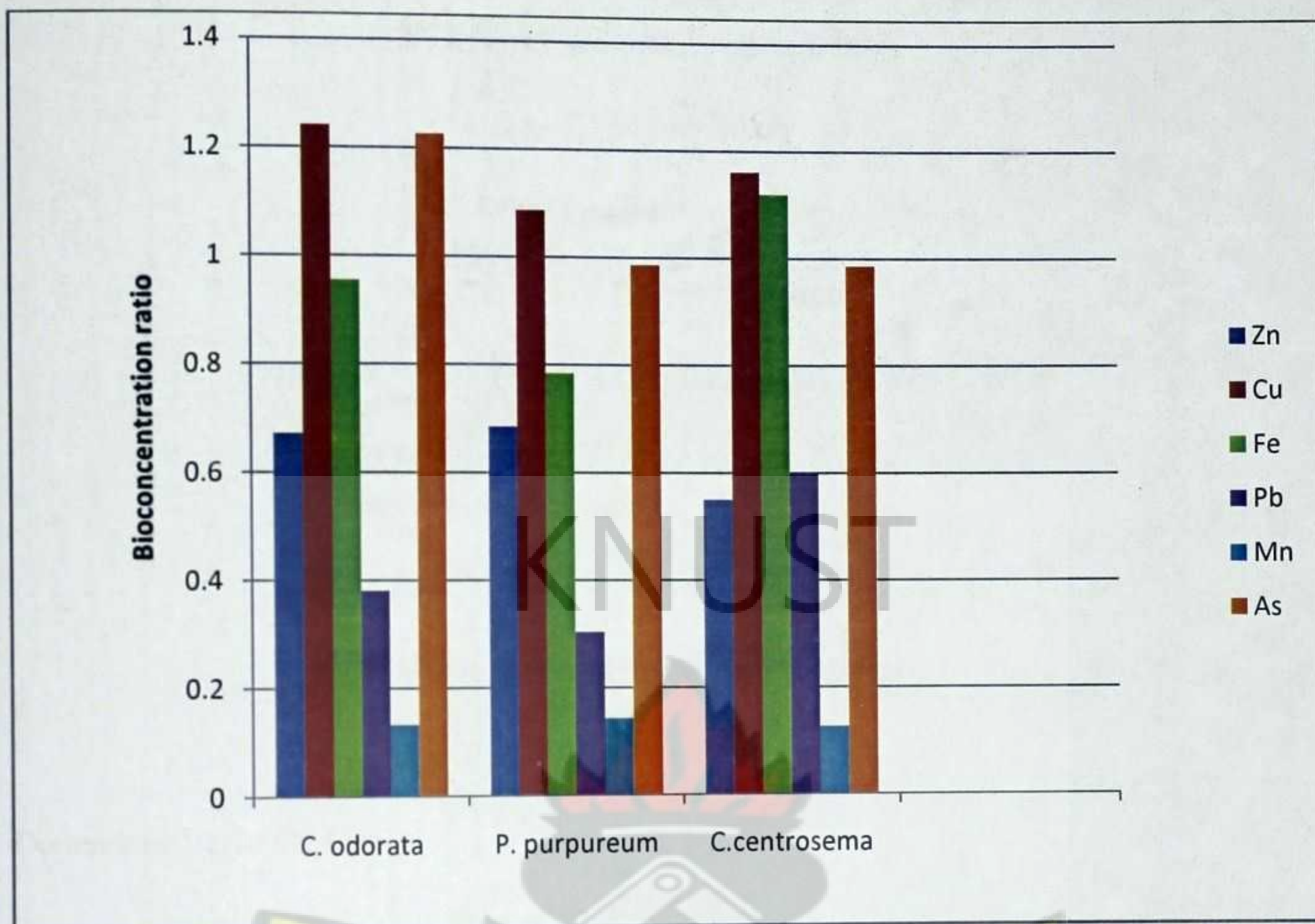


Figure3 a graph showing the bioaccumulation of metals by *C. odorata*, *P. purpureum* *C. pubensis*

