RADIOLOGICAL AND ELEMENTAL

CHARACTERIZATION OF SOIL AND MILL TAILING SAMPLES FROM ANGWAN MAIGERO GOLD MINING SITES, NIGER STATE, NIGERIA.

BY

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DEPARTMENT OF PHYSICS,
FACULTY OF PHYSICAL SCIENCE
AHMADU BELLO UNIVERSITY, ZARIA
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DECEMBER, 2016

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DEPARTMENT OF PHYSICS,
FACULTY OF PHYSICAL SCIENCE,
AHMADU BELLO UNIVERSITY, ZARIA
NIGERIA

DECEMBER, 2016

DECLARATION

I hereby declare that the work in this dissertation titled "**Radiological and Elemental Characterization of Soil and Mill Tailing Samples from Angwan Maigero Gold Mining Sites, Niger State, Nigeria**" has been carried out by me in the Department of Physics, Ahmadu Bello University, Zaria, under the supervision of Prof. I.G.E. Ibeanu and Dr. Y.I. Zakari. The information derived from various literatures have been duly acknowledged, cited and listed. No part of this work has been presented for another degree or diploma at any institution.

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Name of student	**Signature**	**Date**

CERTIFICATION

This dissertation titled "**Radiological and Elemental Characterization of Soil and Mill Tailing Samples from Angwan Maigero Gold Mining Sites, Niger State, Nigeria**" meets the regulations governing the award of the of Master's Degree in Radiation Biophysics of Physics Department of Ahmadu Bello University, Zaria and is approved for its contribution to knowledge and literary presentation.

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Dean, School of Postgraduate Studies	**Signature**	**Date**

DEDICATION

I dedicate this work to the Almighty God, whom in His infinite mercy granted me His grace to successfully complete this work and to my parents who gave the financial support to make this work a success.

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I am greatly indebted to my supervisors, Prof. I.G.E. Ibeanu and Dr. Y.I. Zakari for their guidance, enormous contributions and support towards the successful completion of this work. I am also grateful to the entire staff of Physics Department for the various contributions, suggestion and encouragement during this study.

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ABSTRACT

Mining activities has been observed to be one of the major contributors of heavy metals in the environment. The effect of heavy metals on gold mining areas is of serious concern, as such threatens life in all forms. This present study involves the determination of heavy metal concentration and its pollution levels in Angwan Maigero Gold mining sites, Niger State Nigeria using Contamination Factor (CF), Pollution load Index (PLI), Degree of Contamination (DC) Index, Geo-accumulation Index (I-geo), Hazard Quotient (HQ) and Ecological Risk Factor (ERF). Soil and mill tailing samples were analyzed using Flame Atomic Absorption Spectroscopy (FAAS) to obtain their composition and concentration. The study area was observed to be contaminated with Pb, Fe, Mn, Cu, Zn, As, Cr, Co, Ni, and Cd. The mean concentrations (ppm) of the studied heavy metals in mine site was observed to be 319.765, 1707.631, 181.614, 90.945, 37.412, 17.992, 93.187, 47.940, 22.282 and 3.232 respectively, Also the mean concentrations (ppm) in the mill tailing site was observed to be 846.867, 1914.993, 177.703, 739.333, 100.405, 43.519, 163.284, 53.476, 45.591, 5.828, respectively. The Highest concentration corresponds to Fe and the lowest corresponds to Cd. The increasing trend was in the order: Cd <As <Ni <Zn <Co <Cu <Cr <Mn <Pb <Fe. The concentrations of all the elements in the mill tailing sites were observed to be higher than that obtained in the mine site with greater variation in relative abundance of most of the metals. These variations can be attributed to the variable dumping of mill tailing deposit. The extremely high contents of the heavy metals are terribly alarming in terms of environmental population, when compared with the world recommended value by WHO. Therefore, the inhabitants and the mine workers, who reside and work at the environment, are at serious risk of heavy metal toxicity and awareness needs to be created. Also, the radionuclide concentrations were determined using the sodium iodide [NaI

(TI)] detector with a low background configuration. The mean activity concentrations of ^{226}Ra, ^{232}Th and ^{40}K were observed to be 43.94 ± 3.02, 38.61 ± 2.06 and 304.17 ± 23.03 Bq/kg at the mine site and 45.39 ± 5.07, 72.41 ± 7.18 and 472.16 ± 35.26 Bq/kg at the mill tailing site, respectively. Effects of different parameters of interest on measured values of gamma dose rates for the region under investigation was carried out. The mean absorbed gamma dose rate (D) was found to be 53.94 ± 2.84 and 84.39 ± 5.21 nGy/h in the mine and mill tailing sites, respectively; the mean annual effective dose equivalent (AEDE) in the mine and mill tailing sites, respectively was found to be 0.07 ± 0.003, in line with the world wide effective dose of 0.07mSv/y and 0.10±0.01mSv/y, higher than the world mean value. The mean values Ra_{eq} was found to be lower than the accepted safety limit value of 370Bq/kg. The ELCR was calculated and was found to be $(0.23 \pm 0.01) \times 10^{-3}$ below the world mean value of (0.29×10^{-3}) in the mine site while the mean value of the mill tailing was found to be $(0.36 \pm 0.02) \times 10^{-3}$ above the world mean value.

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CHAPTER 1: INTRODUCTION

1.1 Background of Research

Mining activities has been observed to be one of the major contributors of heavy metals in the environment. The effect of heavy metals on gold mining areas is of serious concern, as such threatens life in all forms. The exposure of man to background radiation from natural sources is an inevitable feature of life on Earth. The natural source of ionizing radiation originates from the terrestrial and cosmic origin. Natural resources are important in the development of any country. Mining refers to the process of extraction of mineral deposits from the surface of the earth. It can only take place where minerals are present and economically viable. The development of natural resources worldwide involves the manipulation of the environment to arrive at specific objectives. Mining refers to the process of extraction of mineral deposits from the surface of the earth. It can only take place where minerals are present and economically viable. Mineral resources are the most exploited natural resource and it involves extracting, grinding, ore concentration and dispersal of tailings (Ferreira, *et al*., 2004). The activities which are carried out through mining of the minerals and processing of the ores most often generate a lot of radioactive contamination heaps in the environment resulting in radiological hazards, chemical wastes and cause various degrees of damages and threats to plants, animals as well as human life (Girigisu, 2012). Mining can generate large concentrations of highly soluble inorganic matter, some of which are considered toxic (Mousa, 1997). Generation of chemical waste as a result of mining activity occurs world-wide and may severely affect natural resources such as vegetation, streams and the ecosystem in general (Ramani, 2001).

According to (Girigisu, *et al.*, 2014), the composition and processing of ores also determine the nature of pollutants. After the precious metals which are usually the ore associates are extracted, varying concentrations of other undesirable inorganic and radioactive elements such as arsenic, copper, lead, zinc, iron, sulphate, cyanide, nitrate, calcium, and magnesium are usually passed on to residual materials known as mill tailings (Ferreira, *et al*., 2004). The tailings together with the sulphide-bearing mine waste rock are often exposed to the weather, thereby resulting in the mobilization of metals and other chemical compounds related to ore processing into nearby water-bodies. Elevated concentrations of these elements in the water bodies pose serious health hazards to host communities (Girigisu, 2012). According to changes in their physicochemical states and the prevailing chemical conditions, these metal contaminants migrate and threaten human health by contaminating soil, streams, ground water and food crops (Girigisu, 2012).

This research focuses on possible problems related to gold mining in Angwan Maigero communities, a settlement in Rafi local government area of Niger state with particular focus on the possibilities of radiological and elemental contamination hazards.

Gold mining can have devastating effects on nearby water resources. Long term effects include acid mine drainage, which raises acid level in rivers and streams, buildup of high concentration of radon gas and its decay products as well as contamination of soil and water way by heavy metals resulting from tailings (Ferreira, *et al*., 2004). In Angwan Maigero mining exercises, the panning/sedimentation processes were carried out right in the middle of stream-bed, which is, as at the study period, in its seasonal water-flow breaks. Water used for panning/sedimentation in these cases are obtained either from the borehole project near the stream bed or from holes dug at various locations in the bed

itself. Heavy metals associated with gold mining waste can dissolve and disperse into surrounding streams due to rain water percolating through the waste (Ferreira, *et al.,* 2004) or dispersed as particles through erosion processes. Impacts of goldmines can be environmentally detrimental and can constitute serious radiological hazards to exposed or contaminated individuals.

Majority of people working in the mines and those associated with various activities within the vicinity of the mines as well as the processing zones are largely unaware of health implications they are likely to be subjected to due to radiations from exposed rock surfaces, crushed and milled rocks and generally, from pilled-up tailings. The degree of ignorance exhibited by workers in small scale mining to the possibilities of their exposure to ionizing radiation shows in their attitudes in and around the mines.



Plate I Illegal miners displaying their haul

In Angwan Maigero mines, as a case for consideration, mine workers have gone deep into the belly of the rock (ahead of 20m in depth) by burrowing with the aid of crude instruments do spend **some of their rest periods in these rocks**. With this nature of implements, these workers in the caves are by no means always in very close proximity to freshly exposed radioactive rock surfaces over the work periods. The hours of rest and work all spent inside the belly of these rocks cumulatively put worker in within a radiation exposure range of 9-12 hours per day.

The pathways of radiation exposure in small scale mining operations are numerous ranging from exposure at the mining sites, mill tailing sites to raise indoor radon levels in the make-shift shops where trading occurs in the mining area. Based on this exposure

pathways and the level of ignorance of the mine workers, the research work becomes a necessity to investigate possibilities of raised radiation level, if it exists and to quantify such level as indications of degree of radiological risk these workers are likely to be subjected to. The work also extends to identification of toxicological properties of the elements in a view of determining possibilities of contamination of the masses in the workplace.

1.2 Statement of the Problem

A recent lead poisoning outbreak has claimed the lives of 28 children from about 65 confirmed cases in some villages in Rafi local government area of Niger state (The Sun, 15th May, 2015). The outbreak was to the new illegal mining sites from where high leaded ores were brought home for crushing and processing by illegal miners (The Sun, 15th May, 2015). However the awareness of possibilities of exposure to ionizing radiation and relative degree of its absorption to human body being factor of time, distance and shielding are information that most mine workers are oblivious of.

Additional integrating problem is often generated by radioactive contamination of the mining area and its environs by tailings and acid mine drainage (AMD). To a very great extent, local miners are ignorant of these stated sources of health problems.

Mining operation in Angwan Maigero has generated a lot of survival related activities such as petty trading, hard and conventional drug peddling and prostitution, developments which exposes people in these groups to radiation from exposed radioactive rock surface, milling processes, tailings, and raised indoor radon levels all of which cumulatively are of grave danger to their health.

The pathway of the seasonal stream, where panning and sedimentation used to take place may be a source of radiation hazards to all users of the stream for agricultural and domestic purposes due to possibility of exposure by contamination from radionuclides in tailings and toxic elements or in dissolved stream water as well as ones taken up by plants.



Plate II Panning/ sedimentation process pollutes water bodies

The seemingly non-existence of government impact in terms of provision of social services to the area, education of communal people about risks to health of the mining operation as well as other related matters that require government participation has serious integrating effects on possibilities of radiation absorption and impacts of toxic elements on these communal people.

1.3 Justification of the Study

Natural background radiation comes from cosmic rays from our solar system and radioactive elements normally present in the soil (Fasae, 2013). Cosmic rays and the natural terrestrial radionuclide sources in soil, air, food and drinking water are the major contributors to worldwide radiation exposure. Sources of heavy metals in the environment

include man made sources of ionizing radiation due to technological advancement such as mining and ore processing operations, nuclear weapon testing, nuclear power reactor and medical radiation in form of diagnostic x-rays. It has been established by (Ibeanu, 1999) that mining areas are highly unsafe radiologically. The determination of the effects of gold mining activities in Angwan Maigero communities will therefore help in environmental monitoring and protection.

Mining activities has been observed to be one of the major contributors of heavy metals in the environment. Heavy metals accumulation in soil is toxic to man and animals, exposure to this heavy metals normally leads to various health problems, when inhaled or ingested, can cause damage to brain, kidney, bone marrow and other body organs. It also interferes with the development of the nervous system and therefore particularly toxic to children. As such it becomes paramount to use the toxicological models of the heavy metals studied in a view of determining possibilities of contamination of the masses in the workplace and community.

1.4 Significance of the Study

The study results may provide information pertinent for future mining legislation and code 7 practice of workers.

The generated data from the study may serve as basis for the public sensitization of the health and radiological hazards of the endangered settlers, mine workers and other related groups of people in the area, of environmental and most importantly radiological and toxicological risks involved by the mining exercise.

The study outcome will help workers, having been educated, to know how to combat anticipated problems as well as adopting protective measures.

1.5 Aim and Objectives

The aim of this study is to assess possibilities of radiation exposure and elemental contamination in Angwan Maigero communities gold mining and ore processing activities.

The objectives of this study are:

1. To measure the gamma radiation dose rate (in-situ) as baseline data.
2. To determine the levels of ^{226}Ra, ^{232}Th and ^{40}K in soil and mill tailing samples and their radiological implications.
3. To determine the elemental composition in soil and mill tailing samples and assess their corresponding pollution levels.

1.6 Scope and Limitation

The extraction and exploitation of gold in Niger state is mainly carried out in Angwan Maigero community in Rafi local government area of the State. The work intends to characterize soil samples for elemental composition and distribution using Flame Atomic Absorption Spectrometry (FAAS) technique, characterize soil samples for NORMs activity concentrations levels using [NaI(Tl)] gamma spectrometry technique and take in-situ dose rate measurements. Furthermore, use risk assessment codes to evaluate metal toxicity in soil and hazard indices to determine the radiological risk associated with mining and ore processing activities in the areas. The study is limited to the extraction and exploitation of

gold by local miners classified as artisanal and small scale gold miners in Madaka district of Rafi Local Government and its effects on the residents of the communities.

CHAPTER 2: LITERATURE REVIEW

2.1 Introduction

The exposure of man to ionizing radiation from natural sources is an inevitable phenomenon of life on earth. Background radiation in soil is a fact of life, as such varies greatly depending on the geological location and altitude man find himself. Man has been exposed to radiation continuously right from the creation of the earth. The natural source of ionizing radiation originates from the terrestrial and cosmic origin. Living organisms are continuously exposed to ionizing radiation from Naturally Occurring Radioactive Materials (NORMs). The origin of these materials lies in the earth crust, but they find their way into soil, building materials, air, water, food and the human body itself. In many parts of the world, building materials containing radioactive materials have been used for generations. As individuals spend more than 80% of their time indoor, the internal and external radiation exposure from building materials creates prolonged exposure situation (ICRP, 1991). The Earth is naturally radioactive, and about 90% of human radiation exposure arises from natural sources such as cosmic radiation, exposure to radon gas and terrestrial radionuclides (Lee, *et al.,* 2004). However, it has been observed that the type and concentration vary considerably depending on the soil type. The effects of the radiation emitted by different radionuclides depend on the over lining soil material (thickness and type), its chelating agents and physico-chemical properties (Belivermis, *et al*., 2009). Exposure to excess level of background ionizing radiation can cause somatic and genetic effects that tend to damage critical organs of the body, and can even lead to death (Ajayi and Ajayi, 1999).

Artificial sources of exposure have appeared due to such human activities as nuclear power plants, nuclear weapons testing, production and use of radioactive sources. In addition, human activities such as mining, milling and processing of mineral sands uranium ores, manufacture of fertilizers and burning of fossil fuels have raised naturally occurring radioactive material concentrations in the environment.

Mining and processing of radioactive ores contribute significantly to the enhancement of terrestrial radiation fields in the environment (Ibeanu, 1999). The earth's crust contains small amounts of uranium, thorium, and radium as well as radioactive isotope of potassium.

The disposal of waste rocks/tailings are directly related to the composition of the ore materials, after mining as well as some chemicals for processing the valuable (target) minerals from the gangue minerals, become the most important environmental issues for Artisanal miners. The oxidation of sulphide rich waste rocks/tailings produces acid mine/rock drainage, requiring restorative treatment prior to disposal into the environment (Bertrand, *et al*., 2000).

Heavy metals occur naturally as chemical elements in the earth's crust and surface soil in varying concentrations. (Iyaka & Kakulu, 2011). The persistent accumulation of heavy metals in soils is of great concern because they constitute health threat and toxicity problems to human life and environment (Wild, 1994).

Our environment is naturally radioactive and has been so since the creation of the world. It has been known that, naturally occurring radioactive materials becomes the focus of regulatory interest with the publication of the International Atomic Energy Agency (IAEA) (Augustine & Aku, 2013). Soils from abandoned mine sites may contain naturally

occurring radionuclide significant amounts and the resulting external radiation exposure pathway to the population have been the subject for study in the field of radiation protection and measurement (Faweya & Babalola, 2010).

Toxic elements from wastewater may contaminate agricultural soils, water supplies and environment and hence human food chain. The metals present in crops can pose serious risk in the consumers in terms of carcinogenesis, mutagenesis and teratogenesis (Radwan, *et al*., 2006).

Arsenic, cadmium, lead and mercury, and their inorganic compounds, are toxic to human health as well as environment also (Lokeshappa, *et al.*, 2012). Other metals of concern in form of soluble compounds also lead to plausible adverse health effect. Arsenic is regarded as human carcinogen from extremely low levels of exposure, having no possible beneficial metabolic functions for humans (NAS/NRC, 1999).

In Nigeria, open dump is almost the verily available option for solid waste disposal, even in the capital cities. Sanitary Landfill, however, is rare and unpopular, except perhaps among few institutions and few affluent people. Financial and institutional constraints are the immediate identifiable reasons for this in Nigeria and some other developing countries, especially where local governments are weak or underfinanced and rapid population growth continues (Nunan, 2000; Elaigwu, *et al.*, 2007).

The infectious medical wastes, toxic and domestic wastes are disposed together (Odunaiya, 2002). The soil is the primary recipient of solid waste (Nyle & Ray, 1999). These wastes end up interacting with the soil system thereby changing the physical and chemical properties (Piccolo & Mbagulu, 1997). The accumulation of contaminants is aided by the

capacity of soil to bind with clay minerals and organic substances. Their accumulation has multiple effects on the usability and functions of soil in the ecosystem.

2.2 Gold Mining and Lead Exposure

Artisanal small scale mining and other industrial activities create lead contamination that affect children worldwide and the grinding of leaded rock is usually done with flour mills and with mortars and pestles (CDC,2010).

Lead is highly toxic and can cause damage to the brain, kidneys, bone marrow and other body systems in humans especially young children (Needleman, 2004). Lead exposure among children is associated with developmental problems, including impaired cognitive function, reduced intelligence, impaired hearing and reduced stature, no toxicologically safe blood lead level (BLL) has been identified (Canfield, *et al.,* 2003).

At high BLLs, lead can cause convulsions, coma and death (Needleman, 2004). Exposure to lead dust through ingestion or inhalation can result in high BLLs in children. In the united states, the centers for disease control and prevention (CDC) defines, an elevated child BLL as a BLL for initiating public health actions and recommends that children with BLLs $\geq$ 45μg/dl receive intensive medical management and chelation therapy (CDC, 2002). Among developing countries, major sources of children lead poisoning include lead mining and smelting, paint leaded gasoline, battery recycling and traditional medicine (Meyer, *et al.,* 2008).

2.3 Heavy metal pollution in Mineral Mining and Ore Processing Operations

Heavy metals are natural components of the earth's crust. They cannot be degraded or destroyed. To a small extent they enter our bodies via food, drinking water and air. Heavy metals are dangerous because they tend to bio-accumulate (Kilic, 2009). The term "heavy metal" refers to any metallic element that has a relatively high density and is toxic or poisonous even at low concentration (Lenntech, 2004). "Heavy metals" is a general collective term, which applies to the group of metals and metalloids with atomic density greater than $4g/cm^3$ or 5 times or more greater than water (Garbarino, *et al.,* 1995, Hawkes, 1997)). However, being a heavy metal has little to do with density but concerns chemical properties. Heavy metals include lead (Pb), cadmium (Cd), zinc (Zn), mercury (Hg), arsenic (As), silver (Ag), chromium (Cr), copper (Cu), iron (Fe), and the platinum group elements. Many of these metals enter the environment and remain environmental problems due to pollution properties. Most of these environmental pollutants also occur naturally through natural weathering processes. However, published reports also indicate that mainly mining and smelting plants release metals from the bed-rock (Walker & Sibly, 2010).

Heavy metals are referred to interchangeably as trace elements depending on the field of study or context in which it is being discussed (Girigisu, 2012). In this work, the toxicological model of the metals studied will be used to determine the level of contamination of the masses at the workplace. Artisanal mining and ore processes activities can result to various degrees of metal pollution which are detrimental to human health and the environment. Thus, these heavy metals can be as a result of its abundance in the Gold mining sites.

2.3.1 Heavy Metal Poisoning and Bio-toxicity

The bio-toxic effects of heavy metals refer to the harmful effects of heavy metals to the body when consumed above the bio-recommended limits. Although individual metals exhibit specific signs of their toxicity, the following have been reported as general signs associated with cadmium, lead, arsenic, mercury, zinc, copper and aluminum poisoning: gastrointestinal (GI) disorders, diarrhea. Stomatitis, tremor, hemoglobinuria causing a rust-red colour to stool, ataxia, paralysis, vomiting and convulsion, depression and pneumonia when volatile vapors and fumes are inhaled (McCluggage, 1991).

Lead is the most significant toxin of the heavy metals and the inorganic forms are absorbed through ingestion by food and water and inhalation (Ferner, 2001). A notably serious effect of lead toxicity is its teratongenic effect. Lead poisoning also causes inhalation of the synthesis of hemoglobin; dysfunctions in the kidneys, joints and reproductive systems, cardiovascular system and acute and chronic damage to the central nervous system (CNS) and peripheral nervous system (PNS) (Ogwuegbu & Muhanga, 2005). Other effects include damage to the gastrointestinal tract (GIT) and urinary tract resulting in bloody urine, neurological disorder and can cause severe and permanent brain damage. While inorganic forms of lead, typically affect the CNS, PNS, GIT and other bio systems. Organic forms predominantly affect the CNS (McCluggage, 1991; INECAR, 2000; Ferner, 2001; Lenntech, 2004). Lead affects children by leading to the poor development of the grey matter of the brain, thereby resulting in poor intelligence quotient (IQ) (Udedi, 2003). Its absorption in the body is enhanced by Ca and Zn deficiencies. Acute and chronic effects of lead result in psychosis.

As with lead and mercury, arsenic toxicity symptoms depends on the chemical form ingested (Ferner, 2001). Arsenic acts to coagulate protein forms complexes with coenzymes and inhibits the production of adenosine triphosphate (ATP) during respiration (INECAR, 2000). It is possibly carcinogenic in compounds of all its oxidation states and high-level exposure can cause death (Ogwuegba & Ijioma, 2003).

Metal leaching is a naturally occurring process. Under certain conditions if the trace elements in the original media are in anomalously high concentrations they may have a negative on the receiving environment. Metal leaching is generally associated with acid mine drainage (AMD), however it can occur under non AMD conditions.

Concentrations of trace elements in soil are important to soil scientists not only for environmental purposes, such as quantifying contamination but also to help solve problems associated with human and plant toxicity. The elements typically studied are Ag, Al, As, Ba, Ca, Cd, Co, Cr, Cu, Fe, Hg, K, Mg, Mn, Na, Ni, P, Pb, Sb, Sc, Si, Sr, Th, Ti, V and Zn. These elements are usually described in terms of their total content and availability. In soil, trace elements retention is dependent on several soil characteristics as well as parent material. Soil characteristics include pH, cation exchange capacity (CEC), particle size distribution, organic matter content and oxide content. These characteristics cause trace elements to either accumulate in the soil or leave the soil for other components of the environment. When parent materials have high trace element concentrations, the soil that develop over them have high trace element concentrations (Evalyn, 2004).

Evalyn, (2004) concluded that “average concentrations of Si, K, Ba, Pb and U are highest in soils formed over felsic parent materials. Average concentrations of these elements

decrease when comparing felsic to intermediate to mafic derived soils. Conversely, Ti, V, Cr, Mn, Co, Ni, Cu and Zn average concentrations of these elements decrease when comparing mafic to intermediate to felsic derived soils. Concentrations of Cd and Hg are too close to the instrument defection limits to male any conclusion on concentration differences."