Waste rock and metal concentration

The GLM Procedure

Class Level Information

Class Levels Values

Treat 5 CENTR CHOD PENNI RECPLT WCP

Number of observations 15

Waste rock and metal concentration

The GLM Procedure

Dependent Variable: Cu

Source	DF	Sum of Squares	Mean Square	F Value	Pr> F
Model	4	0.12270667	0.03067667	14.84	0.0003
Error	10	0.02066667	0.00206667		
Corrected Total	14	0.14337333			

R-Square	CoeffVar	Root MSE	Cu Mean
0.855854	12.13361	0.045461	0.374667

Source	DF	Type III SS	Mean Square	F Value	Pr> F
Treat	4	0.12270667	0.03067667	14.84	0.0003

Waste rock and metal concentration

The GLM Procedure

Dependent Variable: Fe

Source	DF	Sum of Squares	Mean Square	F Value	Pr> F
Model	4	98.0240000	24.5060000	15.76	0.0003
Error	10	15.5533333	1.5553333		
Corrected Total	14	113.5773333			

R-Square	CoeffVar	Root MSE	Fe Mean
0.863060	9.966409	1.247130	12.51333

Source	DF	Type III SS	Mean Square	F Value	Pr> F
Treat	4	98.02400000	24.50600000	15.76	0.0003

Waste rock and metal concentration

The GLM Procedure

Dependent Variable: Mn

Source	DF	Sum of Squares	Mean Square	F Value	Pr> F
Model	4	1.53764000	0.38441000	5241.95	<.0001
Error	10	0.00073333	0.00007333		
Corrected Total	14	1.53837333			

R-Square	CoeffVar	Root MSE	Mn Mean
0.999523	2.348306	0.008563	0.364667

Source	DF	Type III SS	Mean Square	F Value	Pr> F
Treat	4	1.53764000	0.38441000	5241.95	<.0001

Waste rock and metal concentration

The GLM Procedure

Dependent Variable: Pb

Source	DF	Sum of Squares	Mean Square	F Value	Pr> F
Model	4	0.01113333	0.00278333	59.64	<.0001
Error	10	0.00046667	0.00004667		
Corrected Total	14	0.01160000			

R-Square	CoeffVar	Root MSE	Pb Mean
0.959770	9.759001	0.006831	0.070000

Source	DF	Type III SS	Mean Square	F Value	Pr> F
Treat	4	0.01113333	0.00278333	59.64	<.0001

Waste rock and metal concentration

The GLM Procedure

Dependent Variable: Zn

Source	DF	Sum of Squares	Mean Square	F Value	Pr> F
Model	4	0.46720000	0.11680000	45.39	<.0001
Error	10	0.02573333	0.00257333		
Corrected Total	14	0.49293333			

R-Square	CoeffVar	Root MSE	Zn Mean
0.947796	11.19001	0.050728	0.453333

Source	DF	Type III SS	Mean Square	F Value	Pr> F
Treat	4	0.46720000	0.11680000	45.39	<.0001

Waste rock and metal concentration

The GLM Procedure

Dependent Variable: As

Source	DF	Sum of Squares	Mean Square	F Value	Pr> F
Model	4	0.03424000	0.00856000	3.12	0.0656
Error	10	0.02740000	0.00274000		
Corrected Total	14	0.06164000			

R-Square	CoeffVar	Root MSE	As Mean
0.555483	10.42729	0.052345	0.502000

Source	DF	Type III SS	Mean Square	F Value	Pr> F
Treat	4	0.03424000	0.00856000	3.12	0.0656

Waste rock and metal concentration

The GLM Procedure

Student-Newman-Keuls Test for Cu

NOTE: This test controls the Type I experiment wise error rate under the complete null hypothesis but not under partial null hypotheses.

Alpha 0.05
Error Degrees of Freedom 10
Error Mean Square 0.002067

Number of Means	2	3	4	5
Critical Range	0.0827048	0.1017522	0.1135584	0.1221598

Means with the same letter are not significantly different.