A related element analysis conducted by Kodom and his members was aimed to investigate the extent of heavy metal pollution in the surface soils at "Suame-Magazine" industrial area of Kumasi and to verify any significant industrial impact. According to Kodom (2008), concentrations of nine (9) heavy metals (Zn, Pb, Cr, Cu, Co, Ni, Cd, Hg and As) were qualitatively measured and examined from six (6) sampling sites, using X-ray fluorescence (XRF) analysis. Four (4) out of six (6) sampling sites were selected from areas of high industrial impact due to the chemical and metallurgical activities highly embarked on at the area whereas, the other two (2) were residential communities relatively of low industrial activity, selected in close proximity to the previous region. The results from ongoing locations of high industrial impact showed that the mean concentration of Zinc present in the soil ranges from 189.2-908.6mgkg^{-1}; lead values were in the range of 133.7-571.3mgkg^{-1}. Chromium concentration varies form 91.3-545.8 mgkg^{-1}, the concentration of copper ranged from 62.9-334.6mgkg^{-1}concentration values for cobalt, nickel, cadmium, mercury and arsenic were respectively in the range 38.6-81.9 mgkg^{-1}. Apart from nickel and arsenic, all the heavy metals recorded values extremely high above their respective thresholds limit values (TLV's). However, comparing these concentrations with those obtained from locations of low industrial activity; relatively very low heavy metals were recorded with chromium, cadmium, and mercury being the heavy metals

which were recorded in enhanced levels far above their TLV's. Considering all the heavy metals obtained at the study area, Zn, Pb and Cr dominated in all the investigated zones of high industrial impact. This is attributed to the indiscriminate disposal of industrial waste as well as anthropogenic point source contamination. The extremely high contents of the heavy metals generally recorded at the study area are terribly alarming in terms of environmental population. Therefore, the inhabitants (mostly children) the numerous workers, who reside and work at such polluted environment, are at serious risk of heavy metal toxicity and awareness needs to be created as such.

2.3.1.1 Silica Dust and Silicosis

Silicosis is a debilitating respiratory disease caused by inhalation of fine crystalline silica dust that is retained in the lungs (Ziskind, *et al*., 1976). The amount of dust inhaled, the percentage of free pr uncombined silica in the dust, the size of the dust particles and the length of exposure all affects the onset of silicosis. The inhaled dust deposited in the bronchioles and alveoli reacts within the lung tissue to form silicotic nodules. The nodules appear on chest radiograms as discrete, rounded opacities or shadows. The presence of silicotic nodules and a history of occupational exposure to silica dust are necessary for a positive diagnosis of silicosis. The earliest symptom of silicosis is shortness of breath. As the disease progresses, the silicotic nodules coalesce and form a continuous mass of fibrotic tissue called progressive massive fibrosis.

2.3.1.2 Mine Tailing

Mine waste consist of two types: mine tailing, which consist if fine-grained mineral materials, sludge's produced by ore treatment processes (i.e., concentration flotation and cyanidation) and slag generated by pyro-metallurgical operations (calcinations); and waste

rock, which consists of the host rock (gangue) that is discarded on the preparation phase (i.e. crushing and grinding) and generally contains no commercially valuable ore.

In gold mining, 99% of the rock materials removed from the ground consists of tailing and waste rock, normally stored on ponds that may cover several hectares of waste risk, piles (hills) that can reach as much as 100m in height (PHAC, 2008).

Arsenic is a common element in the natural environment and is frequently found in gold deposits (Vladimir & Robert, 1990). Arsenic sources to the biosphere associated with gold mining include waste soil and rocks, residual water from ore concentration, roasting of some types of gold-containing ores to remove sulfur and sulfur oxides and bacterially enhanced leaching (Eisler, 2004).

During mining a lot of chemicals including lead and mercury are exposed and with wrong mining technique applied by the artisanal miners enormous damage is done to the human body and the environment (CDC, 2002). Arsenic is an element which indicates the presence of gold in geological settings and when the gold is removed as mineral, arsenic is left behind (Vladimir & Robert, 1990).

2.3.1.3 Radon

When radon decays, it releases a "large" atomic alpha particle and the atom is transmuted into polonium, an alpha particle is essentially a helium atom stripped of its electrons. It is at this point that the real hazards associated with radon are encountered. For it is not the radon which is responsible for the health problems but rather the short lived radon daughters (SLRD's) and their decay produces such as the alpha particles. The radon may be thought of merely a source and a vehicle for the SLRD's (Connell, 2006). There are

several different kinds (isotopes) of naturally occurring radon. Of this radon- 222 which have a life of 3.8 days have radioactive properties that pose hazards to health.

Once the radon atom decays, it subsequently undergoes four rapid decays starting with polonium 218 which decays into lead 214, which decays into bismuth 214, which decays into polonium 214. Each of these daughters has a half-life of less than 30 minutes (the polonium 214 has a half-life of only 0.00016 seconds). Furthermore, the daughters are electrically charged. During each decay, at least one of the three types of ionizing radiation is emitted by the SLRD's: alpha, beta and gamma. The alpha particles are easily stopped by a single piece of paper or layer of clothing. Since the alpha particles are large and easily stopped, the alpha particles transfer virtually all of its energy to the material which has stopped the particle. Beta has less probability of being stopped and imparts less energy into the stopping material. The gamma radiation is similar to X-rays and has an even lower probability of being stopped.

Since the radon is airborne, these daughters have a high probability of being airborne. Inhaled progeny of radon are the most important source of irradiation of the human respiratory tract. Epidemiological studies of underground miners of uranium and other minerals have provided reasonably firm estimates of the risk of lung cancer associated with exposure to radon progeny (Yu, *et al.*, 2008). If daughters are inside the lung when they decay, the lining of the lung wall becomes the stopping material. Since the alveolar cells of the lung wall do not have a significant protective coating, an alpha particle can collide with the live cell, imparting an enormous amount of energy to the cell, possibly disrupting the DNA within the cell. This is the interaction which is thought to initiate the cancers associated with the SLRD's.

Conell, (2006) further asserted that ‘when a daughter is airborne, it has an electrical charge associated with and it has a higher probability to adhere to other airborne particulates and dust’. Radon-222 is one of the decay products in the radioactive series that originates with uranium. It contributes a significant portion of terrestrial radiation to a great percentage. On average two-thirds of the dose people receive from terrestrial sources comes from the gas, radon-222 (Bafandeh, 2004).

2.3.2 Contamination and Irradiation

There are two broad types of damage that internal radioactive materials may produce: first the radiation produced may be expected to exhibit the usual sort of either direct or indirect destructiveness. Second, many radioactive species are heavy metals and are therefore chemically toxic, even at extremely low concentration.

Piccard (2005) in his submission on radiation damage and dose measurement made assertions about graph of the rate of removal of a substance (in moles/sec) versus the concentration (in moles/l) of that substance within the relevant tissue (perhaps the circulating blood). Such a graph, according to him, will typically exhibit absorption passing linearly through the origin and in theory may exhibit at high concentration a leveling off, or saturation. On the other hand, if the concentration is low, the rate of removal will decline as the concentration is reduced, hence the graph will head downward at a decreasing slope, flattening out. The mathematics is exactly parallel to that of radioactive decay: during this final phase of removal, the concentration will decline exponentially with time (Piccard, 2005).Mercury contamination of the environment (water and soil) is another very important environmental issue.

2.3.3 Naturally Occurring Radionuclides Materials (NORMs)

The naturally occurring radioactive elements of interest as far as NORMs are concerned belong to the two natural decay series of U-238 and Th-232. The third natural decay series led by U-235 is usually not considered in radiological assessments for NORMs because it is less abundant than U-238.

When they are left undisturbed for a sufficient amount of time, the daughter radionuclides in each of the chains reach an equilibrium state with the parent such that the activity concentration of each member of the decay series is the same. This state is called secular equilibrium. Disturbances to this equilibrium can arise naturally but also through human activities such as those taking place in ore processes and they are due to the different physical and chemical properties of the element to which individual decay products are related to. It is these disturbances that are responsible for the radioactivity enhancement or depletion observed inmineral-ore processes. For example, Ra-226 in the U-238 decay series is soluble in water and chemically very different from uranium. By emitting alpha particles, it produces radon (Rn-222), an inert gas that does not react chemically but can escape via gaseous pathways. Lead (Pb) and polonium (Po) which are produced further down the U-238 chain are highly volatile and this is the reason why Pb-210 and Po-210 activity concentrations can be enhanced in compounds which are volatile at high temperatures, making it possible for them to escape by airborne routes. They can also subsequently be adsorbed onto respirable aerosols, thus depositing and contaminating local surfaces as well as lungs (ATSDR, 2007).

2.3.4 Exposure pathways

Individuals are exposed to radiation in different ways and these are commonly referred to as *Exposure pathways*. Relevant exposure pathways to be taken into account in mineral and ore processing operations are the following:

a) External exposure to gamma radiation

b) Internal exposure through dust inhalation

c) Internal exposure through ingestion

d) Skin contamination (from material deposited directly on the skin).

Occupational exposures occur when mine workers come into close and prolonged contact with NORMs or inhale dust generated during the process. The most common routes of exposure from the processes involving naturally occurring radionuclides tend to be external gamma radiation, for example from large quantities or piles of stored material or from residues deposited inside process equipment, and inhalation of dust.

Inhalation of re-suspended radioactivity can occur when someone breathes dust-loaded air with NORMs-containing particles that have been re-suspended into the air, for example when the material is transported, broken up or dumped at a landfill. Inhalation doses are estimated on the basis of dust concentrations, breathing rates and radionuclides concentrations in the fine dust fraction. Dose calculations related to inhalation of furnace fume and precipitator dust usually only take into account Pb-210 and Po-210 because of their volatile properties.

It is unlikely for NORMs residues and waste to be ingested in a direct way, although dust deposited on the skin can always be inadvertently ingested by mine workers or members of the public. However, general health and safety legislation and good practice requires that

mineral and ore processing activities has the potential to produce airborne dust, mine workers are required to wear personal protective equipment (overcoat, gloves and masks) to prevent or reduce the risk of contamination through this pathway.

Skin contamination is usually not considered to be relevant in dose assessments related to NORMs because they are low specific activity materials. Exposures of the public may arise from the product(s) of a process, from the atmospheric or liquid discharges, or from the disposal of solid waste(s). The most important routes of radiation exposure of the public are usually external gamma radiation, inhalation and ingestion.

2.4 Gamma Radiation

Gamma radiation also known as gamma rays are electromagnetic radiation of high frequency and therefore high energy with very short wavelength ($10^{-3}A^0$to 1 A^0) and therefore they have no electric charge and cannot be deflected by electric and magnetic fields (Satya, 2014).Gamma rays are ionizing radiation and are thus biologically hazardous. Gamma rays are produced from the decay from high energy states (highly unstable) of atomic nuclei. They can also be created in other processes.

Gamma rays are produced from naturally occurring radioactive isotopes and secondary radiations from atmospheric interactions with cosmic rays particles. Gamma rays are produced by number of astronomical process in which very high energy of electron are produced that in turn cause secondary gamma rays by the mechanism of Bremsstrahlung, inverse Compton scattering and Synchrotron radiation.

Gamma rays typically have frequencies above (10^9Hz) and therefore have energies above 100 KeV and wavelengths less than 10Pico-meters (less than the diameter of an atom). Gamma rays from radioactive decay are defined as gamma rays no matter what their

energy. Gamma decay commonly produces energies of a few hundred keV and almost less than 10 MeV.

2.4.1 Sources of gamma radiation

Natural sources of gamma rays on earth include gamma decay from naturally occurring radioisotopes such as potassium- 40. The high energy gamma ray produces secondary gamma rays by different process. A large fraction such as astronomical gamma rays is screened by earth atmosphere and must be detected by space craft. A notable artificial source of gamma rays includes fission which occurs in nuclear reactors and high energy physics experiments such as nuclear pion decay and nuclear fusion. Originally, the electromagnetic radiations emitted by X-ray tubes almost invariably have a longer wavelength than the gamma rays emitted by radioactive nuclei. X-ray and gamma rays can be distinguished on the basis of wavelength. With radiation shorter than some arbitrary wavelength such as 10^{-11}m defined as gamma rays. The classification of X-rays and gamma rays can be done on their origin. X-rays are emitted by electrons outside the nucleus, While gamma rays emitted by nucleus.

2.4.2 Interaction of gamma rays with matter

When a beam of gamma ray photon is incident on any material is removed individually in a single event. The event may be an actual absorption process, in which case, photon disappears or the photon may be scattered out of the beam. When a gamma rays passes through matter, probability for absorption is proportional to thickness of the layer, the density of the material, and absorption, cross section of the material. The total absorption

shows an exponential decrease of intensity with distance from the decrease of intensity with distance from the incident surface.

$$I(x) = I_0 e^{-\mu x} \quad (2.1)$$

Where,

$x$is the distance from the incident surface,

 is the absorption coefficient, measured in cm^{-1}

$n$is the number of atoms per $cm^3$of the material (atomic density),

$\sigma$is the absorption cross section in cm^2

Three processes are mainly responsible for absorption of γ rays. These are as follows

1. Photoelectric effect
2. Compton effects
3. Pair production

The predominant mode of interaction of gamma rays with matter depends on the energy of incident photons and the atomic number of the material with which they are interacting. At low energies and with high (Z) materials the photoelectric effect is main interaction process. At intermediate energies and in low (Z) materials, the Compton scattering is dominating. At very high energies, pair production is the most dominant interaction process.

2.4.2.1 Photoelectric effect

When the photon collides with an atom, it may impinge upon an orbital electron and transfer all of its energy to this ejecting it from the atom. When the incident electron transfer all of its energy to this electron by ejecting it from the atom. When the incident

photon energy (hv) exceeds the electronic binding energy (or ionization energy) E_B, the electron is ejected with a kinetic energy

$$E_{Kin} = hv - E_B \tag{2.2}$$

This phenomenon is known as the *photoelectric effect* and the equation is known as Einstein photoelectric equation, in the photoelectric process photon transfers all its energy to an electron (*photo electrons, ejected out of a material in the photo-effect*), which subsequently is removed from the atom (ionization).

The kinetic energy the electron receives equals the photon energy less than the binding energy of the struck electron. This process, in the course of which the photon disappears completely, takes place exclusively in the direct vicinity of the nucleus. Normally, as a precondition, the law of preservation of impulse plays a prominent role. The impulse the photon has due to its energy and velocity can, because the mass is too low, be transferred to an electron for a small part only. The rest of the impulse must therefore be transferred to the nucleus. So, the process can only take place with K or L-electrons and occur more often with substances with a high atomic number(Z). After all, the heavier the nucleus is, the more capable it is of taking over the surplus of impulse. However, when the photon energy is too high, a nucleus with a high Z cannot handle the surplus of impulse either, that is why the photoelectric effect only occurs up to a limited energy value. Once it is freed, the photo electron can ionize other atoms again along its route. The electrons freed from this, the so called secondary electrons, in their turn cause ionizations again along their routes. The formed electron gap in the struck atom is filled by an electron from a shell situated more on the outside.

2.4.2.2 Compton Effect

Characteristic of the Compton Effect is that only part of its total amount of energy is transferred from the entering photon to an electron. The freed electron, which is called Compton electron (recoil electron), reaches a certain velocity that is dependent on the energy transferred to the electron. The rest of the energy continues as a photon of lower energy in another direction, and is therefore called a scattered photon. Because of the lower energy the scattered photon has a longer wavelength than the original. The Compton process occurs only then when the photon energy passes the limiting value of the photoelectric process. Since the impulse and the energy are divided among the Compton electron and the scattered photon, the law of preservation of impulse is complied with, and the process occurs with the electrons from the outer shells as well. For this reason, the atomic number (Z) of the material is less influential. The freed Compton electrons can, depending on the energy content, ionize other atoms along their routes. The scattered photon continues its way and continues to enter into Compton processes up until the energy is reduced to such an extent that a photoelectric process takes place. Only then the photon has disappeared.

Because the electron binding energy is very small compared to the gamma ray energy, the kinetic energy of electron is nearly equals to the energy lost by the gamma

$$E_e = E_\gamma - E' \tag{2.3}$$

Where,

E_e is the energy of scattered electrons

E_γ is the energy of incident gamma ray

E' is the energy of scattered gamma ray

2.4.2.3 Coherent Scattering

In the case of Rayleigh scattering whole atoms works as the target. When the incident photon is scattered by the atom and changes its direction, the target atom recoils to conserve momentums before and after scattering. The recoil energy of the atom is very less and can be negligible because of the large atomic mass therefore; the photon changes its direction only and retains the same energy after scattering. As a result no energy is transferred. *Coherent scattering*, often called Rayleigh scattering, involves the scattering of a photon with no energy transfer (elastic scattering) (Gordon, 2008). The electron is oscillated by the electromagnetic wave from the photon. The electron, in turn, re-radiates the energy at the same frequency as the incident wave. The scattered photon has the same wavelength as the incident photon. The only effect is the scattering of the photon at a small angle. This scattering occurs in high atomic number materials and with low energy photons. This effect can only be detected in narrow beam geometry.

2.4.2.4 Pair Production and Annihilation

With photon energies larger than 1.022MeV pair production may occur as an alternative to the Compton process. When such a high energetic photon comes close to a nucleus, transformation of energy into mass can occur because of the electric field of the nucleus. With this the photon is converted into an electron and a positron with the same mass, but the reverse charge. If the photon energy is, for example, 2MeV, 2×0.511 =1.022MeV goes to the electron-positron pair and the remainder (0.978MeV) is divided as kinetic energy among the electron and the positron. In this process, in which the original photon disappears completely, the surplus of impulse is transferred to the nucleus. Summarizing, it can be posed that a photon, in comparison with a β-particle, loses a large part of its energy

in a long route of interaction, and eventually disappears completely. The penetrating ability of photons in matter is therefore a lot bigger than that of the β-particles.

2.5 Attenuation

When a beam of photon traversing through a slab of material can be absorbed or scattered through large angle. If we assume that the gamma ray is well collimated in a geometry both the scattering sign absorption cross-section (ss and sa) contribute to the loss in transmitted intensity Iwhich is given by

$$I = I_o e^{-N\sigma x} \qquad (2.4)$$

Where, $s = \sigma s + as$and the other symbols have their usual meaning. This equation can also be written as

$$I = I_0 \, e \, x \, p \, (- \Sigma x) = I_0 \text{e x p } (^{-x}/\lambda) \qquad (2.5)$$

Where summation = Nσ is called the macroscopic total cross section, and λ=1/summation is the mean attenuation length. For gamma rays theseequations only refers to mono energetic radiation that is collimated. The attenuation coefficient is a quantity that characterizes how easily a material or medium can be penetrated by beam of light, sound, particles or other energy or matter. A large attenuation coefficient means that beam is quickly attenuated as it passes through the medium, and a small attenuation coefficient means that the medium is relatively transparent to the beam. Attenuation coefficient is measured using units of reciprocal length. The attenuation coefficient is also called linear attenuation coefficient.

2.5.1 Linear Attenuation Coefficient

The linear attenuation coefficient describes the extent to which the intensity of an energy beam is reduced as it passes through a specific material. The linear attenuation coefficient

gives information about the effectiveness of a given material per unit thickness, in promoting photon interactions. The large value of attenuation coefficient is more likely to the given thickness of material. The magnitude of attenuation coefficient varies with thickness of material and its density, as we imply, with photon energy, while specific values of the attenuation coefficient will vary among materials for photons of specified energy. The plots of attenuation coefficient versus photon energy are similar for different materials. In general, trends shows high values of attenuation coefficient at low photon energies that decreases as photon energy increases goes through a rather minimum value, and then increases as energy continues to increase. The reason of these trends is that the linear attenuation coefficient is made up of three major components, each of which depends upon different types of photon interaction. At lower energy, a process is called photoelectric effect is the dominant interaction mode that has strong energy dependence, decreasing approximately as the inverse cube of the energy. At intermediate energies the dominant interaction is Compton scattering, which shows a decreasing trends with increasing energy. Finally, at higher energies the dominant interaction is pair production, this shows increasing nature as energy increases. This process is occurred in the energy 1.022MeV. Thus, at low energies photoelectric contribution decreases which causes in the attenuation coefficient as energy increases.

Linear attenuation coefficient $\mu(cm^{-1})$ is determined by using a well collimated narrow beam of photon passing through a homogeneous absorber of thickness 't', the ratio of intensity of emerging beam from the source along the incident direction, to the intensity is given by the Beer Lambert law

$$I = I_0 e^{-\mu x} \quad (2.6)$$

Where,

I_o - is the incident photon intensity,

I- is the transmitted photon intensity,

t- is the thickness of absorber.

The linear attenuation coefficient is used in the contest of X-ray or gamma rays where it is represented by symbol μ and measured in (cm^{-1}). It is used in acoustic for charactering particle size distribution. It is also used for modeling solar and infrared radioactive transfer in the atmosphere.

2.5.2 Mass Attenuation Coefficient

The ratio of linear attenuation coefficient (μ) to the density (ρ) is calledthe mass attenuation coefficient (μ/ρ) and has the dimension of area perunit mass (cm^3/gm).A narrow beam of mono-energetic photons with an incident intensity I_0, penetrating a layer of material with mass thickness t and density, emerges with intensity I given by the following relation,

$$\frac{I}{I_0} = e^{-(\mu/\rho)x} \quad (2.7)$$

$$\frac{\mu}{\rho} = x^{-1} \ln(I_o/I) \quad (2.8)$$

From which the mass attenuation coefficient can be obtained from the measured values of incident photon intensity(I_0), transmitted photonintensity (I) and thickness of the absorber (t). The thickness of the absorber is defined as the mass per unit area, and it is obtained by multiplying thickness t and density of the absorber, that is; $x = \rho t$.

2.5.3 Half-Value Layer (HVL)

The half-value thickness, or half-value layer, is the thickness of the material that reduces the intensity of the beam to half its original magnitude. When the attenuator thickness is equivalent to the HVL, N/N_0 is equal to ½. Thus, it can be shown that

$$HVL = \ln 2/\mu \tag{2.9}$$

This value is used clinically quite often in place of the linear attenuation coefficient.

The mean free path X_m is related to the HVL according to

$$X_m = {}^{HVL}/_{\ln 2} \tag{2.10}$$

2.5.4 Mean Free Path

The mean free path X_m, or relaxation length, is the quantity given as

$$X_m = {}^{1}/_{\mu} \tag{2.11}$$

This is the average distance a single particle travels through a given attenuating medium before interacting. It is also the depth to which a fraction 1/e (~37%) of a large homogeneous population of particles in abeam can penetrate. For example, a distance of three free mean paths,$3/\mu$, reduces the primary beam intensity to 5%.

The linear attenuation coefficient and mass attenuation coefficient are related such that the mass attenuation coefficient is simply m/r, where r is the density in g/cm. When this coefficient is used in the Beer-Lambert law, then "mass thickness" (defined as the mass per unit area) replaces the product of length time's density. The linear attenuation coefficient is also inversely related to mean free path. Moreover, it is very closely related to the absorption cross section.

2.6 Energy Absorption Coefficient

The effects which photons produce in matter are actually almost exclusively due to the secondary electrons. A photon produces primary ionization only when it removes an electron from an atom by a photoelectric collision or by a Compton collision, but from each primary ionizing collision the swift secondary electron, which is produced, may have nearly much kinetic energy as the primary photon. This secondary electron dissipated its energy mainly by producing ionization and excitation of the atoms and molecules in the absorber. For electrons of the order of 1 MeV, an average of about 1 per cent of the electrons energy is lost as Bremsstrahlung. If, on the average, the electron loses about32 MeVper ion pair produced, then a 1 MeV electron produces the order of30,000 ion pairs before being stopped in the absorber. The one primary ionization is thus completely negligible in comparison with the very large amount of secondary ionization.

2.6.1 Energy Absorption in Medium

By "energy absorption" we mean the photon energy, which is converted into kinetic energy of secondary electrons. This kinetic energy eventually is dissipated in the medium as heat and in principle could be measured with a calorimeter. The energy carried away from the primary collisions as degraded secondary photons in not absorbed energy. Suppose that a collimated beam containing n photons per (cm^2)(sec),each having energy hϑ (MeV), is incident on an absorber in which the linear attenuation coefficients are σ, τ, and k (cm^{-1}). The incident gamma ray intensity I of the beam is

$$I = nh\upsilon(Mev/(cm^2)(sec) \qquad (2.12)$$

In passing a distance dx into the absorber, the number of primary photons suffering collision will be

$$dn = n(\sigma + \tau + k)dx = n\mu_o dx\ photons/(cm^2)(sec) \quad (2.13)$$

The total energy thus removed from the collimated beam in hυdn MeV/(cm^3)(sec). But a significant portion of this energy will be in the form of secondary photons

In the Compton collisions, the average kinetic energy of the Compton electrons is hυ (σ_a/σ), and the Compton linear absorption coefficient is of the order of $^1/_2\,x$for 1 to 2 MeV photons. In the photoelectric collisions, the energy of photoelectron is $(h\upsilon - B_e)$, where $B_e$is the average binding energy of the atomic electron. In the pair-production collisions the total kinetic energy of the positron-negatron pair is (hυ–$2m_o c^2$). Combiningthese considerations, we find that the true energy absorption in a thickness of dx is

$$dI = n\left[\sigma h\upsilon\, {}^{\sigma_a}/_{\sigma} + \tau(h\upsilon - B_e) + k(h\upsilon - 2m_o c^2)\right]dx\ Mev/(cm^2)(sec) \quad (2.14)$$

For the light elements B_e and$2m_o c^2$ are usually neglected. Then the usual, but approximate expression for energy absorption becomes

$$dI = I(\sigma_a + \tau + k)dx = I\mu_a dx\ Mev/(cm^2)(\text{sec}) \quad (2.15)$$

Where,$\mu_a = (\sigma_a\tau + k)$ is the linear absorption coefficient and that μ_a is smaller than the total attenuation coefficient μ_0, because μ_0 includes as scattering coefficient μ, which represents the energy content of all the secondary photons (Compton, X-rays, and annihilation radiation). Then, rigorously, and in the usual approximation, neglecting B_e and $2m_o c^2$

$$\begin{gathered}\mu_a = (\sigma_a + \tau + k)cm^{-1} \\ \mu_s = \sigma_s\ (cm^{-1})\end{gathered} \quad (2.16)$$

A simple and very general result, which follows at once from equation,

$$Z_{eff} = \frac{\sigma_{tot}}{\sigma_{ele}} \tag{2.17}$$

is that the rate of energy absorption per unit volume is simply the incident intensity times μ_a

$$\frac{dI}{dx} = I \mu_a MeV/cm^2)(sec) \tag{2.18}$$

This is valid for any size and shape of volume element, throughout which the intensity I is essentially constant.

2.7 Radioactivity Concepts

2.7.1 Radioactivity

Radioactivity is defined in accordance to the radioactive decay law. According to this law: The rate of disintegration of a radioactive atom $N(t)$ is proportional to the original number of atoms $N(0)$

$$\frac{dN(t)}{dt} \alpha\ N(0) \tag{2.19}$$

Where:

$N(t)$= Number of radioactive atom remaining at time t

$N(0)$= Original number of nuclei.

In radioactivity, the nucleus of an unstable atom is spontaneously transformed into a more stable one accomplished by different mechanisms, some of which include: alpha (α) particle emission, beta negative (β^-) (or negatron) and positive (β^+) emission and orbital electron capture. Each of these mechanisms may or may not be accompanied by gamma radiation as the spontaneous nuclear transformation of an unstable nucleus into more stable

ones takes place. The two factors that determine the exact mode of radioactive transformation are:

1. The mass-energy relationship among the parent nucleus, daughter nucleus and the emitted particle.
2. The particular type of nuclear instability that is whether the neutron to proton ratio is either too high or too low for the type of nuclide consideration.

The substances whose nuclei undergo these transformations are to be radioactive and the transformation process is known as radioactive decay. The elements of radiation mentioned above as alpha particle, beta particle, and gamma rays carry energy along, which they deposit on matter they impinge. Alpha apart from depositing their energies, they also ionize the different media they interact with either directly or indirectly. The amount of energy absorbed (or deposited) per unit mass on any matter they impinge is called the absorbed dose measured in gray (Gy) or Joule per Kilogram.

2.7.2 Serial Radioactive Decay and Decay Equilibrium.

A serial radioactive decay is produced when one or more radionuclides are produced in a chain(Turner, 1995), suppose that a parent radionuclide A, having half-life T_a and decay constant $\lambda_{a,}$ decays to a daughter radionuclide B of half-life T_a and constant $\lambda_{b,}$ which in turn decays to a radionuclide C as illustrated below:

$$A^{\underline{\lambda a,Ta}}\ B\ ^{\underline{\lambda b,Ta}}\ C \qquad (2.20)$$

The rate of change,$\frac{dN_b}{dt}$in the number of daughter atoms,N_b,per unit time equals the rate $\lambda_a N_a$ at which they are produced minus their decay rate. By applying radioactive decay equation, we then have the equation showing the number of daughter atoms as follows:

$$N_b = \frac{\lambda_a N_{o,a}}{\lambda_b - \lambda_a}(e^{-\lambda_a t} - e^{-\lambda_b t}) \quad (2.21)$$

In a particular case where the parent, A is long-lived and its daughter, B decays at a much faster rate such that $T_a \gg T_b$, a point is soon reached at which the instantaneous amount of A, that decays is equal to that of B. Under these conditions B is said to be in secular equilibrium with its parent A.

As time t, increases beyond seven daughter half-lives (for practical purposes) when $\lambda_a \ll \lambda_b$, when $T_a \gg T_b$, the below equation is arrived at.

$$\lambda_b N_b = \lambda_a N_{0a} \quad \text{or} \quad \frac{\lambda_a}{\lambda_b} = \frac{N_b}{N_a} \quad (2.22)$$

From the above equations, it is evident that at secular equilibrium, the activity of the daughter equals that the parent, and that the ratio of the decay constant of the parent and daughter are in the inverse ratio of the equilibrium concentrations of the parent and daughter.

From the above it is possible to deduce from calculation the activity of particular long-lived parent like ^{232}Th by using the activity of its short-lived daughter ^{208}Tl when in secular equilibrium with it.

When the half-life, T_a of a parent is greater than that of the daughter, T_b but not greatly so (i.e., $T_a > T_b$ and not $T_a \gg T_b$), the activity of the daughter $\lambda_b N_b$ builds up steadily initially as the time, t increases, $e^{-\lambda bt}$ eventually becomes negligible with respect to $e^{-\lambda at}$, since $\lambda_b > \lambda_a$. Knowing that the product of the number (N) of atoms in a substance and its decay constant is its activity, he then obtained

$$A_b = \frac{\lambda_b A_a}{\lambda_b - \lambda_a} \quad (2.23)$$

Where A_a and A_b are respective activities of A and B measured in Bq/kg.

Any initial rise in the daughter activity, a corresponding fall in the activity of the parent activity is obtained at the same time. The form of equilibrium occurs after a maximum is attained in the activity of the daughter which clearly depicts another type of equilibrium which is the transient equilibrium. Further also, a situation can arise where no equilibrium occurs, principally because the activity of the daughter builds up and falls, i.e. $T_a < T_a$ indicating that the parent have a shorter half-life, it eventually decays away and only the daughter is left.

2.7.3 Radioactivity Terms

2.7.3.1 Activity

The quality that characterizes the speed of radioactive decay is called activity (or emitter activity). The activity A is defined as the number of decays per unit time interval, or the loss in the number of undecayed nuclei per unit time interval:

$$A_t = -\frac{dN_t}{dt} = \lambda N_t \qquad (2.24)$$

The unit of activity is the Becquerel (Bq), one of disintegration per second. The older unit of activity was the curie ($1Ci = 3.7 \times 10^{10} Bq$).

2.7.3.2 Specific Activity

The concentration of radioactivity or the relationship between the mass of radioactive material and the activity is called the specific activity. Specific activity is the number of Becquerel (or curies) per unit mass or volume.

$$SA = \frac{\lambda N}{Nm/N_a} = \frac{\lambda N_a}{M} \quad (2.25)$$

2.7.3.3 Half Life

A clearer quantity used to characterize the radionuclide instead of the decay constant λ is half-life. The half-life $T_{1/2}$ is the time it takes for the number of atoms to decrease to half the initial value. Thus the half-life is related o the decay constant through the equation

$$T = \frac{0.693}{\lambda} \quad (2.26)$$

Half – lives vary greatly among different types of atoms, from less than a second to billions of years. For example, it will take about 4.5 billion years for half of the atoms in a mass of ^{238}U to spontaneously disintegrate, but only 24,000 years for half of the atoms in a mass of ^{239}Pu to spontaneously disintegrate. In contrast ^{131}I, commonly used in medicine, has a half – life of only 8 days.

2.7.3.4 Average Life

The average life is defined simply as the sum of the lifetime of the individual atoms divided by the total number of atoms originally present.

$$\tau = \frac{1}{\lambda} \quad (2.27)$$

If the expression for the transformation constant in terms of the half life and mean life is to be: $\tau = 1.44T_{1/2}$ (2.28)

2.8 Assessment of Radiological Hazard

2.8.1 Absorbed Dose Rate in Air (D)

In order to assess any radiological hazard, the exposure to radiation arising from radionuclide present in soil can be determined in terms of many parameters. A direct connection between radioactivity concentrations of natural radionuclide and their exposure is known as the absorbed dose rate in the air at 1 meter above the ground surface. The mean activity concentrations of ^{226}Ra (^{238}U), ^{232}Th and ^{40}K (Bqkg^{-1}) in the soil samples are used to calculate the absorbed dose rate given by the following formula

$$D\ (nGy/h) = 0.462A_{Ra} + 0.604A_{Th} + 0.0417A_k \quad (2.31)$$

Where D is the absorbed dose rate in nGyh^{-1},A_{Ra}, A_{Th} and A_k are the activity concentration of ^{226}Ra (^{238}U), ^{232}Th and ^{40}K, respectively. The dose coefficients in units of nGyh^{-1}per Bq.kg^{-1} were taken from the UNSCEAR (2000).

2.8.2 Annual Effective Dose Equivalent (AEDE)

The absorbed dose rate in air at 1 meter above ground surface does not directly provide the radiological risk to which an individual is exposed (Jibiril, 2009; Jibiri & Biere, 2011). The absorbed dose can be considered in terms of the annual effective dose equivalent from outdoor terrestrial gamma radiation which is converted from the absorbed dose by taking into account two factors, namely the conversion coefficient from absorbed dose in air to effective dose and the outdoor occupation factor. The annual effective dose equivalent can be estimated using the following formula (Beliverimis, *et al.*, 2009; UNSCEAR, 2000):

$$AEDE\ (\mu Sv/y) = D(nGy/h) \times 8760h \times 0.7Sv/Gy \times 10^{-3} \quad (2.32)$$

The values of those parameters used in the UNSCEAR report (2000) are 0.7 $Sv.Gy^{-1}$ for the conversion coefficient from absorbed dose in air to effective dose received by adults and 0.2 for the outdoor occupancy factor (UNSCEAR, 2000).

2.8.3 Radium Equivalent Activity (Ra_{eq})

Due to a non- uniform distribution of natural radionuclide in the soil samples, the actual activity level of ^{226}Ra, ^{226}Th and ^{40}K in the sample can be evaluated by means of a common radiological index named the radium equivalent activity (Ra_{eq}) (Bertka & Matthew, 1995). It is the widely used index to assess the radiation hazards and be calculated using Equation 2.21 given by. This estimates that 370 $Bq.kg^{-1}$ of ^{226}Ra, 259 $Bq.kg^{-1}$ of ^{232}Th and 4810 $Bq.kg^{-1}$ of ^{40}K produce the same gamma-ray dose rate.

$$Ra_{eq} = A_{Ra} + 1.43_{Th} + 0.077A_k \quad (2.33)$$

Where A_{Ra}, A_{Th} and A_k are the activity concentration of ^{226}Ra, ^{232}Th and ^{40}K in $Bq.kg^{-1}$, respectively. The permissible maximum value of the radium equivalent activity is 370 $Bq.kg^{-1}$ (UNSCEAR, 2000) which corresponds to an effective dose of 1 mSv for the general public (Ajayi, *et al.,* 2009).

2.8.4 External Hazard Index (H_{ex})

To limit the radiation exposure attributable to natural radionuclide in the sample to the permissible dose equivalent limit of $mSv.y^{-1}$, the external hazard index base on a criterion have been introduced using a Kreiger model which is given by (Nadal, *et al.*, 2004)

$$H_{ex} = (A_{Ra}) + (A_{Th}) + (A_k) \quad (2.34)$$

In order to keep the radiation hazard insignificant, the value of external hazard index must not exceed the limit of unity. The maximum value of H_{ex} equal to unity corresponds to the upper limit of radium equivalent activity 370 $Bq.kg^{-1}$(Nadal, *et al.*, 2004).

2.8.5 Gamma Index (I)

The representative level of gamma index is (I) was calculated using the following equation (UNSCEAR, 2000):

$$I = 0.0033A_{Ra} + 0.005A_{Th} + 0.0005A_k \quad (2.35)$$

According to UNSCEAR (2000), for radiological effects to be considered negligible, the values of each of I must be less than 1.

2.8.6 Excess Lifetime Cancer Risk (ELCR)

When assessing the impact of radiation exposure it is a common practice to present the final conclusions in terms of excess lifetime cancer risk in a population exposed to a given dose. To propose a simple approximation to the excess lifetime risk, the validity of the approximation for levels of exposure is justified empirically as well as theoretically. This approximation provides important guidance in understanding the influence of the various

factors involved in risk projections. The Excess Lifetime Cancer Risk (ELCR) will be calculated using the following equation (Taskin, *et al*., 2009);

$$ELCR=AEDE \times DL \times RF \quad (2.36)$$

Where, AEDE is the Annual Equivalent Dose Equivalent, DL is the average duration of life (estimated to 70 years), and RF is the Risk Factor (Sv^{-1}),i.e. fatal cancer risk per Sievert. For stochastic effects, ICRP uses RF as 0.05 for public (Taskin, *et al*., 2009).

2.9 Assessment of the Level of Contamination

2.9.1 The Pollution Load Index (PLI)

The pollution load Index (PLI) is aimed at providing a measure of the degree of overall contamination. The PLI of the place is calculated by obtaining the n-root from the nCFs that was obtained for all metals. Generally pollution load Index (PLI) is given as follows:

$$PLI = \sqrt[n]{CF_1 \times CF_2 \times \ldots\ldots CF_n} \quad (2.37)$$

Where,

$CF =$ Contamination factor

$n =$Number of metals studied

The PLI value of > 1 is polluted, whereas < 1 indicates no pollution (Harikumar, *et al*., 2009). This measurements has however been defined by some authors in several ways. In this study, the author found it appropriate to express the PLI as the geometric mean of the studied pollutants since this methods tends to reduce the outliers, which might bias the reported results.

2.9.2 Contamination Factor (CF)

The level of contamination of soil by metals is expressed in terms of the contamination factor (CF). It is calculated as

$$CF = \frac{C_m\, sample}{(C_m\, Background\ )} \tag{2.38}$$

Where $C_m\ sample$ is the concentration of the sample

$C_m\ Background$ is the sample along the control area

If $CF < 1$: indicates moderate contamination

$1 < CF < 3$: indicates moderate contamination

$3 < CF < 6$: indicates considerable contamination

$CF > 6$: indicates very high contamination

2.9.3 Degree of Contamination (DC)

The degree of contamination (DC) of a determined area is the sum of all Contamination factors:

$$\mathrm{DC} = \sum \mathrm{CF} \tag{2.39}$$

Table 2.1 Classification of degree of contamination.

Degree of Contamination	Classification
DC<1	Low
1≤DC<3	Moderate
3≤DC<6	Considerable
DC≥6	Very high

Source: (Hakanson, 1980)

2.9.4 Geo-accumulation Index (I_{geo})

This method assesses the metal pollution in terms of seven (0 to 6) enrichment classes, ranging from background concentration to very heavily polluted (Muller, 1969), as follows

$$I_{geo} = \log_2 \left[\frac{C_m\, sample}{1.5 \times (C_m\, Background\ )} \right] \quad (2.40)$$

The factor 1.5 is introduced in the equation to minimize the effects of possible variation in the background values. The classification of the index is tabulated below.

Table 2.2 Classification of Geo-accumulation index

I_{geo} Value	I_{geo} Classification	Level of Concentration
$\mathbf{5 < I_{geo} \leq 10}$	6	Extremely contaminated
$\mathbf{4 < I_{geo} \leq 5}$	5	Strongly to extremely contaminated
$\mathbf{3 < I_{geo} \leq 4}$	4	Strongly contaminated
$\mathbf{2 < I_{geo} \leq 3}$	3	Moderately to strongly contaminated
$\mathbf{1 < I_{geo} \leq 2}$	2	Moderately contaminated
$\mathbf{0 < I_{geo} \leq 1}$	1	Uncontaminated to moderately contaminated
$\mathbf{I_{geo} \leq 0}$	0	Uncontaminated

2.9.5 Ecological Risk Factor (ERF)

Ecological risk Factor (ERF) is critical to measure both risk factor and metals concentrations in soil. The potential ecological risk index can be determined through the following formula:

$$ERF = CF * TRF \quad (2.41)$$

Where ERF is the potential ecological risk factor, TRF represents the toxic-response factor, and CF represents contamination factor. The toxic response values for some of the heavy toxic elements are As=10, Cr=2, Cd=30, Cu=5, Pb=5, Ni=5, and Zn=1 as suggested by (Hakanson, 1980). This is calculated separately for each metal. The results are as follows:

Table 2.3 Classification of Potential Ecological Risk factor

Potential Ecological Risk	Classification
PER < 40	Low
40 ≤ PER < 80	Moderate
80 ≤ PER ≤ 160	Considerable
160 ≤ PER ≤ 320	High
PER ≥ 320	Very high

Source: (Hakanson, 1980)

2.9.6 Hazard Quotient (HQ)

The soil Hazard Quotient (HQ) is the ratio of the heavy metal concentration of surveyed soil samples to reference permissible limit and is computed using the relation;

$$HQ=Cc/Cp \quad (2.42)$$

Where Cp and Cc = reference maximum permissible limit of metal concentration and the concentration obtained in the sampled area respectively. The maximum allowed limits were set by W.H.O and the Romanian guideline and were tabulated in Tables 2.4 and 2.5.

Table 2.4 Maximum allowed concentration (ppm) limit of some toxic metals in soil

Toxic Metal	WHO Maximum Allowed limits (ppm)
Nickel(Ni)	80.00
Copper(Cu)	30.00
Cadmium(Cd)	3.00
Chromium(Cr)	100.00
Lead(Pb)	100.00
Zinc(Zn)	300.00

Source: (WHO, 1996)

Table 2.5 Romanian guidelines on toxic metals level permitted in soil for pollution assessment

Element	Literature Values		Romanian norms				
	European median in soil	World median	NV*	ALS*	ALLS*	ITS*	ITLS*
As	7.03	6	5	15	25	25	50
Cd	0.145	0.35	1	3	5	5	10
Cr	60	70	30	100	300	300	600
Cu	13	30	20	100	250	200	500
Ni	18	50	20	75	200	150	500
Pb	22.6	35	20	50	250	100	1000
Zn	52	90	100	300	700	600	1500

*: NV=normal value; ALS and ITS=Alert level and Intervention threshold in the sensitive area; ALLS and ITLS=Alert level and Intervention threshold in the less sensitive area (Romania, 1997).

2.10 Factors Affecting Toxicity

Numerous factors influence the action of toxic substances, the dose physical and chemical nature of the substance, the critical organ, whether exposure is single or repeated and age (Schmidt, 2009)

1. **Dose:** Whether a specific substance shall have a harmful effect is determined in many cases by the amount of the substance absorbed. If the absorption reaches the toxic dose, the toxic effects become apparent (toxic dose). At even higher dose their effects may be fatal (lethal dose). The toxicity of a substance is usually measured in terms of the mean lethal dose (LD_{50}), which is the dose which will kill 50% of concerned population. Absorption of toxic substances in the gastrointestinal track depends on age and nutritional factors (Charke & Clarke, 1967). The absorption is much higher (50% against 10%) in children than for adults. For

instance most cases among children occur in summer, perhaps because of enhanced intestinal absorption of lead from the action of vitamin D. Exposure to sunlight increases the body's production of this vitamins (Grolier Incorporated, 1985). Fatty and oily food also enhances the absorption of heavy metals.

2. **Physical and chemical nature**: The physical state that is whether solid, coarsely or in powdered form or in solution will affect the toxic dose, absorption of heavy meals from the coarsely crystalline structure is relatively slow and therefore has a relatively low toxicity. Finely divided particles on the other hand are highly dangerous especially the nanoparticles that are very small enough to enter into cells to trigger inflammatory responses and oxidative stress (Schmidt, 2009).

If in solution, as in lotions and eye drops, the nature of the solvent is of considerable importance since many substances are absorbed more readily from oily than from aqueous solution. The solvent is of particular importance when considering percutaneous absorption. There are four recognized routes if entry of toxic substances into the body; inhalation, ingestion, skin absorption and skin puncture (Health Canada, 2009). The chemical nature of a substance usually affects its action; compound containing trivalent arsenic are much more toxic than those containing the pentavalents. Hexavalent chromium has been recognized as a cause of lung cancer while the trivalent is not (Lilisa, *et al.*, 2000).

The rate at which a toxic substance compound is eliminated depends on the chemical state of the substance: the radical to which the toxic element is attached determines its elimination rate which is quantified by its biological half life (T_b). $T_b$is defined as the

time required for the body or an organ to eliminate one half of the absorbed amount of any substance.

Table 2.6 Biological half life of some compounds

Biological half life	**Class D (<10 days)**	**Class W 10-100 days**	**Class Y >100 days**
Examples	Phosphates Carbonates Sulphates	Oxides Halides Nitrates	Oxides Hydroxides

3. **The critical organ:** For a given toxic element, the organ or tissue whose damage by the toxic element result in the greatest damage to the body is said to be the critical organ for that element (Hostyneck, 1998). That is toxicity depends on the organ concentrating a given element. Therefore, T_b is both organ and element specific.Table 2.9 showsT_b values of the some elements of interest and their respective organs.

Table 2.7 Biological half life and critical organ(s) of some elements

Element	Biological half life (T_b)	Critical Organ
Arsenic	200 hours	Blood
Cadmium	30 years	Bone
Cobalt	9.5 days	Whole body
Chromium	616 days	Whole body
Lead	32 years	Bone
	7 years	Kidney
	2 months	Blood
Mercury	20 years	CNS
Nickel	3.5 years	Nasal mucosa

T_b is related to the elimination constant k by

$$T_b = \frac{\log_e 2}{k} \tag{2.43}$$

Elimination is accomplished through one or more of the routes of exit; exhalation sweat, faeces and urine (Hall & Giacca, 2006).

2.11 Review of Related Works

(Augustine, *et al.,* 2014), assessed the natural radioactivity and determine heavy metals in soil around industrial dumpsites in Sango-Ota, Ogun State, Nigeria; his result shows mean activity concentration of ^{226}Ra, ^{232}Th and ^{40}K to be 30.0 $\pm$ 1.2, 33.3 $\pm$ 9.8 and 122.1 $\pm$ 20.6 Bqkg^{-1}, respectively. The mean concentration of heavy metals in the soil samples were 33.6, 2.9, 3.8, 2.7, 48.9, 1, 5, 34.5 and 0.8mgl^{-1} for Cu, Mg, Ca, P, Fe, Pb, Zn and Cl

respectively. The concentrations of Cd, Cu, and Pb were higher than the permissible range in soil.

(Attia, *et al*., 2014) assessed natural and artificial radioactivity levels and radiation hazards and their relation to heavy metals in the industrial area of Port Said City, Egypt. The concentration levels ranged from 0.2 to 21ppm for U – 238 and 0.01 for k – 40, whereas the total count values ranged from 8.7 to 123.6ppm. The dose rate values range from 0.12 to 1.61mSv/year. The activity concentrations of Ra-226, Th-232 and K-40 in the measured samples ranged from 18.03 to 398.66Bqkg^{-1}, 5.28 to 75.7Bqkg^{-1} and 323.88 to 583.12Bqkg^{-1} respectively.

(Olayiwole, 2013) analyzed samples of soil and vegetation for their heavy metal contents using standard methods. He collected samples from around a steel rolling mill along Ikirun, Osogbo road, Osun State. His results suggested that the area was grossly contaminated mostly with iron, chromium, lead and nickel in the soil.

Iyaka & Kakulu, (2012) studied heavy metal concentrations in top agricultural soils around ceramic and pharmaceutical industrial sites in Niger State, Nigeria and determined the presence of leas, copper, nickel and zinc using flame atomic absorption spectrophotometry technique. Mean contents for all sample locations of the two industrial sites were 18 ± 7.5, 15 ± 5.7, 1.9 ± 0.96 and 28 ± 22mg/kg for lead, copper, nickel and zinc respectively. The obtained values were of higher contents than the background levels accounted for by anthropogenic activities.

Akpan & William, (2014) assessed the elemental concentrations of roadside soils in relation to traffic density in Calabar, Nigeria. Twenty one elements viz; Al, Si, P, Cl, K, Ca, Ti, V, Cr, Mn, Fe, Ni, Cu, Zn, Ga, As, Rb, Sr, Zr, Nb and Pb with concentrations (in

ppm) ranging from 9.0 ± 3.8 (Ga) to 192560.3 ± 789.5 (Si) respectively were detected from two different sample locations. Significant enrichment were obtained for Mn (7.677); Cu (5.789); Zn (5.203, 5.177 and 6.554); Ti (5.723, 5.395 and 5.00); Cr (6.901, 7.323 and 14.321); P (5.683, 5.750) and Si (6.747) respectively indicating that their concentration is sufficient to pose environmental problems.