SNK Grouping	Mean	N	Treat
A	0.46000	3	CHOD
A	0.43333	3	CENTR
A	0.40333	3	PENNI
A	0.37333	3	WCP
B	0.20333	3	RECPLT

Waste rock and metal concentration

The GLM Procedure

Student-Newman-Keuls Test for Fe

NOTE: This test controls the Type I experiment wise error rate under the complete null hypothesis but not under partial null hypotheses.

Alpha	0.05
Error Degrees of Freedom	10
Error Mean Square	1.555333

Number of Means	2	3	4	5
Critical Range	2.2688574	2.7913893	3.1152693	3.3512344

Means with the same letter are not significantly different.

SNK Grouping	Mean	N	Treat
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A	15.733	3	CENTR
A			
A	14.033	3	WCP
A			
A	13.400	3	CHOD
B	11.000	3	PENNI
C	8.400	3	RECPLT

Waste rock and metal concentration

The GLM Procedure

Student-Newman-Keuls Test for Mn

NOTE: This test controls the Type I experiment wise error rate under the complete null hypothesis but not under partial null hypotheses.

Alpha	0.05
Error Degrees of Freedom	10
Error Mean Square	0.000073

Number of Means	2	3	4	5
Critical Range	0.0155792	0.0191672	0.0213912	0.0230114

Means with the same letter are not significantly different.

SNK Grouping	Mean	N	Treat
A	0.803333	3	RECPLT
B	0.706667	3	WCP
C	0.106667	3	PENNI
C			
C	0.103333	3	CHOD
C			
C	0.103333	3	CENTR



Waste rock and metal concentration

The GLM Procedure

Student-Newman-Keuls Test for Pb

NOTE: This test controls the Type I experiment wise error rate under the complete null hypothesis but not under partial null hypotheses.

Alpha	0.05
Error Degrees of Freedom	10
Error Mean Square	0.000047

Number of Means	2	3	4	5
Critical Range	0.0124279	0.0152902	0.0170643	0.0183568

Means with the same letter are not significantly different.

SNK Grouping	Mean	N	Treat
A	0.120000	3	WCP
B	0.076667	3	CENTR
C	0.056667	3	RECPLT
C			
C	0.053333	3	CHOD
C			
C	0.043333	3	PENNI

Waste rock and metal concentration

The GLM Procedure

Student-Newman-Keuls Test for Zn

NOTE: This test controls the Type I experiment wise error rate under the complete null hypothesis but not under partial null hypotheses.

Alpha	0.05
Error Degrees of Freedom	10
Error Mean Square	0.002573

Number of Means	2	3	4	5
Critical Range	0.0922876	0.113542	0.1267161	0.1363142

Means with the same letter are not significantly different.

SNK Grouping	Mean	N	Treat
A	0.71000	3	RECPLT
B	0.60333	3	WCP
C	0.40333	3	CHOD
D	0.30000	3	PENNI
D	0.25000	3	CENTR

Waste rock and metal concentration

The GLM Procedure

Student-Newman-Keuls Test for As

NOTE: This test controls the Type I experiment wise error rate under the complete null hypothesis but not under partial null hypotheses.

Alpha	0.05
Error Degrees of Freedom	10
Error Mean Square	0.00274

Number of Means	2	3	4	5
Critical Range	0.0952293	0.1171612	0.1307553	0.1406593

Means with the same letter are not significantly different.

SNK Grouping	Mean	N	Treat
A	0.59667	3	CHOD
A			
A	0.48667	3	WCP
A			
A	0.48000	3	PENNI
A			
A	0.48000	3	CENTR
A			
A	0.46667	3	RECPLT

KNUST



Waste rock and metal concentration

The MEANS Procedure

		N			
	Treat	Obs	Variable	Mean	Std Error
Mn Pb	CENTR	3	Cu	0.4333333	0.0333333
			Fe	15.7333333	0.6359595
	CHOD	3	Cu	0.4600000	0.0305505
			Fe	13.4000000	0.8717798
Mn Pb	PENNI	3	Cu	0.4033333	0.0033333
			Fe	11.0000000	1.1547005
	RECPLT	3	Cu	0.2033333	0.0033333
			Fe	8.4000000	0.3055050
Mn Pb	WCP	3	Cu	0.3733333	0.0371184
			Fe	14.0333333	0.0333333