Kowsar, *et al*., (2012) carried out research to assess the radiological and chemical threat caused due to urban and industrial wastage drainage to the inhabitants of the Chittagong city in Bangladesh. The chemical effect along with the measurement of radiological threat and the physicochemical characteristics of the sediment samples were analyzed. The activity concentrations of naturally occurring radionuclide are ^{238}U, ^{232}Th, ^{226}Ra and ^{40}K in the investigated samples has been found to be higher than those of the world average values. The artificial radionuclides ^{137}Cs in the samples studied was not detected. The mean was found to be 240.94 ± 23.12 $Bqkg^{-1}$ and 115.82 ± 10.81$nGyh^{-1}$ respectively. The radiation doses were measured directly by employing beta-gamma survey meter from where the samples have been collected. The average values of the radiological parameters were calculated from the activity concentrations of the radionuclide mentioned in the sediment samples found to be higher than those of the corresponding world average values.

Faweya & Oniya, (2012) conducted a radiological safety assessment and physico-chemical characterization of soil mixed tailings used as building material from oke-kusa mining sites in Ijora, Nigeria. His results showed that mica site has the highest radioactivity level of 961 ± 15Bqkg-1 (kaolin site); 14 ± 4$Bqkg^{-1}$ (columbite-tantalite site) for ^{40}K ^{226}Ra and ^{232}Th respectively. The average outdoor and indoor effective dose rates were far less than 1$mSvyr^{-1}$ recommended limit for the members of public.

Stihi, *et al*., (2011) determined the heavy metal content of the fruiting bodies of Iycoperdonperlation and pleurotusostreatus and their substrate collected at various distances form a metal smelter in dambovita country, Romania. The concentration of Mn, Fe, Gu and Zn in the samples were determined by energy dispersive X-ray fluorescence (EDXRF) spectrometry and the concentrations of Cr, Ni, Se, Cd and Pb were determined by atomic absorption spectrometry. The coefficient of accumulation of heavy metal was calculated and the highest accumulation of Fe, Cu and Zn from substrate was observed for all the analyzed mushrooms samples. Furthermore, a high accumulation of Pb was observed in mushrooms growing in the vicinity of the metal smelter.

Montes, *et al.*, (2010) determined the depth of profile concentration of both naturally and anthropogenic gamma-ray emitter nuclide in soil samples collected in an area located at 34^{o} 54.425^{1}, 58^{o} 8.365^{1} W, down to 50 cm in depth, using a HpGe spectrometry .The soil samples were also characterized by means of massbauer spectrometry and X-ray diffraction. The activities of ^{238}U and ^{232} the natural chains remain constant in depth at 41Bq/kg between 2.5cm and 25.5cm of depth. The only anthropogenic detected nuclide is ^{137}Cs, whose activity changes from 1.4Bq/kg to values lower than the detection limit (LD) from depths below 25cm exhibiting a maximum at beneath the surface. The mossbauer spectral show two magnetic sextets associated with α-Fe_2O_3 and Fe_3O_3, as well as two Fe^3 and Fe^2 doublets probably originated in octahedral and tetrahedral site of paramagnetic places. The Fe^{3+} paramagnetic signal relative fraction increases up to 82% at the expense of the α-Fe_2O_3, one when the depth increases. No correlation between Fe_3O_4 and the ^{137}Cs was identified.

Diabi, *et al.,* (2008) determined the radioactivity concentration of soil samples collected from cultivated area, where phosphate fertilizers produced by Abu-zabal factory using high purity germanium detectors (HpGe). Specified activity levels die to ^{226}Ra, ^{232}Th and ^{40}K were measured in 23 fertilized soils samples at different depth from zero to 25cm with a step of 5cm depth. The average activity concentration (Bq/kg) in the collection of soil samples were found in the range from 6.0 ± 1.2 to 87.12 ± 2.22Bq/kg with an average value of 264.1 ± 11.04Bq/kg. The absorbed dose in air at the cultivated term was found to be 31nGy/h within the range (9.7-57nGy/h). the radium equivalent activity (Ra_{eq}), the external hazard index (H_{ex}) and the annual dose equivalent were also calculated and compared with the international recommended values and found to be within the international level.

Ramasamy, *et al.,* (2009) evaluated the radionuclides (U-235, Th-232 and K-40) content and outdoor terrestrial gamma dose rates in the Ponnaiyar river sediments. 40 samples were collected from which 40 locations were selected. All sediments samples were collected at 0-10cm depth. The collected samples were dried at room temperature in open air for two days and stored in black polythene bags. The gamma ray spectrometer with Na1(TI) detector was used to determine the concentration of the primordial radionuclides. The mean activity concentration for ^{238}U, ^{232}Th and ^{40}K, are BDL-11.60 ± 9.20Bq/kg with an average 46.85 ± 5.25Bq/kg and 201.23 ± 19.90, 467.71 ± 34.34Bq/kg with an average 384.03 ± 26.82Bq/kg respectively. The determined activity concentration of all sites falls within the typical world Indian average values although some extreme values were determined. The radiological hazard indices were below the internationally recommended values.

Faweya & Babalola, (2010) conducted a radiological assessment and occurrence of heavy metals in soils from designated waste dumpsites used for building and composting in southwestern Nigeria. Twelve major sites were identified from the highly industrial cities in the region, soil samples were collected to a depth of 20cm using a coring tool that was thoroughly cleaned and dried before each same was collected. Gamma spectrometry measurement was carried out with coaxial-type Ge detector of 50% relative efficiency and resolution of 2.4kev at 1.33Mev, the activity concentration of radionuclides were 1.46Mev for ^{40}K, 0.19Mev for ^{226}Ra, 0.09Mev for ^{210}Pb, 0.16Mev for ^{214}Bi, 0.09Mev for ^{109}Cd and 0.58Mev for ^{208}Ti respectively. Eight radionuclides ^{40}K, ^{226}Ra, ^{228}Th, ^{137}Cs, ^{210}Pb, ^{214}Bi, ^{109}Cd and ^{208}Ti was detected and measured in all the samples. The maximum concentration of ^{40}K, ^{228}Th, ^{137}Cs were found to be 357 ± 12, 68±7, 132 ± 10, and 0.96Bq/kg^{-1} at Ado, Abeokuta, Lagos and Ibadan while the minimum concentration are 180 ± 6, 40 ± 5, 22 ± 2 and 0.19Bq/kg^{-1} at Akure, Ado, orogbo and Abeokuta respectively. The maximum concentration of heavy metals ^{208}Ti, ^{210}Pb, ^{214}Bi and ^{109}Cd are 35 ± 1, 46 ± 5, 104 ± 1 and 49 ± 1Bq/kg^{-1} at Lagos, Ibadan and ado while the minimum concentrations are 5 ± 0.24, 20 ± 3, 17 ± 2 and 40 ± Bq/kg^{-1} at Ado and Ibadan respectively. The highest and the lowest average outdoor effective dose rates were recorded at Lagos and Ibadan.

Fuad, (2008) determined the activity concentration of soil samples collected from four different sites of state of Qatar. The soil samples were analyzed using high-resolution gamma-ray spectrometry. From the measured gamma-ray spectra, the activity concentrations were determined for two series (^{238}U and ^{232}Th) and two non-series (^{40}K and ^{137}Cs) radionuclides. In addition to the activity concentration of two samples with high concentrations if ^{238}U and ^{232}Th which are uranium-oxide ($U0_2$) and monazite (mineral

sand rich (^{232}Th) were also measured. The determination of the specific activity of ^{238}U was found from the decay progeny ^{214}Bi (608.90kev and 1120.3kev) to be 7.5 ± 0.7 MBq/kg and 8.5 ± 1.4MBq/kg respectively whereas the activity of ^{232}Th from the gamma-ray emission lines ^{228}Ac 910.81 kev was measured to be 474 ± 49kBq/kg. The activity concentration observed in this study is higher than presented for the same soil in other studies. The obtained results confirm that one of the samples (soil-228) is more radioactive than the other soil samples. One concern arising from this study was the observation of significant gamma-ray self-attenuation in the samples which causes an apparent reduction in the measured activity concentration by up to a factor of two low energies gamma-ray photons.

Psichoudaki & Papethymiou, (2008) also carried out studies to determine specific activities of natural radionuclide ^{238}U, ^{232}Th, and ^{40}K in the surface soil samples collected from the city of ptolemias, which is located near ignite-power plant using gamma-ray spectroscopy. The mean activity values for ^{232}U, ^{226}Ra, ^{232}Th and ^{40}K were found to be 42 ± 11, 27 ± 6, 36 ± 5 and 496 ± 56Bq/Kg respectively. These values fall within the range of typical world and Greek values for soil. The indoor radiological concentrations levels which were also measured in 66 dwellings by the means of Solid State Nuclear Track Detector (SSNTD), which ranged from 12 to 1.29Bq/m^3 with an average value of 36 ± 2Bq/m^3. These values are for indoor radon door external irradiation of terrestrial origin and to indoor internal irradiation from the short lived decay products ^{222}Rn was estimated to be 1.22mSv/yr for adult, which is lower than the global effective dose due to natural sources of 2.4mSv/yr.

Jibiri, *et al.*, (2007) measured the radioactivity on some terrestrial food crops in Jos-Plateau. The results showed a higher concentration of potassium among the essential

elements in all the crops. Calcium concentration in both maize and guinea corn was below detection level and also zinc in green beans, sweet potato and cassava. The trace elements Sc, Hf, Sm and Zn were below detection limit in all the crops except in green beans. The activity concentration of the radionuclide in the food samples was found to vary between 83 and 129 Bqkg^{-1}for ^{232}Th. They therefore rejected that the mining activity in the area influenced the concentration levels of those elements and radionuclide in the food crops.

(Bashir, *et al*., 2013) assessed the radiological risk in flooded soil of Kudenda, Kaduna State Nigeria. The results of the activity concentration of^{235}U, ^{232}Th, and ^{40}K was found to be 81 $\pm$ 1.3 <-> 4514.4, 38 $\pm$1.3 <-> 149.6 $\pm$ 3.9 and 400.5 $\pm$ 3.9 <-> 873.7 $\pm$ 11.6Bqkg^{-1} respectively. The AEDE was found to be varying from 0.060 $\pm$ 0.003 to 0.170 $\pm$ 0.005 mSvy^{-1} with arithmetic mean value of 0.110 $\pm$ 0.004mSvy^{-1} which is higher than the worldwide effective dose of 0.07mSvy^{-1}.

(Girigisu, 2012) assessed the radiological levels in soils from Bagega artisanal gold mining sites Zamfara state, Nigeria. The results shows that mean values of activities in the order ^{40}K > ^{226}Ra > ^{233}Th gives $370.79 \pm 7.98 Bqkg^{-1}, 18.13 \pm 1.77\ Bqkg^{-1}, 16.86 \pm 0.94 Bqkg^{-1}$. The mean value of outdoor gamma dose rate is $27.25 nGgh^{-1}$ which were below the global average value. The mean annual effective dose is $33.44 \mu Svy^{-1} (= 0.0033 mSvy^{-1})$. These values are low radiological speaking and should not cause health concerns, however the unprofessional practices of the miners which includes non usage of gas masks put them at risk of exposure to respiratory crystalline silica as well as exposure to radon gas inhalation.

(Girigisu, *et al*., 2014) assessed the radiological levels from Awwal artisanal gold mining exercises in Kebbi State. Soil samples were collected from Awwal artisanal gold mining

sites in Kebbi state, using gamma-ray spectrometry technique. Results show mean values of activities of ^{40}K>^{226}Ra>^{232}Th numerically as $425.96 \pm 5.56,\ 23.85 \pm 2.01\ and\ 18.80 \pm 1.21 Bq/Kg$, respectively. The average outdoor gamma dose was $34.26 nGy/h$ while the mean annual effective dose rate was $42.15 \mu Sv/year\ (= 0.042 mSv/year)$, which is less than $0.07 mSv/year$ benchmark given in (United Nations, 1993). Radiological speaking the values obtained are low and do not imply any significant health concerns effects on the local population. However, the observed unprofessional practices such as lack of use of gas mask while working in the dust filled mine caves and at the mills could expose workers to possible risks from inhalation of respiratory crystalline silica as well as exposure to radon gas.

(Innocent, *et al.*, 2014) determined the quality and quantity of radioactivity in the soil samples randomly collected from 10 abandoned mine sites in Gusau, Zamfara State. The results obtained show that the activity concentrations for ^{40}K, ^{226}Ra and ^{232}Th ranged from 227.10 ± 7.54 to $570.44 \pm 10.57 Bqkg^{-1}, 4.68 \pm 3.52\ to\ 18.98 \pm 0.84 Bqkg^{-1} and\ 40.58 \pm 1.85\ to\ 94.92 \pm 2.76 Bqkg^{-1}$ respectively. The values of the radiological hazards indices further shows that radium equivalent activity ranged from $94.41 \pm 5.64\ to\ 191.34 \pm 7.11\ Bqkg^{-1}$ with mean value of $130.93 \pm 4.94 Bqkg^{-1}$. the mean values of gamma index level, external and internal hazard indices are 0.48, 0.35 and 0.39 respectively.

(Audu, *et al.*, 2013) assessed the radiological effects of iron ore mining and processing at Itakpe, Kogi State. The mean measured activity concentrations of ^{40}K, ^{226}Ra and ^{234}Th were $207.6 \pm 39.4\ (ND - 515.0), 17.6 \pm 2.4 (ND - 31.5), and\ 31.7 \pm 8.6\ (ND - 118.0) BqKg^{-1}$, respectively, which were all less than the world mean of 370.0, 25.0 and

$25BqKg^{-1}$for ^{40}K, ^{226}Ra and ^{232}Th, respectively. The calculated outdoor mean effective doses for the sites were respectively $74.8 \pm 20.0\mu Sv,\ 47.0 \pm 7.0\mu Sv\ and\ 17.0 \pm 1.0\ \mu Sv$ for industrial plant, mining waste and mill tailing deposit sites. These values are low comparable to the control mean value of $74.3 \pm 46.2\mu Sv$, thus suggesting that the mining company does not constitute radiological problem to its workers and the public.

(Fasal, *et al.*, 2013) assessed the natural radioactivity in locally produced building materials in Ekiti State, southwestern Nigeria. The activity concentration of ^{40}K, ^{226}Ra and ^{232}Th in the brick block and concrete block samples was determined using gamma ray spectrometry. The result showed that the mean concentration values of $572.6 \pm 175.9Bqkg^{-1},\ 47.9 \pm 9.8Bqkg^{-1} and\ 63.8 \pm 9.4Bqkg^{-1}$ for ^{40}K, ^{226}Ra and ^{232}Th respectively in concrete blocks are higher compared to the mean activity concentration values $351.1 \pm 3.1Bqkg^{-1}, 18.7 \pm 6.2Bqkg^{-1}\ and\ 39.8 \pm 3.5Bqkg^{-1}$ obtained in brick blocks. However the use of brick block and cement block materials for constructing buildings for human inhibition in the study area may pose a health risk to the population if there is no remediation by regulatory body in the State.

(Popoola, *et al.*, 2012) assessed the concentration of potentially harmful heavy metals in playground soil from public primary schools in metropolitan Lagos. The Investigation revealed that Pb has the highest concentrations of all the metals. The mean metal concentration in play soils were $Pb_{HD} 23.08 \pm 11.11, Pb_{LD} 23.54 \pm 14.55;\ Cr_{HD}\ 5.99 \pm 5.79,\ Cr_{LD}\ 3.80 \pm 3.83;\ Cd_{HD} 0.33 \pm 0.33, Cd_{LD} 0.39 \pm 0.31;\ Mn_{HD} 1.60 \pm 0.14, Mn_{LD}\ 1.61 \pm 0.005\mu g/g$. (LD is Lower Density while HD is Higher Density group).

(Arogunjo, 2009) determined the activity concentration in soils and mineral sands from the Nigerian tin mining area of Bisichi, located in the Jos Plateau. The activities of uranium

and Thorium measured in the soils and mineral sands from Bisichi ranged from $8.7 Bqkg^{-1}\ to\ 51 Bqkg^{-1}\ for$ ^{238}U and from $16.8\ Bqkg^{-1}$ to $98\ Bqkg^{-1}$ for ^{232}Th respectively. These values were significantly higher than those in the control areas of Jos city and Akura as well as the reference values reported in literature (UNSCEAR, 2000).

(Nguelem, *et al.*, 2013) measured the activity concentration of ^{222}Rn, ^{40}K and ^{232}Th in ground water samples using high purity germanium (HpGe) detector. The estimated average annual effective dose due to consumption in the water samples were $113.007 \pm 3.940\ and\ 76.568 \pm 2.321 \mu Sv/y$ respectivily. These were compared with the estimated average annual dose to ingestion of nuclides in WHO ($100 \mu Sv/y$ and the estimated average annual dose to ingestion of radionuclides in food and water ($290\ \mu Sv/y$) by (UNSCEAR, 2000). The results show that consumption of ground water may not pose any radiological health hazard to the public.

(Zakari, *et al.*, 2013) determined the outdoor absorbed dose rate and estimated annual effective dose rate from the naturally occurring radionuclides ^{238}U, ^{232}Th and ^{40}K around Birnin Gwari artisanal gold mine using instrumental neutron activation analysis (INAA). The result showed that the mean absorbed dose rate is 96.52 nGy/h with a standard deviation of 33.3nGy/h and the standard error of 9.62nGy/h. the range falls between 26.67-123.85 nGy/h. the mean effective dose rate was estimated to be 0.675 mSv/year with a standard deviation of 0.234mSv and standard error of 0.068mSv and in range between 0.183-0.867mSv.

(Amasi, *et al.*, 2014) determined the mean activity concentration of ^{226}R, ^{232}Th, and ^{40}K in Tanzania content ranging from 2.6 to 93.2, 1.3 to 172.8 and 6.3 to 997 Bq/kg using gamma spectrometer coupled with HpGe detector respectively. The average activity concentrations

in the cement were 46, 28 and 228 Bq/kg respectively. The average annual effective dose to an occupant from use of these materials equals to 0.45mSv.

(Kalagbor, *et al.*, 2014) investigated the presence of Cr, Mn, Ni, Co, Cu, Cd, Zn, and Pb in four of the most commonly consumed vegetables in the southern part of Nigeria, which are fluted pumpkin, bitter leaf, scent leaf, and water leaf. The metal analysis results showed the concentrations (mg/Kg) as follows; Cr (1.50-10.25), Mn (9.75-62.75), Ni (15.75-19.25), Co (1.75-3.00), Cu (7.75-11.00), Cd (1.25-1.50), Zn (79.75-186.95) and Pb (6.25-8.00). The concentrations of the metals are in the order $Zn > Mn > Ni > Cu > Pb > Cr > Co > Cd$. The water leaf having the highest concentration of the metals Zn, Mn, Ni, Co, Cr and Cd. Fluted pumpkin has the second highest concentration of Mn and the highest for Cu. This therefore implies that the consumption of these vegetables will eventually lead to serious health problems in the organ and circulatory system.

(Oyedele, *et al.*, 2008) measured the activity concentrations of the naturally occurring ^{238}U, ^{232}Th, and ^{40}K in soil; samples collected from four major towns across the northern region of Namibia which were determined using gamma-ray spectrometry with the objective of providing baseline data on the radiation level in the region. The average concentrations of radionuclides in the towns vary from 7.5 ± 2.3 *to* $14.2 \pm 3.3\ Bqkg^{-1}$ for ^{238}U, 5.8 ± 2.6 *to* $24.9 \pm 6.1\ Bqkg^{-1}$ for ^{232}Th and 52.1 ± 28.7 *to* $380.1 \pm 112.9\ Bqkg^{-1}$ for ^{40}K. These concentrations were used to calculate the mean absorbed dose rate and the mean annual effective dose in the region. The low value of $21 \pm 16\ \mu Sv$ obtained for the mean annual effective dose indicates that the region has normal background radiation.

(Avwiri & Esi, 2014) performed a radiometric survey of Aluu Landfill in Rivers State, Nigeria and reported that the mean site radiation levels in all the four cardinal points range

from 0.0123 ±0.0026 mR/hr (1.034mSv/y) to 0.0151 ± 0.0012mR/h (1.270mSv/y). The equivalent dose has an average value range of 1.001mSv/y to 1.270mSv/y. They concluded that the radiation levels exceeded the normal background level of 0.013mR/h and equivalent dose is slightly above the recommended dose limit of 1.0mSv/y for the general public.

(Oladapo, *et al.,* 2012) determined the natural radionuclide level in waste land soil around olusoson dumpsite Ojota, Nigeria using Na (Tl) detector. A total of 30 samples (15 from active site and 15 from dormant site) were analyzed. The mean value of ^{288}U, ^{232}Th and ^{40}K concentrations determined for soil samples from the active dumpsite were 69.69± 19.10 $Bqkg^{-1}$, 14.49 ± 3.22 Bq kg^{-1} and 409.44 ± 86.08$Bqkg^{-1}$ respectively. For the soil samples from the dormant dumpsite, the mean value of ^{238}U, ^{232}Th and ^{40}K concentrations for soil samples from both the active and dormant dumpsites were higher than the permissible global values of 52.2$Bqkg^{-1}$, 41.6$Bqkg^{-1}$, and 230.0$Bqkg^{-1}$ respectively by UNSCEAR1 The mean annual effective dose obtained for soil samples from active dumpsite is 0.2767 while 0.2550 was obtained for soil samples from the dormant dumpsites. Both were found to be below the recommended standard by (UNSCEAR, 2000).

(Okeyode & Rufai, 2011) determined the elemental composition of copper, manganese, iron, zinc, lead, nickel, magnesium and sodium in the soil samples from some selected dumpsites and 20m away from the dumpsites within Abeokuta metropolis in Ogun state, Nigeria, using atomic absorptions spectrophotometer. At the dumpsites, the concentrations of iron, manganese, zinc, lead, nickel, magnesium, sodium and copper ranged from 86.89 – 277.33mg/l, 1.9731 – 5.6644mg/l, 0.3299 – 4.847mg/l, 0.4132 – 2.8137mg/l, 0.0896 – 0.1376mg/l, 1.9731 – 5.6644mg/l, 0.9203 – 29.00mg/l and 0.1645 – 5.4191mg/l

1UNSCEAR means United Nations Scientific Committee on the Effects of Atomic Radiation

respectively, while at the control sites (20m away from dumpsite), the concentrations ranged from 37.64 – 109.94mg/l, 0.5342 – 1.7910mg/l, 0.1015 – 0.3692mg/l, 0.2510 – 0.7595mg/l, 0.0590 – 0.0881mg/l, 0.0927 – 0.3976mg/l, 0.4385 and 0.9876mg/l and 0.004 and 0.416mg/l respectively. The concentrations of metals in solid at the decomposed biodegradable wastes dumpsites and 20m away from dumpsites showed that there is an evidence of relative increase in concentrations of metals in solids at dumpsites compared to those 20m away from dumpsites.

(Said, *et al*., 2010)assessed the environmental radioactivity impacts and health hazard indices at Wali Sahu Area, Sinai, Egypt, the effective annual dose rate (EDR) of the total area is ranging between 0.18 – 3.50mSv/a with range value 1.84mSv/a.

(Odunaike, *et al*., 2009)surveyed the radiation dose of refuse dumpsites in Abeokuta in Ogun state in the southwestern zone of Nigeria, the result revealed the absorbed dose ratio from the dumpsite to range from 12.3 to 49.1mSv $year^{-1}$ and mean annual absorbed dose rate to be 360.0µSv $year^{-1}$.

CHAPTER 3: MATERIALS AND METHODS

3.1 Materials

The materials used in this study are listed below:

i. Topological map of the study area

ii. Global positioning system (GPS)

iii. Survey meters for field measurements

iv. Detectors for laboratory soil/tailing samples analysis

v. 30cm soil auger for removal of stones and vegetable matter

vi. Polythene bags

vii. Gas mask

viii. Adhesive masking tape

ix. Indelible marker for collected samples labeling

x. Surgical gloves

xi. Gamma ray spectrometry system

xii. Atomic absorption spectrometry system

xiii. Rados model RDS 120

3.2 Methodology

3.2.1 The Study Area

Angwan Maigero gold mining site is located on latitude 10^0 1’ 29”N to 10^0 1’ 30”N and longitude 6^0 28’ 30”E to 6^0 28’ 31”E.

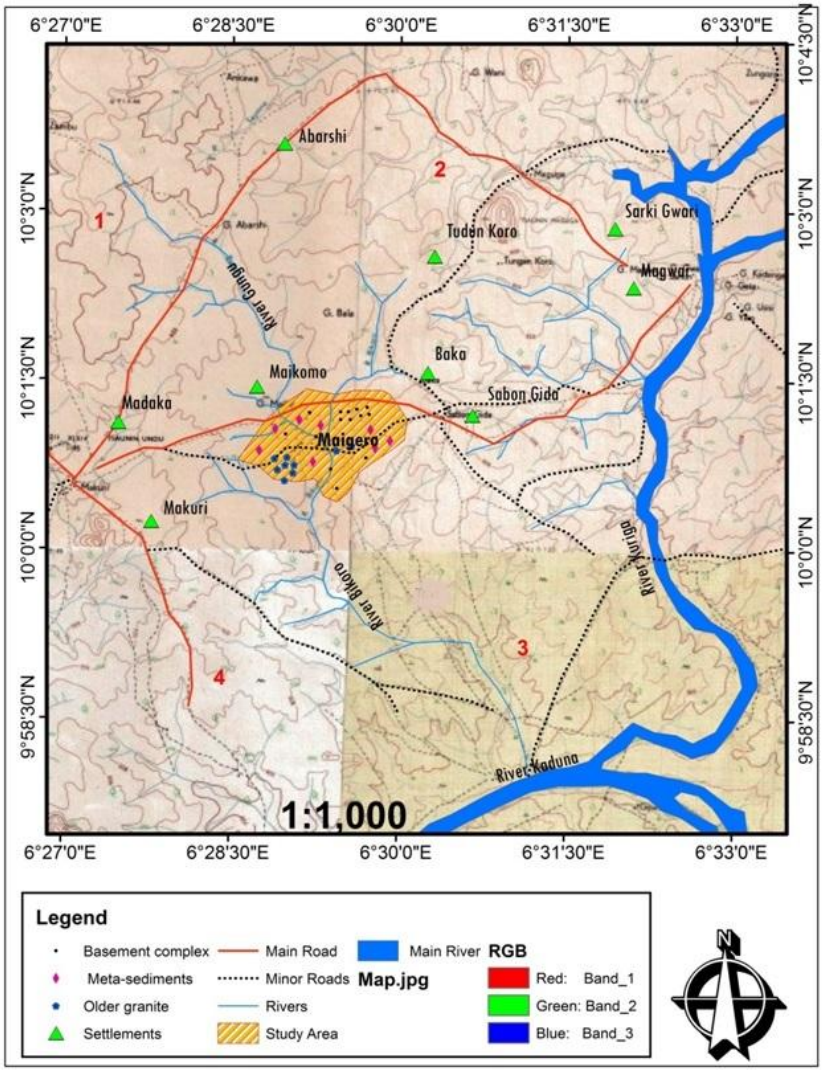


Figure 3.1 Topological map of Rafi Local Government Area showing Angwan Maigero

Source: GIS laboratory, Department of Geography, ABU, Zaria (2015)

3.2.2 Climate of the study area

Niger state experiences distinct dry and wet season with annual rainfall varying from 1100mm in the northern part to 1600mm in the southern part. The maximum temperature is recorded between March and June, while the minimum is usually between December and January. The rainy season lasts for about 150 days in the northern part to about 120 days in the southern part of the state.

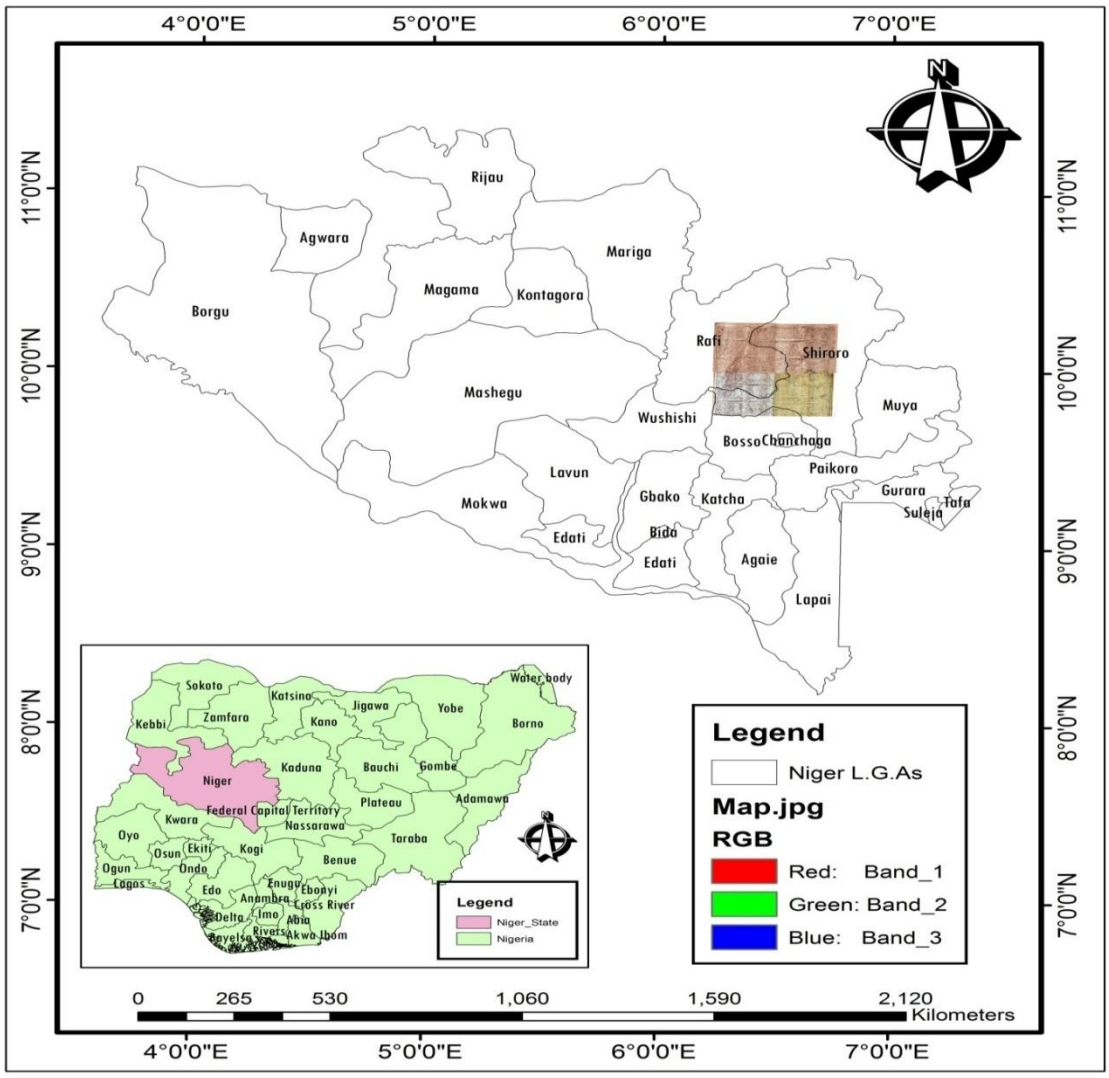


Figure 3.2 The 25 constituent local government areas of Niger State

Source: GIS Laboratory, Department of Geography, ABU Zaria (2015)

3.2.3 Geology of the study area

Niger State is characterized by two major rock formations; the sedimentary and basement complex rocks. The sedimentary rocks to the south are characterized by the sandstones and alluvial deposits; to the north is the basement complex rock.

Angwan Maigero is characterized by the basement complex rock, with granitic outcrops which can be found in the vast topography of rolling landscape.

3.2.4 Site Preparation for Sampling

The mine site is 80m^2 which was divided into 32 grid points of 20m^2, 16 grid points at each horizontal level for both right and left direction of the mining pit. Also for the mill tailing site the dimension of sheds on average were about 3m by 3m, the mill tailing site was divided into 9 grid points of 3m^2. A 30cm hand auger was used for the removal of stones and vegetable matter.

3.2.5 Sample Collection Procedures

The sampling strategy chosen was the horizontal profile sampling, since the Gold mining site is a virgin site, hence no depth profile needed. The field measurement and sampling were made at the artisan mines. For the purposes of adequate data interpretation the sampled positions were marked with a portable hand held Global Positioning System (GPS) and samples were systematically collected from a girded area of 80 m^2, 49 soil and mill tailing samples were collected altogether from the mine and mill tailing sites, respectively. Control soil samples were collected from an undisturbed site far from the main studied site.

3.2.5.1 Horizontal Profile Sampling

The mine site was partitioned into $20m \times 20m$ grid points labeled R11-L44. The grid points R11-R14, R21-R24, R31-R34 and R41-R44 are parallel while R11-R41, R12-R42, R13-R43 and R14-R44 in the right direction of the mine site are perpendicular to the mining pit and also grid points L11-L14, L21-L24, L31-L34 and L41-L44 are parallel

while L11-L41, L12-L42, L13-L43 and L14-L44 in the left direction of the mine site are perpendicular to the mining pit, respectively. Sixteen (16) soil samples, each were collected from both right and left direction in the mine site. A total of 32 soil samples were collected from the grided area as shown in fig. 3.2. Four (4) soil samples were collected from the direction of sampling locations in the mining pit. In the mill tailing site the dimension of the sheds on average were about 3m by 3m. “Most of this space is swallowed by the milling machine and its accessories, the sacs of crushed rocks for milling as well as sacs of milled materials”. “Added notable features associated with the sheds are lack of strong partitioning of one shed from the other, and the reduction in visibility due to generated dust from the pulverized radioactive rocks in the mill tailing site”. Nine (9) mill tailing samples were collected from the mill tailing site. In-situ gamma dose rate measurements were taken and samples were collected at the middle of each grid using hand auger. Control samples were collected 3km away from the mining area. At each sample point, small amount of surface soil and mill tailings were taken for AAS elemental analysis while another as much as $350g$ were also taken for gamma spectroscopy using sodium iodide thallium drifted [NaI(TI)] detector type. Each sample collected was put into a sampling polythene bag, tied and labeled appropriately. The bags were doubled for each sample to prevent leakage and cross contamination of samples. A total of 49 samples were taken with 4 samples being control.

3.2.6 In-Situ Measurements

In-situ measurements of terrestrial gamma radiation were made based on the assumption that there exists laterally uniform distribution of natural radionuclides in the environment and that the vertical contribution from soil is limited to the first horizon (Ibeanu, 1999).

These measurements were made with a portable survey meter (Rados model RDS 120) having sensitivity range of $0.05\mu Svh^{-1}$-$10Svh^{-1}$. Three in-situ dose rate measurements were made per sampled point. Their mean provided the exposure rate at that point while the mean at each horizontal level were recorded as the exposure rate of that location.

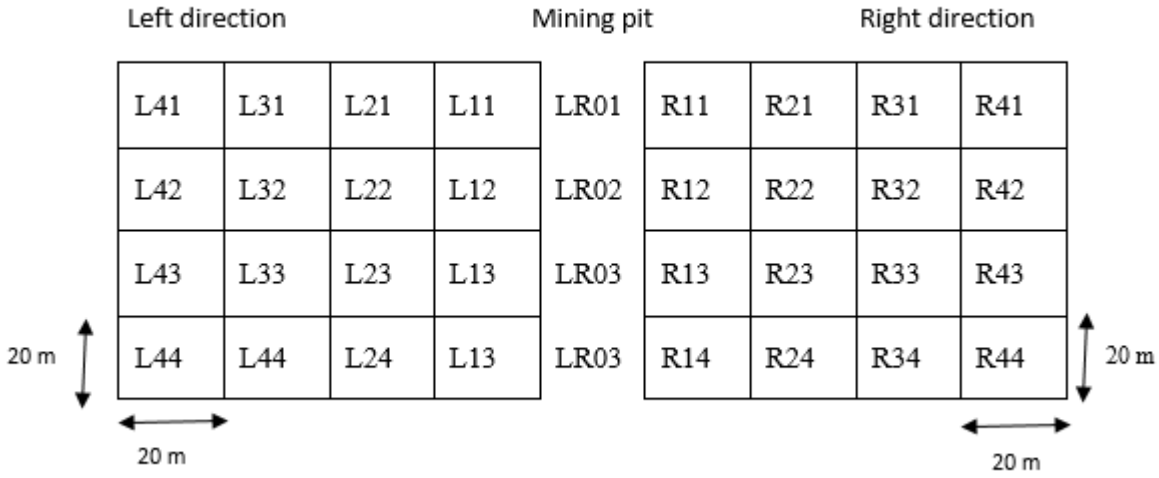


Figure 3.3 Schematic diagram showing sample collection grid points

3.3 Analytical Methods

The techniques used in the laboratory were the gamma ray spectrometry system for characterization of activity concentration levels of ^{40}K, ^{226}Ra and ^{232}Th (NORMS) in the soil and mill tailing samples collected and flame atomic absorption spectrometry system for characterization of soil and mill tailing samples for elemental composition and distribution.

3.3.1 Sample Preparation

Sample preparation was based on the sample types and the intended analytical technique to be used in the analysis.

3.3.1.1 Gamma Spectroscopy

The soil and mill tailing samples were dried for 72 hours under laboratory temperature (ambient temperature of 27℃) and average relative humidity of 70% (IAEA, 1989). The samples were then grounded and packed to fill already weighted cylindrical plastic containers of dimension 7.2cm in diameter and 6.0cm high to satisfy the selected best sample container height that adequately match the detectors dimensions, which was allowed for the adequate covering of the detector shield (Ibeanu, 1999). Before sealing, the mass of each were determined and recorded.

The sealing of the sample containers were done in three stages, namely: Vaseline wax sealing, Candle wax sealing and adhesive masking tape, all done to prevent the escape of ^{226}Ra gas. The samples were then stored for a minimum of four weeks to achieve an approximate secular equilibrium and their respective progeny prior to counting.

3.3.1.2 Atomic Absorption Spectrometry

The samples were air-dried at ambient laboratory temperature and grounded manually to fine powder using an agate mortar and pestle to grain size of less than 2mm and stored for analysis.

3.4 Sample Analysis

3.4.1 Gamma Spectrometry

The characteristic photon energy can be measured in gamma spectrometry because it is possible to record the count rates in distinct energy windows specific for the gamma radiation energy of a particular nuclide (Ibeanu, 1999).

3.4.2 Experimental Set-up

The use of gamma spectrometry in the analysis of bulk samples requires that the emitted characteristic gamma rays from the samples were assessed either directly or indirectly to quantify the amount of particular radionuclide/element of interest. In the use of gamma spectrometry, the gamma spectral from natural (un-irradiated) samples was measured in order to determine the bulk uranium, thorium and potassium content of the sample.

According to Ibeanu, (1999), gamma spectrometry is a widely used analytical technique that has seen many applications in natural environmental studies. The method has been documented to have various advantages such as the cheapness, speed of analysis, relative ease of sample preparations and the simplicity of the procedures of data reduction.

The instrument for measuring gamma radiation converts photon energies into electrical current. The most widely used instruments are solid state detectors and scintillation counters out of which NaI(TI) gains very wide acceptance. In a gamma spectrometer, the characteristic photon energy is measured and used to quantify the different radionuclides. This was done by recording the count rate in distinct energy windows specified for the gamma radiation energy of particular nuclide. For this work, below is a simplified block diagram of experimental set-up of NaI(TI) gamma spectrometry used. The detector assembly is enclosed in a 6.0 cm thick lead shield supported internally by line-ups of cadmium and copper sheets. The lead shield minimizes the effect of background radiation, while the inner sheets absorb the fluorescent x-rays and scattered photons. Fig 3.2 shows a simplified NaI(TI) detector Assembly of the experimental setup.

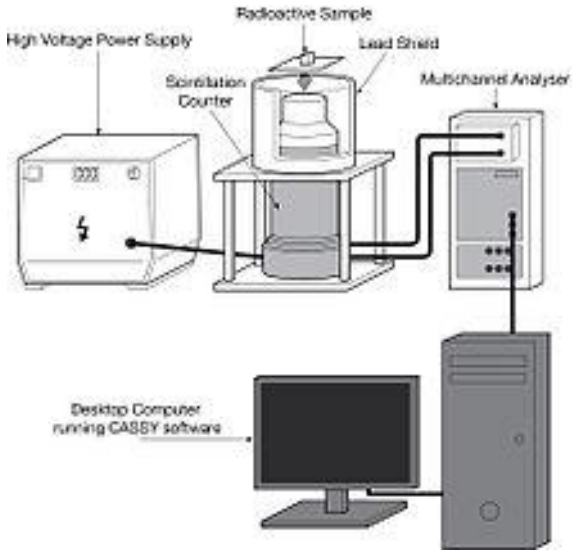


Figure 3.4 Experimental Assembly

The assembly consist of a 7.62cm ×7.62 cm Na(TI) detector optically coupled to a photomultiplier tube (PMT). The stated set-up is then incorporated with NIM bin consisting of preamplifier, amplifier and MCA system.

The backbone of the gamma spectrometry is the analysis of one of the physical properties of most radionuclides undergoing decay; emitted gamma rays (Ibeanu, 1999). The detection of radionuclides by gamma spectrometry depends on the interaction of the photons with the detecting materials. For efficient conversion of the gamma energies, detector materials with the characteristics of effective absorption of the incident gamma energies (that is good scintillation properties) are used. In these materials, the photon energies are converted to lower energy photons which then undergo further conversion into electrical pulses via a photomultiplier tube (PMT). The intensity of each pulse is proportional to the quantity of the nuclide characterized by the particular gamma energy. The most commonly used scintillation materials are the high density TI-activated sodium iodide NaI(TI). This has the advantage of possessing high atomic number, which increases

the probability of interaction. This high probability of photon interaction increases the conversion efficiency of the detector which ultimately results in relatively high photo peak efficiency of the detector.

Ibeanu (1999) stated there are two renowned ways of achieving spectra analysis for the energy discrimination needed for the qualitative and quantitative analysis of the radionuclides; the integral and differential spectrometry. The integral spectrometry approach involves the recording of the whole spectrum from a predetermined low position that covers the energy peak of interest. The differential spectrometry involves acquisition of information on the energy peaks with an energy window set at about the peak.

In this particular work, differential spectrometry was employed in three channels. The ^{40}K channel which was determined from the 1460keV energy peak, the ^{238}U channel of 1764keV energy peak and the ^{232}Th channel having energy peak of 2614keV.

Table 3.1 Spectral energy windows used in analysis using NaI(TI) gamma spectroscopy

Element analyzed	Isotope used	γ-Ray energy (keV)	Energy windows (keV)
Th	Tl	2614.5	2460-2820
U	Bi	1764.0	1620-1820
K	K	1460.0	1380-1550

3.4.3 Build-Up to Analysis

The quantitative determination of the activity concentration of a radionuclide is usually obtained by absolute or relative approach. Using the absolute approach, the equation below by (Hayumbu *et al*., 1995) was employed in the calculation.

$$A_a = \frac{C_n\ 1}{\varepsilon_\gamma P_\gamma} \quad (3.1)$$

Where,

A_a = Activity of the radionuclide (in Bq)

ε_γ =Detector efficiency for the analytical γ-line

P_γ = intensity of the gamma source for the analytical γ-line

C_n = Count rate (cps) of the radionuclide for the analytical γ-line

The operation requires running of the acquisition interface control program as a startup procedure. This program stays resident and initializes the hardware for the multichannel analyzer (MCA) program which provides the main display and control functions of the system.

3.4.4 Calibration of NaI(TI) Spectrometer

The calibration of MCA is important for the generation of platform for finding an expression that relates particle energy to the channel number (particle-energy storage). For this work, the MCA housing of a MAESTRO (a version of software analyzer) was used. The process applied in `calibrating the NaI(TI) detector involved the setting of amplification system steadily and gradually to 900V along with the routine procedure of initializing readings. The calibrating standard sources selected for use was ^{60}Co and ^{137}Cs. One after the other, each of the selected standards were placed inside the detector and counted for a period of 100 seconds. The resulting gamma spectrum of each standard source output was then obtained, the centroid of the peak of each energy was determined and their associated uncertainties. These were saved and served as the calibrated benchmark for the measurement.

The standard source were removed from the detector and replaced in their housing. Samples to be analyzed were then placed in turn, in the detector. With the use of the calibrated energy, the new centroid energies of required energy peak were determined after a count of 29000 seconds. The required energy peaks of interest in this observation were of ^{238}U, ^{232}Th, and ^{40}K.

The resulting data from the analysis were converted to conventional units by the use of calibration factors for determination of their activity concentrations. The characteristic photon energy detected will be measured and used to quantify different radionuclides.

3.4.5 Background Measurement

The results of the net count rates obtained from the system included that contributed by the natural radionuclides in the laboratory, therefore to obtain that due to the samples alone, the laboratory background measurement were made and later subtracted from the net counts rates obtained from the system. The laboratory background measurement was made by counting an empty detector without any sample for 29000s. From the spectrum obtained, only the naturally occurring radionuclides produced observable measurements.

3.4.6 Calibration factors (CF) and Methods of Measured Data Interpretation

Determination of calibration factors of the standard nuclide implored for the analysis was done by application of the following relations:

$$\text{CF } ^{40}\text{K} = \frac{cps\,(40_K)}{ppm\,(40_K)} \text{ or } \frac{cps\,(40_K)}{Bq((40_K)/kg} \tag{3.2}$$

$$\text{CF } ^{226}\text{Ra} = \frac{cps\,(226_{Ra})}{ppm\,(226_{Ra})} \text{ or } \frac{cps(226_{Ra})}{Bq((226_{Ra})/kg} \tag{3.3}$$

$$\text{CF } {}^{232}\text{Th} = \frac{cps\,(232_{Th})}{ppm\,(232_{Th})} \text{ or } \frac{cps\,(232_{Th})}{Bq((232_{Th})/kg} \quad (3.4)$$

Source; (Ibeanu, 1999)

Based on the above, the obtained calibration factors for this work and subsequently the activities at calibration were used to arrive at current activities of the radioisotopes. For the estimation of concentration of these nuclides in the samples, they were counted and net count per second (cps) obtained with the same region of interest as set in standards. Then the evaluation of activity concentrations will be made thus:

$${}^{40}\text{K} = \frac{cps\,(40_K)\text{sample}}{CF_K} \quad (3.5)$$

$${}^{226}\text{Ra} = \frac{cps\,(226_{Ra})\text{sample}}{CF_{Ra}} \quad (3.6)$$

$${}^{232}\text{Th} = \frac{cps\,(232_{Th})\text{sample}}{CF_{Th}} \quad (3.7)$$

The calibration factors (CF), were obtained from the experimental work of (Ibeanu, 1999) on the same instrument and same set of radionuclides. The resulting table from the stated work on the calibration factor is as below:

Table 3.2 Energy calibration for Quantitative spectra analysis

Isotope	Calibration Factor		Conversion Factor	Detection Limit	
	$\times 10^{-3}$cps/ppm	$\times 10^{-4}$ cps/Bqkg^{-1}	**(Bqkg^{-1}/ppm)**	Ppm	Bq/kg
^{40}K	0.026	6.431	0.032	454.54	14.54
^{226}Ra	10.500	8.632	12.200	0.32	3.84
^{232}Th	3.612	8.768	4.120	2.27	9.08

Source: (Ibeanu, 1999)

3.5 Analysis for Atomic Absorption Spectrometry

With the aid of spatula and weighing bottle, 0.5g of each sample was obtained using a high sensitive chemical balance (model SNRK96452, China.). This was placed in a Teflon beaker and transferred to a fume-cupboard for digestion. The digestion was carried out using concentrated nitric (10 mL) and concentrated perchloric (5 mL) acids in the ratio of 2:1 and the oven was maintained at 200^{o}C. After one hour, the mixture was allowed to cool before leaching the residue with $5cm^3$ of 20% HNO_3. Digested samples were then filtered and made up to $100ml$with deionized water.

A blank determination was treated in the Atomic Absorption Spectrometer but without sample. Solution of samples were then taken and aspirated into Atomic Absorption Spectrophotometer (Unicam Solar A.A.S 969 model) for analyzing metals. Blank determination was also carried out as in a similar way as described above except for the omission of the sample. A calibration graph was plotted for each element using measured absorbance and the corresponding concentration. The calibration curve was used to determine the concentration of the metal.

3.6 Assessment of Radiation Hazards Associated with NORM

Radiological parameters such as the radium equivalent indices, external index, gamma index, absorbed dose rate, annual effective dose and excess lifetime cancer risk were used to determine the potential radiation hazard associated with soil samples associated with the artisanal mines.

3.6.1 Radium Equivalent Activity (Ra_{eq})

The Radium equivalent activity is used to assess the hazards associated with minerals that contain ^{226}Ra, ^{232}Th and ^{40}K in Bq/kg, and it is mathematically defined by (Girigisu, *et al.*, 2014; Beretka & Matthew, 1995). It appears the activity levels of ^{226}Ra, ^{232}Th and ^{40}K by a single quantity which takes into account the radiation hazards associated with equation 3.9

$$Ra_{eq}(Bq.Kg^{-1}) = A_{Ra} + 1.43A_{Th} + 0.077A_K \quad (3.8)$$

From it has been assumed that 10Bq.Kg^{-1} of ^{226}Ra, 7Bq.kg^{-1} of ^{232}Th and 130Bq.Kg^{-1} of ^{40}K produce equal gamma dose.

3.6.2 Absorbed Dose (D)

The uniform distribution of the naturally occurring radionuclides (^{226}Ra, ^{232}Th and ^{40}K) was calculated based on guidelines provided by (UNSCEAR, 2000). It is assume that the contributions from other naturally occurring radionuclides were insignificant. Therefore, the Dose, D, was calculated by (UNSCEAR, 2000) as:

$$D(nGy.h^{-1}) = 0.462A_{Ra} + 0.621A_{Th} + 0.0417A_K \quad (3.9)$$

3.6.3 Annual Effective Dose Equivalent (AEDE)

To estimate the annual effective dose rates, the conversion coefficients from absorbed dose in air to effective dose (0.7 Sv.Gy^{-1}) and outdoor occupancy factor (0.2) proposed by (UNSCEAR, 2000) will be used. Therefore the annual effective dose rate (mSv.yr^{-1}) will be calculated using:

$$AEDE\ (\mu Sv/y) = D\ (nGy/h) \times 8760h \times 0.7Sv/Gy \times 10^{-3} \quad (3.10)$$

3.6.4 External Hazard Index (H_{ex})

A widely used hazard index (reflecting the external exposure) called the external hazard Index, H_{ex}, defined by (Girigisu, *et al*, 2014; Bertka & Matthew, 1995)

$$H_{ex} = \left(\frac{A_{Ra}}{370}\right) + \left(\frac{A_{Th}}{259}\right) + \left(\frac{A_K}{4810}\right) \quad (3.11)$$

The value of the index (H_{ex}) must be less than unity for the radiation hazard to be negligible.

3.6.5 Gamma Index (I)

The gamma index (I) was calculated using Equation 3.13 (UNSCEAR, 2000):

$$I = 0.0033A_{Ra} + 0.005A_{Th} + 0.0005A_k \quad (3.12)$$

According to UNSCEAR (2000), for radiological effects to be considered negligible, the values of each of I must be less than 1.

3.6.6 Excess Lifetime Cancer Risk (ELCR)

The Excess Lifetime Cancer Risk (ELCR) was calculated using Equation (3.14) (Taskin, *et al*., 2009);

$$ELCR = AEDE \times DL \times RF \quad (3.13)$$

where, AEDE is the Annual Equivalent Dose Equivalent, DL is the average duration of life (estimated to 70 years), and RF is the Risk Factor (Sv^{-1}), i.e. fatal cancer risk per Sievert. For stochastic effects, ICRP uses RF as 0.05 for public (Taskin, *et al*., 2009).

3.7 Assessment of Heavy Metal Contamination

Risk assessment codes such as Contamination Factor (CF), Pollution load Index (PLI), Degree of Contamination (DC) Index, Geo-accumulation Index (I-geo), Hazard Quotient (HQ) and Potential Ecological Risk Factor (ERF) were used to determine the heavy metal accumulation and its pollution status.

3.7.1 Contamination Factor (CF)

The Contamination factor (CF) was calculated using

$$CF = \frac{C_m\, sample}{(C_m\, Background\ )} \tag{3.14}$$

Where $C_m\ sample$is the concentration of the sample,

$C_m\ Background$ is the sample along the control area

If $CF < 1$: Indicates low contamination

$1 < CF < 3$: Indicates moderate contamination

$3 < CF < 6$: Indicates considerable contamination

$CF > 6$: Indicates very high contamination

3.7.2 Degree of Contamination (DC)

The Degree of Contamination (DC) of a determined area is the sum of all Contamination factors:

$$\mathrm{DC} = \sum \mathrm{CF} \tag{3.15}$$

Table 3.3 Classification of degree of contamination.

Degree of Contamination	Classification
$DC < 1$	Low
$1 \leq DC < 3$	Moderate
$3 \leq DC < 6$	Considerable
$DC \geq 6$	Very high

Source: (Hakanson, 1980)

3.7.3 Pollution Load Index (PLI)

The pollution load Index (PLI) is aimed at providing a measure of the degree of overall contamination. The PLI of the place was calculated by obtaining the nth root from the $nCFs$ that was obtained for all metals. Generally pollution load Index (PLI) is given as follows:

$$PLI = \sqrt[n]{CF_1 \times CF_2 \times \ldots . CF_n} \quad (3.16)$$

Where,

$CF =$ Contamination factor

$n =$Number of heavy metals studied

The PLI value of > 1 is polluted, whereas < 1 indicates no pollution (Harikumar, *et al*., 2009). This measurements has however been defined by some authors in several ways. In this study, the author found it appropriate to express the PLI as the geometric mean of the studied pollutants since this methods tends to reduce the outliers, which might bias the reported results.

3.7.4 Geo-Accumulation Index (I_{geo})

This method assesses the metal pollution in terms of seven (0 to 6) enrichment classes, ranging from background concentration to very heavily polluted (Muller, 1969), as follows

$$I_{geo} = \log_2 \left[\frac{C_m\, sample}{1.5 \times (C_m\, Background\ )} \right] \quad (3.17)$$

The factor 1.5 is introduced in the equation to minimize the effects of possible variation in the background values. The classification of the index is tabulated below.

Table 3.4 Classification of Geo-Accumulation Index

I_{geo} Value	I_{geo} Classification	Level of Concentration
$\mathbf{5 < I_{geo} \leq 10}$	6	Extremely contaminated
$\mathbf{4 < I_{geo} \leq 5}$	5	Strongly to extremely contaminated
$\mathbf{3 < I_{geo} \leq 4}$	4	Strongly contaminated
$\mathbf{2 < I_{geo} \leq 3}$	3	Moderately to strongly contaminated
$\mathbf{1 < I_{geo} \leq 2}$	2	Moderately contaminated
$\mathbf{0 < I_{geo} \leq 1}$	1	Uncontaminated to moderately contaminated
$\mathbf{I_{geo} \leq 0}$	0	Uncontaminated

Source: (Muller, 1969)

3.7.5 Ecological Risk Factor (ERF)

Ecological risk Factor (ERF) is critical to measure both risk factor and metals concentrations in soil. The potential ecological risk index can be determined through the following formula:

$$ERF = CF * TRF \quad (3.18)$$

Where ERF is the potential ecological risk factor, TRF represents the toxic-response factor, and CF represents contamination factor. The toxic response values for some of the heavy

toxic elements are As=10, Cr=2, Cd=30, Cu=5, Pb=5, Ni=5, and Zn=1 as suggested by (Hakanson, 1980). This is calculated separately for each metal. The results are as follows:

Table 3.5 classification of potential ecological risk factor

Potential Ecological Risk	Classification
$PER < 40$	Low
$40 \leq PER < 80$	Moderate
$80 \leq PER \leq 160$	Considerable
$160 \leq PER \leq 320$	High
$PER \geq 320$	Very high

Source: (Hakanson, 1980)

3.7.6 Hazard Quotient (HQ)

The soil Hazard Quotient (HQ) is the ratio of the heavy metal concentration of surveyed soil samples to reference permissible limit and is computed using the relation;

$$HQ = Cc / Cp \quad (3.19)$$

Where Cp and Cc = reference maximum permissible limit of heavy metal concentration and the concentration obtained in the sampled area respectively.

CHAPTER 4: RESULT AND DISCUSSION

Assessment of radionuclide in soils and rocks in many parts of the world has been on the increase in the past two decades because of their hazards on the health of the populace (Belivermis, *et al.,* 2009). The research into natural radionuclide in the soil and rocks of north western Nigeria is grossly at the lowest ebb. To open focus on this area, a number of toxicological models were used so as to ascertain the pollution level of the heavy metals of interest and also radiological indices where used to access the radiological risk involved in the area. The mean values of in-situ readings were obtained and the necessary comparisons were made. The results will form baseline data for the region under investigation and will resonantly open up the unobserved health risks that lock behind the gold mining activities to the mine workers and inhabitants around the area. As a measure to ensure that hazard assessments covers an acceptable range, elemental assessment was carried out on the samples collected so as to identify the chemical properties of the elements involved, with a view of predicting possible hazards of the heavy metal pollution levels. As a means of giving direction to this work, the results are organized in two phases namely:

Phase1. Elemental assessment

Phase2. Radiological assessment

4.1Elemental Assessment

Mining of heavy metals is a major route of exposure of toxic metals. This phase was carried out to determine the heavy metal accumulation and its pollution status using Contamination Factor (CF), Pollution load Index (PLI), Degree of Contamination (DC)

Index, Geo-accumulation Index (I-geo), Hazard Quotient (HQ) and Potential Ecological Risk Factor (ERF).

4.1.1 Heavy Metal Concentrations

Tables 4.1, 4.2 and 4.3 summarize the concentrations (ppm) of the ten (10) heavy metals observed in the mine site, mill tailing site and control area.

Table 4.1 Concentration of the heavy metals at the mine site

Heavy Metal	Concentration (ppm)	
	Mean	Range
Pb	319.76 ± 26.20	50.10-650.01
Fe	1707.63 ± 237.79	1000.22-1980.57
Mn	181.61 ± 82.38	40.23-360.66
Cu	90.95 ± 59.42	17.82-232.62
Zn	37.41 ±33.30	0.00-121.80
As	17.99 ± 4.53	6.09-26.52
Cr	93.19 ± 47.39	20.55-205.25
Co	47.94 ± 5.71	35.85.60.45
Ni	22.28 ± 6.17	9.99-35.55
Cd	3.23 ± 2.07	0.00-5.67

Table 4.2 Concentration of heavy metals at the mill tailing site

Heavy Metal	Concentration (ppm)	
	Mean	Range
Pb	846.87 ± 100.27	62.01-950.00
Fe	1914.99 ± 89.06	1783.86-2039.58
Mn	177.70 ± 50.51	121.66-258.80
Cu	739.33 ± 337.57	238.14-1031.25
Zn	100.41 ± 41.37	0.00-141.34
As	43.52 ± 7.97	22.61.48.67
Cr	163.28 ± 39.48	105.55-240.35
Co	53.48 ± 2.79	51.39-58.85
Ni	45.59 ± 18.54	28.95-90.35
Cd	5.83 ± 0.48	5.35-6.75

Table 4.3 Concentration of heavy metals at the control area

	Concentration (ppm)	
Heavy Metal	Mean	Range
Pb	21.40 ± 6.11	15.58-30.01
Fe	1331.00 ± 142.75	1134.75-1469.13
Mn	68.10 ± 23.99	40.23-95.97
Cu	35.72 ± 16.45	17.82-56.37
Zn	11.91 ± 8.91	0.00-18.81
As	5.73 ± 2.71	2.32.-8.875
Cr	47.85 ± 4.78	42.54-52.50
Co	18.17 ± 5.65	12.45-25.54
Ni	14.26 ± 3.26	9.99-17.05
Cd	0.23 ± 0.30	0.00-0.67

The mean and standard deviations of the concentrations of heavy metals of interest in the mine site was observed to be 319.76 ± 26.20, 1707.63 ± 237.79, 181.61 ± 82.38, 90.95 ± 59.42, 37.41 ± 33.30, 17.99 ± 4.53, 93.19 ± 47.39, 47.94 ± 5.71, 22.28 ± 6.17, and 3.23 ± 2.07 respectively. Also the mean and standard deviations of the concentrations of heavy metals of interest in the mill tailing site was observed to be 846.87 ± 100.27, 1914.99 ± 89.06, 177.70 ± 50.51, 739.33 ± 337.57, 100.41 ± 41.37, 43.52 ± 7.97, 163.28 ± 39.48, 53.48 ± 2.79, 45.59 ± 18.54, 5.83 ± 0.48. The Highest concentration corresponds to Fe and the lowest corresponds to Cd. The increasing trend was in the order: Cd <As <Ni <Zn <Co<Cu <Cr <Mn <Pb <Fe. The concentrations of all the elements in the mill tailing site were observed to be higher than that obtained in the mine site with greater variation in relative abundance of most of the metals. These variations can be attributed to the variable dumping of mill tailing deposit. The mean and standard deviations of the concentrations of heavy metals in the control area were observed to be 21.40 ± 6.11, 1331.00 ± 142.75, 68.10 ± 23.99, 35.72 ± 16.45, 11.91 ± 8.91, 5.73 ± 2.71, 47.85 ± 4.78, 18.17 ± 5.65, 14.26 ± 3.26 and 0.23 ± 0.30 respectively.

Tables 4.1, 4.2, and 4.3 presents the mean measured concentrations of each of the heavy metals studied in the mine site, mill tailing site and control area respectively. The mean concentrations observed in the mine and mill tailing sites were seriously higher than that in the control area for all the studied heavy metals. Figures 4.1 - 4.20 presented the concentrations (ppm) of the heavy metals analyzed (Pb, Fe, Mn, Cu, Zn, As, Cr, Co, Ni, Cd) in the mine site and mill tailing site respectively. Heavy metals were detected at varying concentrations in the samples. The Mean concentrations in the mine site, mill tailing site and control area were compared with the world health organization guideline on the maximum limits of toxic metals in the soils (WHO, 1996) as provided in Table 4. Most the observed heavy metals in the mine site and mill tailing site were observed to have concentrations above the world recommended value by WHO, while in the control area the concentrations were below the world recommended value by WHO. Pb was observed to have concentration 3 times higher than the recommended world recommended value by WHO, so also Co, was observed to have concentration 2 times higher than the world recommended value by WHO, and Cd, was observed to have concentration of about 2 times higher than the world recommended value by WHO, in the mine site, and for the mill tailing site Pb was observed to have concentration 8 times higher than the world recommended value by WHO and also Co, Cd, Ni and Cr were observed to have concentrations about 2 times higher than the world recommended value by WHO.

The obtained concentration of the heavy metals (ppm) in the mine site, mill tailing site and control area were compared with the Romanian guideline for the allowed maximum normal concentration values in the soils and world median values as presented in presented in section 2. Based on the guideline and the obtained concentration of the study area, it

could be deduced that (i) Pb, As, Cd, Cr, Cu exceeded the world median and intervention threshold in the sensitive area at both mine site and mill tailing site.

(ii) Ni exceeded the normal legal value but within the world wide median.

While the concentration in the control area were

(i) Pb and As are above the normal legal value but within the world median value.

(ii) Cd, Cu and Cr above the normal legal value and worldwide median

(iii) Zn and Ni were within the normal legal value and worldwide median.

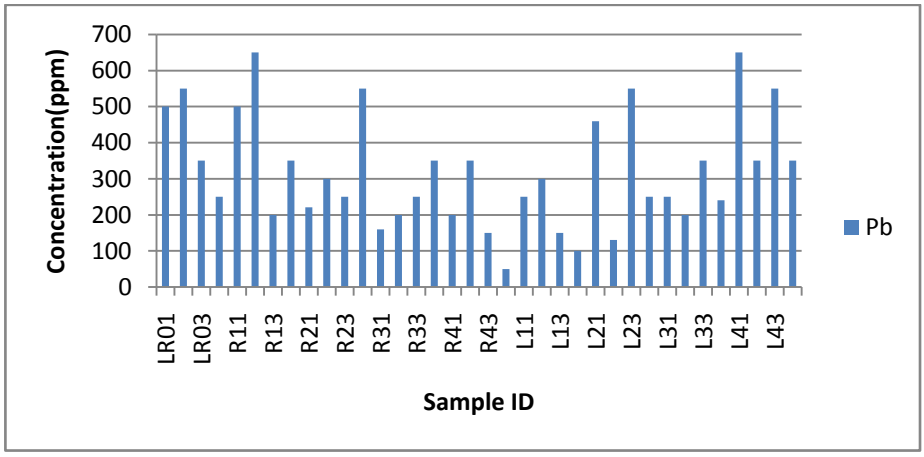


Figure 4.1 Concentration of Pb at the mine site

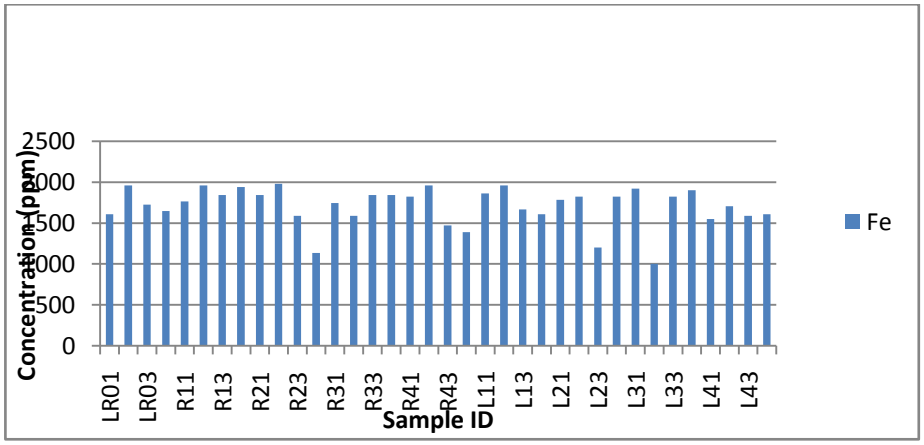


Figure 4.2 Concentration of Fe at the mine site

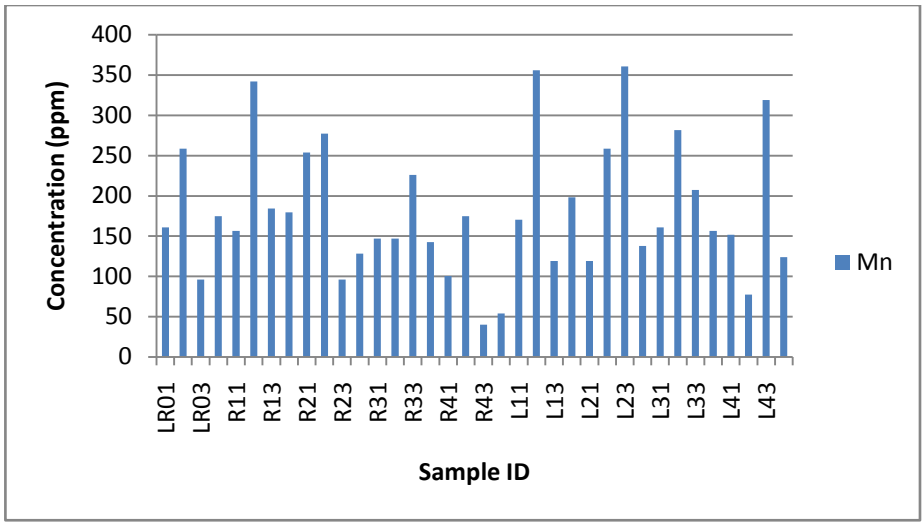


Figure 4.3 Concentration of Mn at the mine site

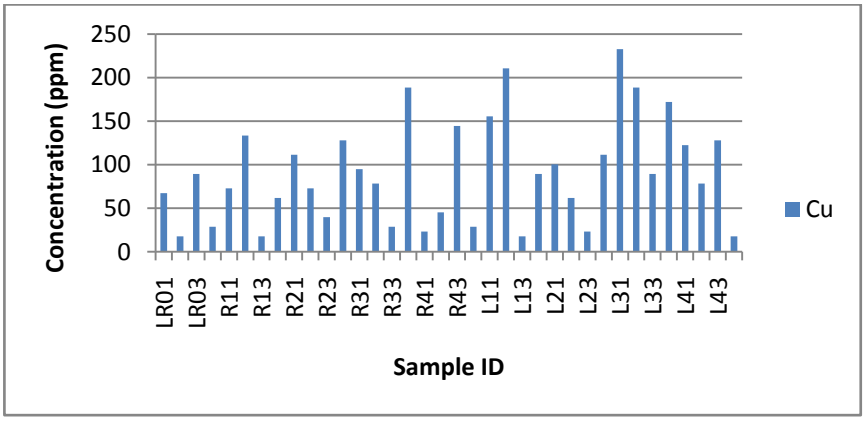


Figure 4.4 Concentration of Cu at the mine site

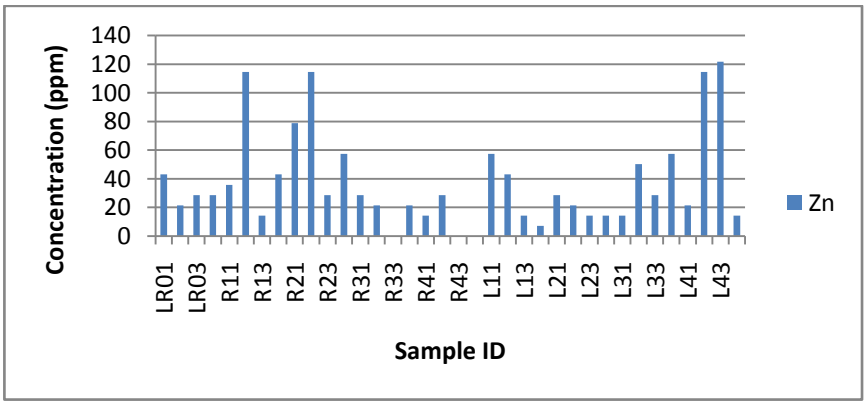


Figure 4.5 Concentration of Zn at the mine site

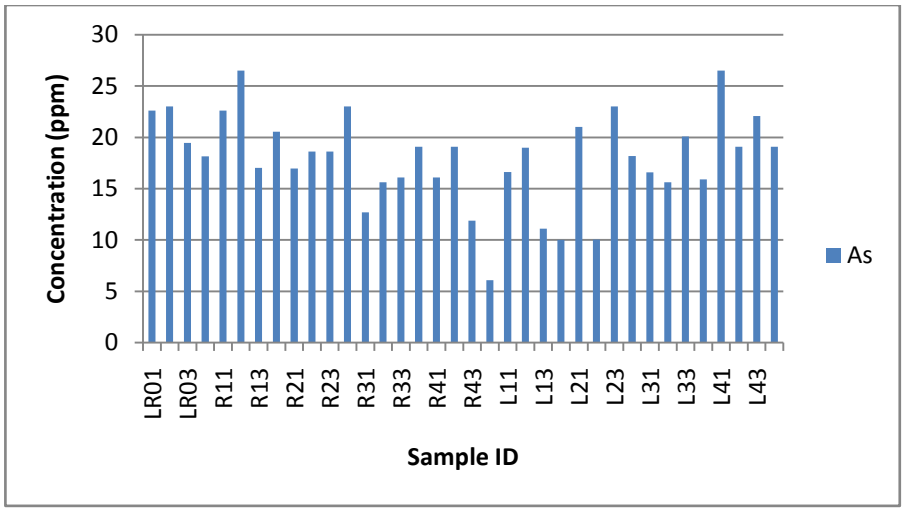


Figure 4.6 Concentration of As at the mine site

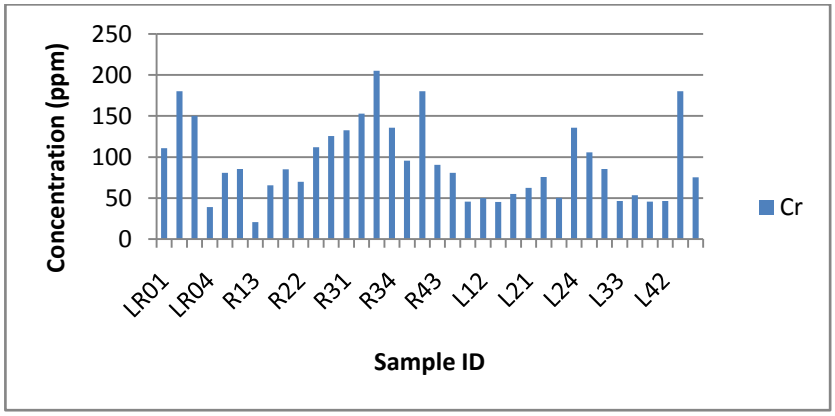


Figure 4.7 Concentration of Cr at the mine site

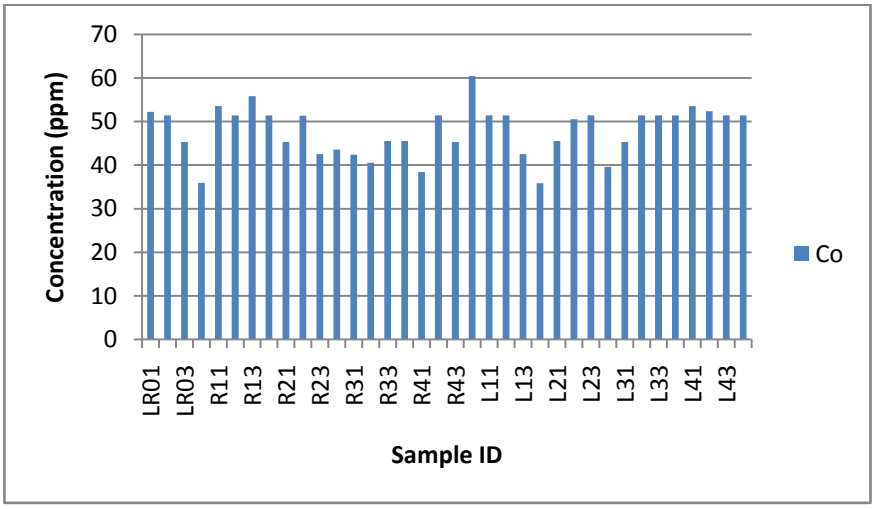


Figure 4.8 Concentration of Co at the mine site

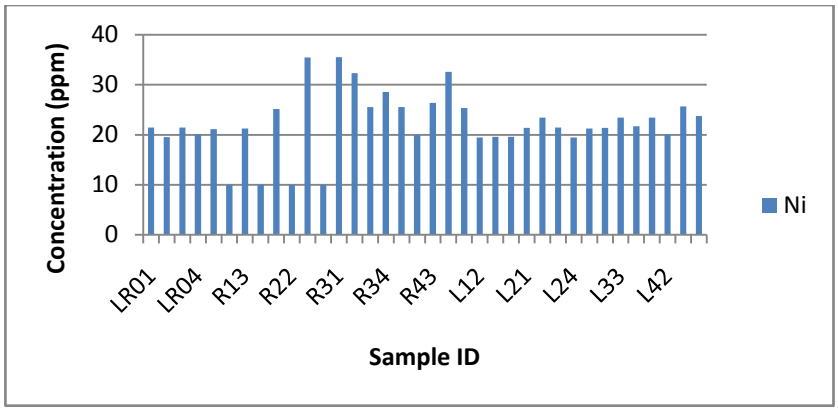


Figure 4.9 Concentration of Ni at the mine site

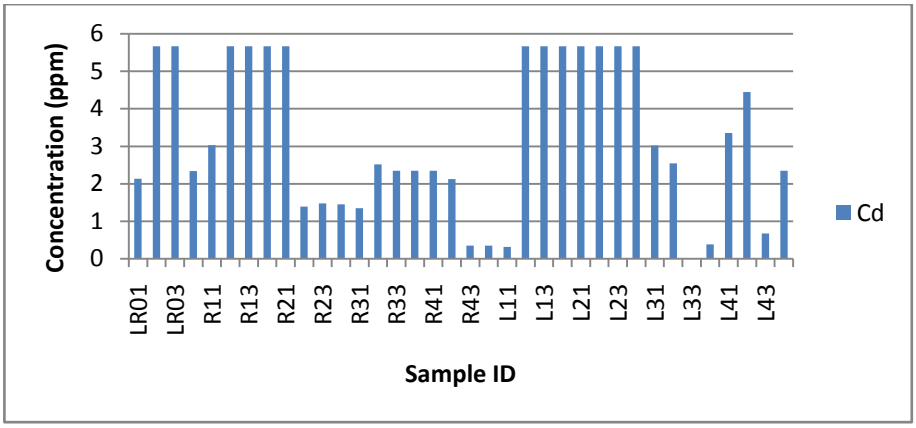


Figure 4.10 Concentration of Cd at the mine site

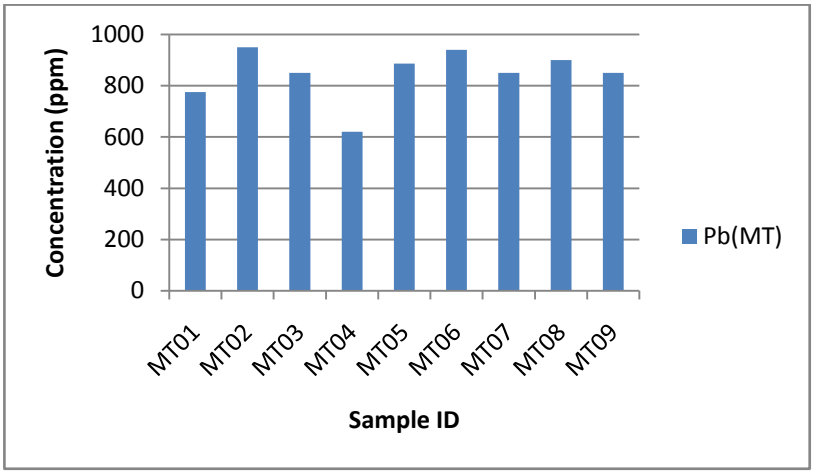


Figure 4.11 Concentration of Pb at the mill tailing site

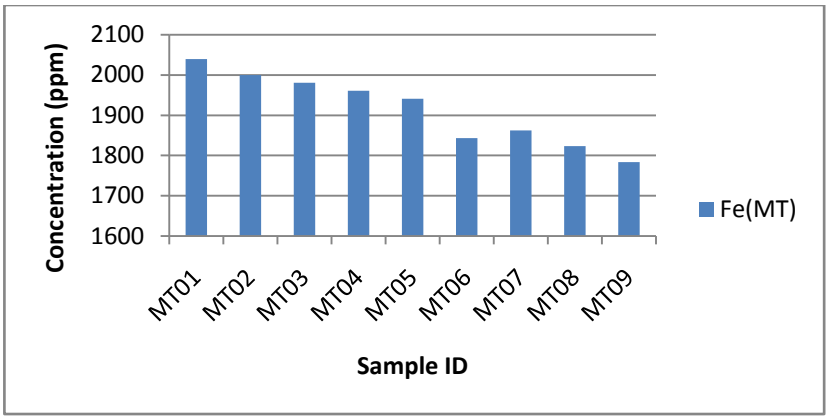


Figure 4.12 Concentration of Fe at the mill tailing site

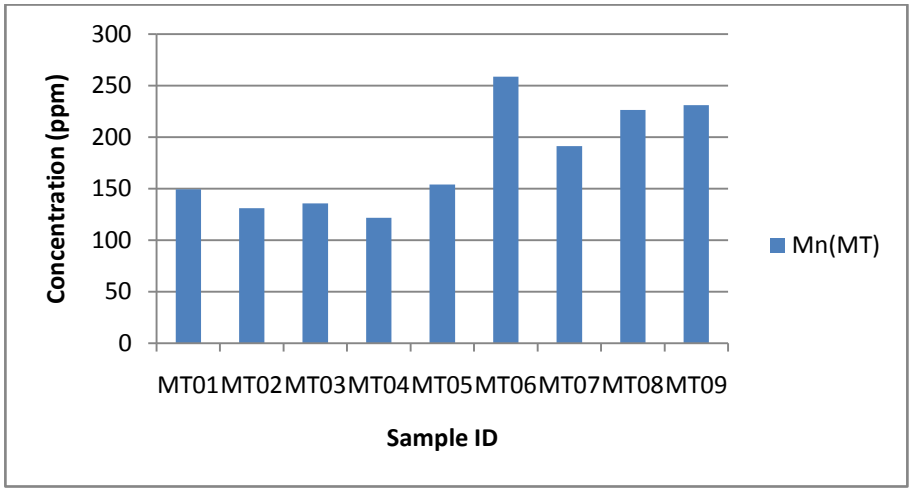


Figure 4.13 Concentration of Mn at the mill tailing site

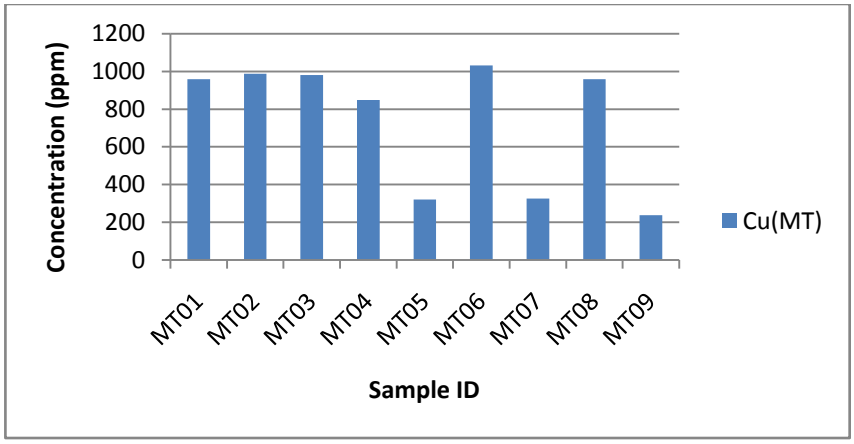


Figure 4.14 Concentration of Cu at the mill tailing site

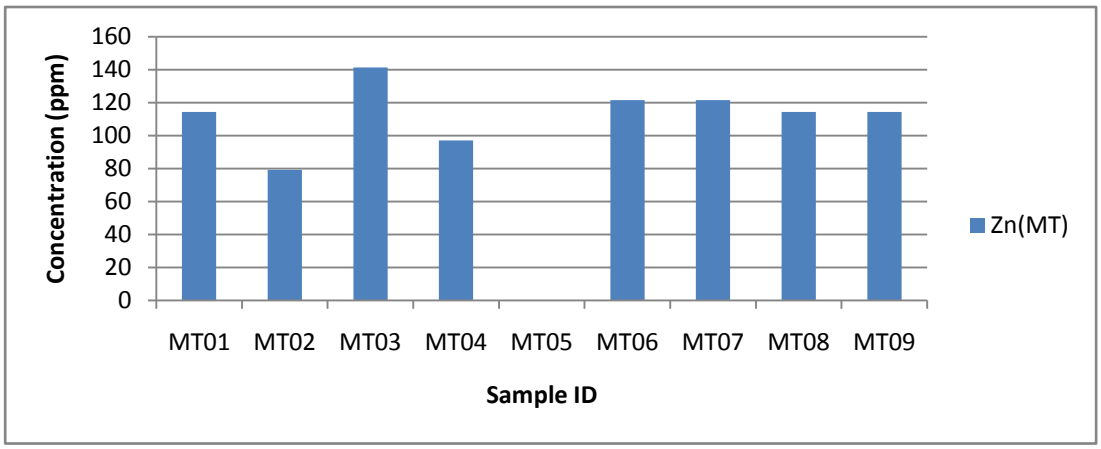


Figure 4.15 Concentration of Zn at the mill tailing site

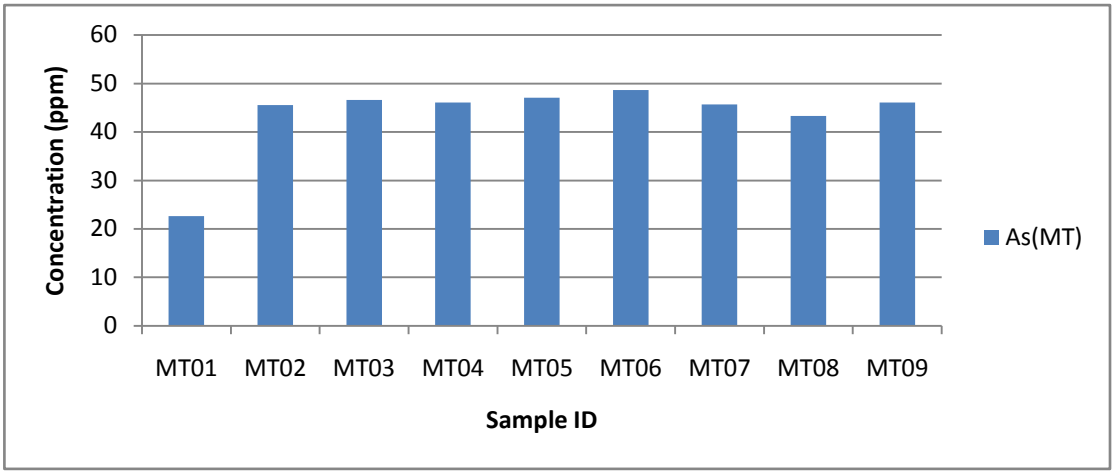


Figure 4.16 Concentration of As at the mill tailing site

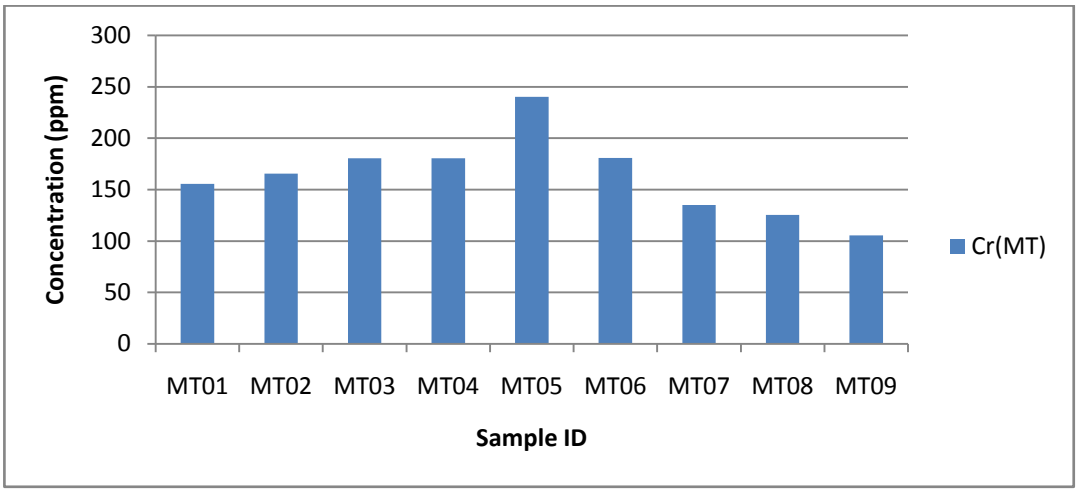


Figure 4.17 Concentration of Cr at the mill tailing site

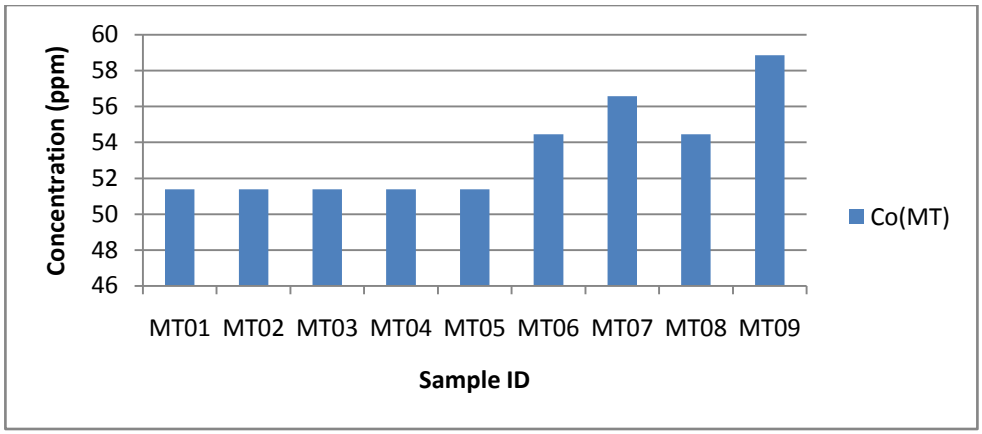


Figure 4.18 Concentration of Co at the mill tailing site

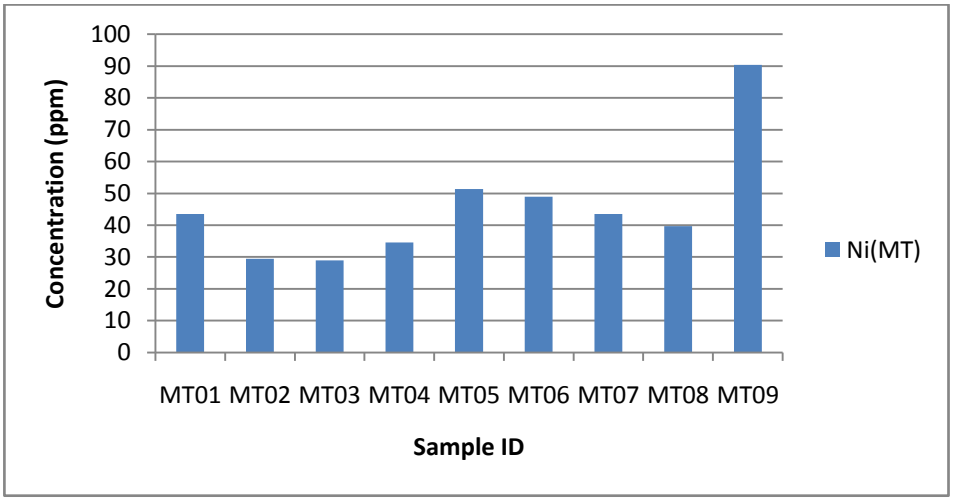


Figure 4.19 Concentration of Ni at the mill tailing site

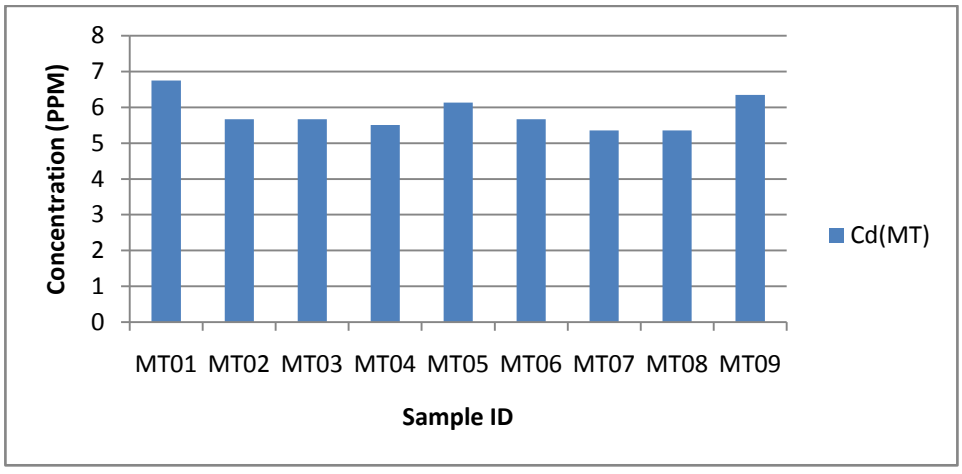


Figure 4.20 Concentration of Cd at the mill tailing site

4.1.2 Contamination Factor, Degree of Contamination and Pollution Load Index

Tables 4.5 and 4.6 present the summary of the contamination factors of the analyzed samples at the mine and mill tailing sites, respectively. The contamination factor was used to assess the level of contamination of each element in the studied samples, based on the categories discussed in chapter 2, the contamination factor could be categorized as follows in the mine site (i) Pb and Cd in the very high contamination category (ii) As and Zn in the considerable contamination category (iii) Mn, Cu, Co, Ni, Fe and Cr in the moderate contamination category and for the mill tailing site (i) Pb, Cu, Zn, As, and Cd in the very high contamination category (ii) Cr and Ni in the considerable contamination category (iii) Fe, Mn, and Co in the moderate contamination category.

Tables 4.7 and 4.8 presented the statistics of degree of contamination and pollution load index. This pollution load index (PLI) was used to assess the overall pollution level of the site resulting from the observed metals. The pollution load index results indicated that the mine site was contaminated with all the observed heavy metals (mean PLI = 2.71) and also the pollution load index results indicated that the mill tailing site was contaminated with all the observed heavy metals (mean PLI = 5.78). The Results of the Degree of contamination index indicated that the mine site is in very high degree of contamination category (Mean DC = 47.84) and also (Mean DC = 115.09) for the mill tailing site as described in section 2.

Table 4.4 Contamination factors of the Soil Samples at the mine site

	Contamination factors		
Heavy Metal	Mean	Range	Standard Deviation
CF (Pb)	14.94	2.34-30.38	7.35
CF (Fe)	1.28	0.75-1.49	0.03
CF(Mn)	2.67	0.59-5.30	1.21
CF(Cu)	2.55	0.50.6.51	0.28
CF(Zn)	3.14	0.00-10.23	0.47
CF(As)	3.14	1.06-4.63	0.13
CF(Cr)	1.95	0.43-4.29	0.17
CF(Co)	2.64	1.97-3.33	0.05
CF(Ni)	1.56	0.70-2.50	0.43
CF(Cd)	13.96	0.00-24.49	1.49

Table 4.5 Contamination factors (CF) of the Soil Samples at the mill tailing site

	Contamination factors		
Heavy Metal	Mean	Range	Standard Deviation
CF(Pb)	39.58	28.98-44.40	4.69
CF(Fe)	1.44	1.34-1.53	0.02
CF(Mn)	2.61	1.79-3.80	0.25
CF(Cu)	20.70	6.67-28.87	3.15
CF(Zn)	8.43	0.00-11.87	1.16
CF(As)	7.61	3.95-8.50	0.46
CF(Cr)	3.41	2.21-5.02	0.28
CF(Co)	2.94	2.83-3.24	0.05
CF(Ni)	3.20	2.03-6.34	0.43
CF(Cd)	25.17	23.11-29.16	0.69

Table 4.6 Degree of contamination (DC) and the Pollution Load Index (PLI) of the mine site.

Statistical parameter	Degree of Contamination	Pollution Load Index
Mean	47.84	2.71
Minimum	14.85	0.00
Maximum	84.68	4.48
Standard deviation	14.24	1.10

Table 4.7 Degree of contamination (DC) and the Pollution load index (PLI) of the mill tailing site.

Statistical parameter	Degree of Contamination	Pollution Load Index
Mean	115.09	5.78
Minimum	100.25	0.00
Maximum	131.33	7.49
Standard deviation	11.06	2.21

4.1.3 Geochemical Accumulation Index

The Geochemical accumulation indices were computed for all the concentrations of the sample location using the equation given in Section 2. Table 4.9 presents the summary statistics of the computed geo-chemical accumulation indices from which the mean elemental concentration of the studied soil samples in the mine site could be categorized as follows:

i. Pb and Cd in the Polluted Category.
ii. Fe, Mn, Cu, Zn, As, Cr, Co and Ni in the Unpolluted Category.

Table 4.10 presents the summary statistics of the computed geo-chemical accumulation indices, as such, the mean elemental concentration of the studied mill tailing samples in the mill tailing site could be categorized as follows:

i. Pb, As, Cu, Zn, and Cd in the Polluted Category.
ii. Fe, Mn, Cr, Co and Ni in the Unpolluted Category.

Table 4.8 Geochemical accumulation (I-geo) Indices of the analyzed metal in the mine site

	Geochemical accumulation indices		
Metal	Mean	Range	Standard Deviation
(Pb)	1.46	0.71-1.83	0.24
(Fe)	0.45	0.22-0.52	0.07
(Mn)	0.72	0.12-1.07	0.22
(Cu)	0.64	0.04-1.16	0.34
(Zn)	0.77	0.13-1.36	0.31
(As)	0.83	0.37-1.01	0.13
(Cr)	0.58	-0.02-0.98	0.23
(Co)	0.76	0.64-0.87	0.05
(Ni)	0.52	0.19-0.74	0.14
(Cd)	1.38	0.49-1.73	0.39

Table 4.9 Geochemical accumulation (I-geo) indices of the analyzed metal in the mill tailing site

	Geochemical accumulation indices		
Heavy Metal	Mean	Range	Standard Deviation
(Pb)	1.94	1.81-1.99	0.06
(Fe)	0.50	0.47-0.53	0.02
(Mn)	1.75	0.60-0.93	0.12
(Cu)	1.60	1.17-1.81	0.26
(Zn)	1.32	1.17-1.42	0.08
(As)	1.22	0.94-1.28	0.10
(Cr)	0.87	0.69-1.05	0.11
(Co)	0.81	0.80-0.86	0.02
(Ni)	0.83	0.65-1.15	0.15
(Cd)	1.74	1.71-1.81	0.04

4.1.4 Hazard Quotient

The hazard quotient for each of the samples in the mine and mill tailing sites were calculated using the Equation given in section 2 and WHO recommended threshold limit for concentrations of metals in soils. The Statistical summary of the hazard quotients of the samples were presented in Tables 4.11 and 4.12. The mean Hazard quotients for all the heavy metals studied such as Pb, Cu, are far greater than unity and also Zn, Cr and Cd are

slightly greater than unity, while Ni are slightly lower than unity in both mine and mill tailing site, respectively. The category of each element are characterized by single element statistical methods used in categorizing the pollution levels in the mine and mill tailing site, respectively.

Table 4.10 Hazard Quotient of the studied heavy metals in the mine site.

Heavy Metal	Hazard Quotient		
	Mean	Range	Standard Deviation
(Pb)	3.20	0.50-6.50	0.26
(Cu)	3.03	0.59-7.75	0.33
(Zn)	0.47	0.00-1.52	0.07
(Cr)	0.93	0.21-2.05	0.08
(Ni)	0.28	0.13-0.44	0.01
(Cd)	1.08	0.00-1.89	0.12

Table 4.11 Hazard Quotient of the heavy metals in the mill tailing site.

Heavy Metal	Hazard Quotient		
	Mean	Range	Standard Deviation
(Pb)	8.47	6.20-9.50	1.00
(Cu)	24.64	7.94-34.38	3.75
(Zn)	1.26	0.00-1.77	0.17
(Cr)	1.63	1.06-2.40	0.13
(Ni)	0.57	0.36-1.13	0.08
(Cd)	1.94	1.78-2.25	0.05

4.1.5 Ecological Risk Factor

The Ecological risk Factors (ERF) was calculated for each metal in each of the samples utilizing the Equation given in section 2.The Summary statistics of this index for some of the studied metals(those whose toxic response factor is in literatures and the contamination factor has been calculated) are presented in Table 4.13 and 4.14.

Thus, the heavy metals computed in the mine and mill tailing sites are categorized as:

For mine site

i. Cd in very high Ecological risk category,
ii. Pb in moderate Ecological risk category,
iii. Cu, Zn, As, Cr, and Ni in the Low potential ecological risk.

For mill tailing sites

i. Cd in very high Ecological risk category,
ii. Pb in the high Ecological risk category,
iii. Cu in moderate Ecological risk category,
iv. As in the moderate Ecological risk category,
v. Zn, Cr, and Ni in the low potential ecological risk category.

Table 4.12 Ecological risk factor of the heavy metal in the mine site

	Ecological risk factor		
Heavy Metal	Mean	Range	Standard Deviation
(Pb)	74.72	11.69-151.89	36.74
(Cu)	12.73	2.50-32.57	8.32
(Zn)	3.14	0.00-10.23	2.80
(As)	31.42	10.63-46.32	7.92
(Cr)	3.90	1.98-8.86	2.58
(Ni)	7.82	3.51-12.47	2.17
(Cd)	418.86	0.00-734.77	267.90

Table 4.13 Ecological risk factor of the heavy metal in the mill tailing site

	Ecological risk factor		
Heavy Metal	Mean	Range	Standard Deviation
(Pb)	197.89	144.88-222.99	23.43
(Cu)	103.51	33.34-144.37	15.75
(Zn)	8.44	0.00-11.87	3.48
(As)	76.00	39.48-84.99	13.91
(Cr)	6.82	4.41-10.05	1.65
(Ni)	16.00	10.16-31.70	2.17
(Cd)	755.22	693.31-874.73	62.26

Table 4.14 Comparison and Correlation of Statistical Methods with heavy metals concentrations in the mine, mill tailing sites and control area with WHO recommended level.

Element	**Mean Background Concentration**	**Mean concentration in the mine site**	**Mean concentration in the mill tailing site**	**WHO recommended level**
Pb	21.40	319.76	846.87	100.00
Fe	1331.45	1707.63	1914.99	N/A
Mn	68.10	181.61	177.70	N/A
Cu	35.72	90.95	739.33	30.00
Zn	11.91	37.41	100.41	300.00
As	5.73	17.99	43.52	N/A
Cr	47.85	93.19	163.28	100.00
Co	18.17	47.94	53.48	N/A
Ni	14.26	22.28	45.59	80.00
Cd	0.23	3.23	5.83	3.00

4.1.6 Correlation between the heavy metals concentration.

The variation in the quantitative trend of the concentration of the heavy metals as well as the horizontal sampling strategy employed in the mine and mill tailing sites indicate a certain measure of similarity between the different sets of samples. This similarity is expected since these concentrations could be influenced as a result of the mining activities of the inhabitants in the area. In an attempt to unravel the relationship between the heavy metals concentrations in the mine and mill tailing sites, the concentrations of the elements

were subjected to Correlation Analysis using Microsoft Excel. The correlation coefficients were displayed in Tables 4.16 and 4.17.

In Tables 4.16 and 4.17 respectively, Pb and As has a very strong correlation, so also with other elements. The implication of this correlation is that where ever one of these metals abounds, the other is to be found as there exist a linear relationship between them. The results of Pearson correlation in the mine site showed a very significant positive correlation between Pb with all the elements except Mn, Fe with Cu, Zn, As, Cr, Co, Ni and Cd, Cu with Zn, As, Cr, Co and Ni, Zn with As, Cr, Co, Ni, and Cd. As with Cr, Co, Ni and Cd. Cr with Co, Ni and Cd. Ni with Cd in the mine site, respectively, so also the results of Pearson correlation in the mill tailing site showed a very significant positive correlation between Pb with all the elements, Mn with Co and Cd, Fe with Cu, Zn, As, Cr, Co, Ni and Cd, Cu with Zn, As, Cr and Ni, Zn with As, Cr, Co, Ni, and Cd. As with Cr, Co, Ni and Cd. Cr with Co, Ni and Cd. Ni with Cd in the mine and mill tailing site respectively. The correlation between the concentrations of each heavy metal at each horizontal range was conducted in the mine and mill tailing sites, respectively. As such, it can be deduced that the correlation between most pair of elements in the mine and mill tailing sites, strongly indicates that the mine site is enhanced from natural sources which is anthropogenically deposited in the mill tailing site, since there exist a linear relationship between them.

Table 4.16 Correlation between pairs of elements in the mine site

	Pb	*Fe*	*Mn*	*Cu*	*Zn*	*As*	*Cr*	*Co*	*Ni*	*Cd*
Pb	1.00000									
Fe	**0.96475**	1.00000								
Mn	-0.07918	-0.30637	1.00000							
Cu	**0.967142**	**0.944996**	0.003733	1.00000						
Zn	**0.953182**	**0.939939**	-0.00285	**0.998502**	1.00000					
As	**0.996794**	**0.973713**	-0.08749	**0.982531**	**0.972848**	1.00000				
Cr	**0.885018**	**0.951052**	-0.52881	**0.801850**	**0.789353**	**0.880523**	1.00000			
Co	**0.882593**	**0.858600**	0.131596	**0.972232**	**0.980575**	**0.911794**	**0.658152**	1.00000		
Ni	**0.961566**	**0.961203**	-0.07083	**0.996596**	**0.997505**	**0.980276**	**0.829185**	**0.966275**	1.00000	
Cd	**0.970481**	**0.875339**	0.093106	**0.91119**	**0.887395**	**0.952332**	**0.792479**	**0.818581**	**0.887255**	1.000

Table 4.17 Correlation between pairs of elements in the mill tailing site

	Pb	*Fe*	*Mn*	*Cu*	*Zn*	*As*	*Cr*	*Co*	*Ni*	*Cd*
Pb	1.00000									
Fe	**0.908665**	1.00000								
Mn	**0.632737**	**0.845778**	1.00000							
Cu	**0.891715**	**0.660981**	0.302755	1.00000						
Zn	**0.956492**	**0.819666**	0.517297	**0.968362**	1.00000					
As	**0.996168**	**0.884627**	0.583948	**0.925399**	**0.976462**	1.00000				
Cr	**0.937334**	**0.911664**	0.563753	**0.769738**	**0.852555**	**0.928249**	1.00000			
Co	**0.782780**	**0.959691**	**0.956329**	0.484122	**0.684693**	**0.747097**	**0.762269**	1.00000		
Ni	**0.955159**	**0.808509**	0.475143	**0.974377**	**0.99763**	**0.977068**	**0.868296**	**0.65775**	1.00000	
Cd	**0.957211**	**0.958697**	**0.817275**	**0.740944**	**0.861742**	**0.931355**	**0.889305**	**0.903044**	**0.846655**	1.00000

4.2 Radiological Assessment

4.2.1 In-Situ Measurements

In-Situ measurements were obtained from the mine and mill tailing site down to the control area**.** These measurements were made with a portable survey meter (Rados model RDS 120) having sensitivity range of 0.05$\mu Sv/h$-10Sv/h. At each location, three readings

were taken. Their mean provided the exposure rate at that point while the mean at each horizontal level was recorded as the exposure rate of that location.

Conversely, a control of the work was carried out (about 3km) having the same underlying geology far away from the study area, readings were taken using the portable hand held survey meters similar to the sample procedure discussed earlier. The result from the survey meter was averaged and the result is shown in Figures 4.90, 4.91 and 4.92. Univariate descriptive statistics was also carried out for the In-Situ dose rate measurements (μSv/h), the annual effective dose rate measurements (Sv/y) and annual effective dose rate measurements in (mSv/y) for the mine site, mill tailing site and control area respectively. It was observed that the dose rate showed some differences in range, indicating that the radioactivity in the control area was not induced by any internal factor, but inherent from the radioactivity of the soil and mill tailing deposit, clearly suggesting that the radiation dose at the mine and mill tailing sites was as a result of the Naturally occurring radioactive materials (NORMs) present in the area.

The annual effective dose was calculated using the ICRP recommendation conversion of 1Sv/y=114.1μSv/y (ICRP, 1991) as shown in Tables 4.16, 4.17 and 4.18. Comparing the annual effective dose of the in-situ with the values set by the International Commission on Radiological Protection (ICRP), which recommends limiting irradiation to the public to an average of 1mSv (0.001Sv) of effective dose per year, not including medical and occupational exposures (ICRP, 1991). Comparing the values of Tables 4.19 and 4.20 with values obtained from around the United States Capitol Building, which was 0.85mSv/y, it can be observed that both are close to the regulatory limit, with the values from the US capitol building slightly lower than the recommended values. The annual effective dose

rate ranged from (1.29-2.03) mSv/y with a mean of 1.44 mSv/y in the mine site and (1.76-2.30) mSv/y with a mean value of 1.76mSv/y in the mill tailing site, higher than the control area values which ranged from (0.86- 0.96) mSv/y, with a mean value of 0.86. As such the annual effective dose of the mine and mill tailing site in this case can be potentially significant to the dose burden of the inhabitants living in the area.

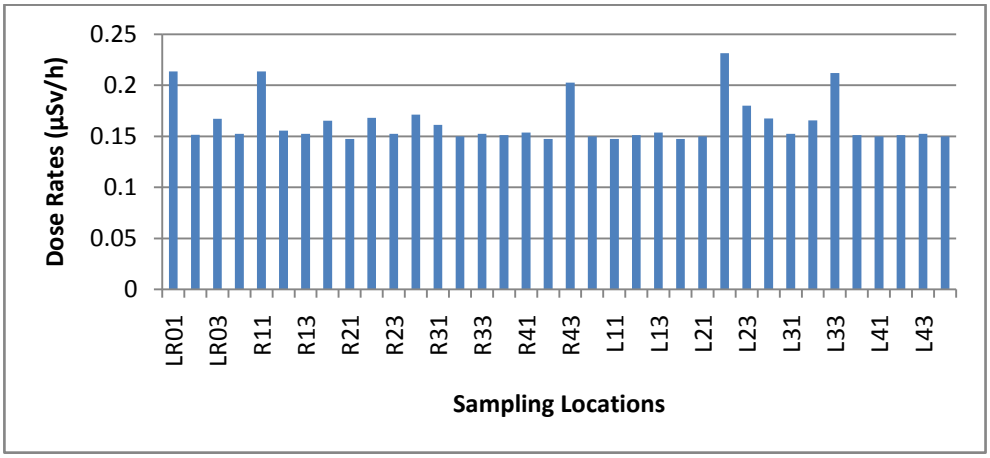


Figure 4.21 In-Situ Dose rate Measurements (µSv/h) in the mine site

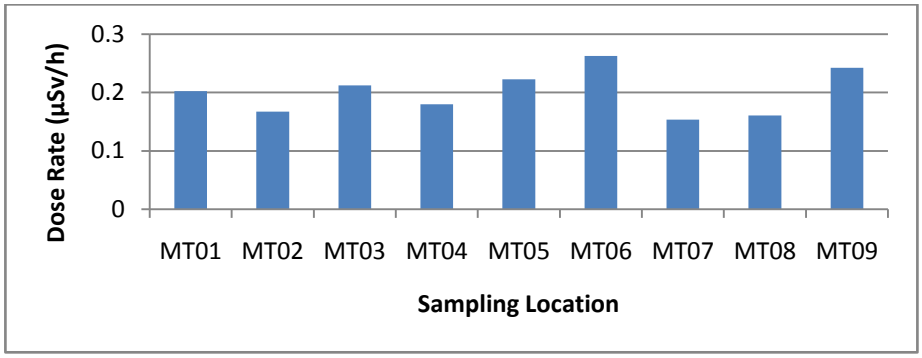


Figure 4.22 In-Situ Dose rate Measurements (µSv/h) in the mill tailing site

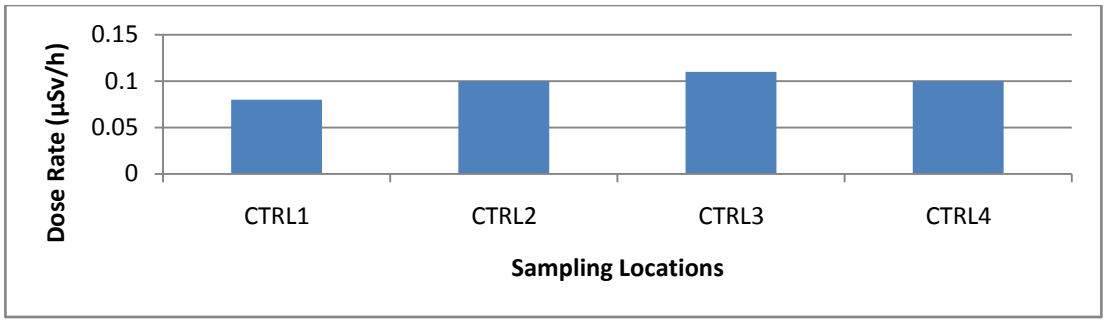


Figure 4.23 In-Situ Dose rate Measurements (µSv/h) in the control area

Table 4.15 In-Situ Dose rate Measurements

	In-situ dose Rate Measurements (µSv/h)		
Location	**Mean**	**Range**	**Standard Deviation**
Mine Site	0.16	0.15-0.23	0.02
Mill Tailing Site	0.20	0.15-0.26	0.04
Control Area	0.10	0.08-0.11	0.01

Table 4.16 Annual Effective Dose rate

Location	**Annual Effective Dose rate Measurements (Sv/y)**			
	Mean	**Minimum**	**Maximum**	**Standard Deviation**
Mine Site	0.0014	0.0013	0.0020	0.0002
Mill Tailing Site	0.0018	0.0013	0.0023	0.0003
Control Area	0.0009	0.0007	0.0010	0.0001

Table 4.17 Univariate descriptive statistics of the Annual Effective Dose rate Measurements in mSv/y

	Annual Effective Dose rate Measurements (mSv/y)			
Location	**Mean**	**Minimum**	**Maximum**	**Standard Deviation**
Mine Site	1.44	1.29	2.03	0.20
Mill Tailing Site	1.76	1.35	2.30	0.33
Control Area	0.86	0.70	0.96	0.11

4.2.2 Activity concentrations of ^{226}Ra, ^{232}Th and ^{40}K

The mean activity concentrations of ^{226}Ra, ^{232}Th and ^{40}K from each sample location at the mine site, mill tailing site and control area, respectively, are presented in Table 1. It can be observed that the concentrations of radionuclides in the mill tailing site are strongly enhanced with respect to the soils collected from the mine site in the study area, as well as the control soils. Higher activity concentrations of ^{232}Th and ^{40}K were observed in all the mill tailing samples. The mill tailing samples were observed to contain high radionuclide concentrations, this may be as a result of the mined minerals in mill tailing site, which are characterized by a significant radioactive concentrations for ^{232}Th and ^{40}K, indicating the dimensions of radionuclide contamination of the site as a result of the mining activities in the area. One possible explanation for the high activity concentrations in the mill tailing site, is that during gold mining and ore processing, the radionuclides are transferred into a mill tailing deposits; as such they can even get enriched in the settled suspension. In contrast, the activity concentrations of the soil collected from the mine site were also observed to be high. Also, it should be noted that, compared to the control soils, the activity concentrations were observed to be lower, this observation could be attributed either to the geological factors or as a result the impact of the mining activity in the area.

The activity concentrations of ^{226}Ra, ^{232}Th and ^{40}K of the soil and mill tailing samples collected at the mine site and mill tailing site are shown in Fig 4.21 and 4.22 respectively. It can be observed that the highest concentrations (Bq/kg) of ^{226}Ra, ^{232}Th and ^{40}K was found to be 43.94 ± 3.02, 38.61 ± 2.06 and 304.17 ± 23.03 in the mine site and 45.39 ± 5.07, 72.41 ± 7.18 and 472.16 ± 35.26 in the mill tailing site, respectively. The lowest concentrations (Bq/kg) was found to be 10.43 ± 0.04, 9.79 ± 0.83 and 42.63 ± 1.07 in the

mine site and 20.54 ± 0.48, 44.63 ± 0.83 and 280.74 ± 7.08 in the mill tailing site, respectively. It is apparent that ^{40}K exhibited the highest activity concentrations for all measured radionuclides in all of the soil and mill tailing samples measured. It can also be observed that the activity concentrations of the mill tailing sites were higher than that of the mine site; this variation can be attributed to the geology and geochemical composition of the mill tailing site. The high values recorded at the mill tailing site are due to the acid intrusive geological feature found in the areas. These areas are igneous acidic and extensively intruded by granitic rocks. The granite is relatively rich in radioactive minerals. The rocks and soil types found in the study area are igneous acidic, which are extensively intruded by granitic rocks and granite is traditionally known to contain higher levels of radioactive minerals (UNSCEAR, 2000).

The results of this study have been compared with the world mean activity concentrations in soil. Table 4.18 compares the results obtained in this work with the world mean values as presented by UNSCEAR (2000). The worldwide activity concentrations of Ra-226, Th-232 and K-40 were reported to be 17-60, 11-64 and 140-850 Bq/kg with the mean concentrations of 35, 30 and 400 Bq/kg, respectively. The obtained results showed that the activity concentrations of ^{226}Rain all the soil and mill tailing samples ranged from 10.43 ± 0.04 to 76.67 ± 5.83 and 20.54 ± 0.48 to 72.39 ± 2.88 respectively, thus, fall within the worldwide range, although the mean concentration of Ra-226 in the mine site were observed to be higher than the worldwide mean value. However, the overall mean concentration (Bq/kg) of ^{226}Ra was observed to be lower than the worldwide mean value. The mean activity concentrations of ^{232}Th in the mill tailing samples showed slightly higher value than the worldwide mean value, which could be due to the presence of

abundant radioactive thorium minerals such as monazite, zircon and thorianite (Okeyode & Akanni, 2009). The ranges of the activity concentrations (Bq/kg) of ^{232}Th and ^{40}K vary from 9.79 ± 0.83 to 59.80 ± 2.60and 42.63 ± 1.07 to 714.32 ± 16.25 at the mine site, respectively and 44.63 ± 0.83 to 94.92 ± 2.91 and 280.74 ± 7.08 to 590.44 ± 16.24 at the mill tailing site respectively. The activity concentrations are above the upper range due to the high concentrations values observed in the mill tailing samples. The mean activity concentrations (Bq/kg) of ^{232}Th and ^{40}K are 38.61 ± 2.06 and 304.17 ± 23.03 of the soil samples, are comparable with the worldwide mean value, while the mean activity concentrations (Bq/kg) of ^{232}Th and ^{40}K are 72.41 ± 7.18 and 472.16 ± 35.26 of the mill tailing samples, are above the worldwide mean value. The high concentrations of ^{232}Th and ^{40}K in the mill tailing samples could be attributed either to geological factors or variable dumping of the mill tailing deposit.

Table 4.18 Mean activity concentrations of Ra-226, Th-232 and K-40 in the mine site, mill tailing site and control area with the world wide mean

	Mean Activity concentration(Bq/Kg)		
Sample location	**Ra-226**	**Th-232**	**K-40**
Mine Site	43.94±3.02	38.61±2.06	304.17±23.03
Mill Tailing Site	45.39±5.07	72.41±7.18	472.16±35.26
Control Area	12.25±0.48	25.42±1.07	157.84±5.71
Worldwide Range	**17-60**	**11-64**	**140-850**
Worldwide Mean	**35**	**30**	**400**

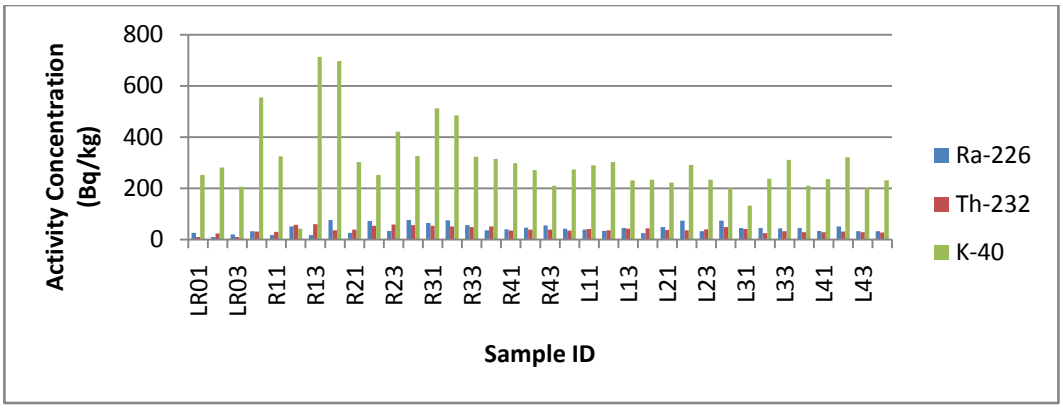


Figure 4.21 Mean activity concentrations of Ra-226, Th-232, and K-40 in the mine site

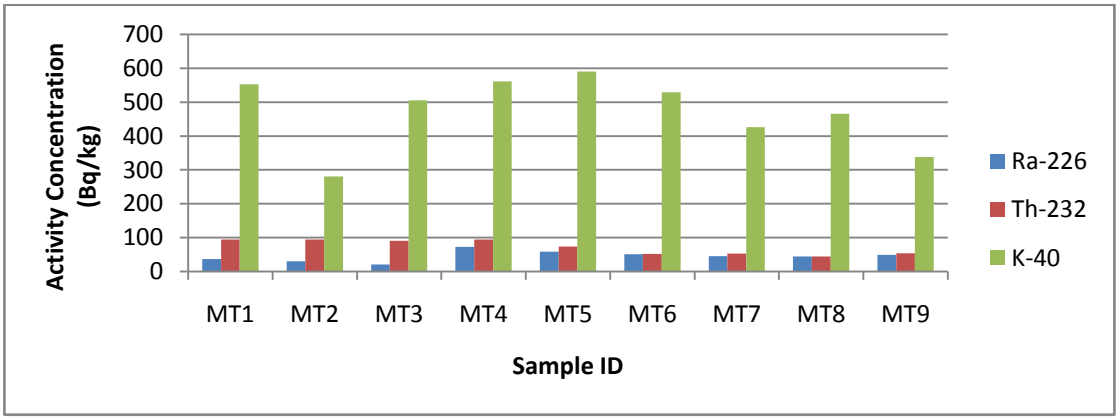


Figure 4.22 Mean activity concentrations of Ra-226, Th-232 and K-40 in the mill tailing site

4.2.3 Assessment of Radiological hazard

The radiation hazard parameters were computed as shown in Table 4.19. The radiological hazard parameters in this study was calculated from the measured activity concentrations of the three primordial radionuclides in the soil and mill tailing samples, using the relations described in section 2. "The estimated absorbed dose rates (nGy/h) based on soil and mill tailing radioactivity ranged from 11.78 ± 1.55 to 86.38 ± 4.87 with a mean value of 53.94 ± 2.84 at the mine site and ranged from 66.86 ± 1.55 to 114.18 ± 3.32, with a mean value of 84.39 ± 5.21 at the mill tailing site, respectively". ^{40}K is the main contributor to the

absorbed dose rate in most samples in this work. The mean absorbed dose rate of the control area is 27.60 ± 1.11 which is less than that of both the mine and mill tailing sites, respectively. Comparing the results obtained with the worldwide values, the average mean value of absorbed dose rate from all the samples in this work are higher than the worldwide mean value.

The annual effective dose equivalent from the outdoor terrestrial gamma radiation was estimated by taking into account the conversion coefficients from the absorbed dose in air to effective dose and outdoor occupancy factor. The effective dose (mSv/y) for the different locations of soil and mill tailing samples in the study varied from 0.01±0.004 to 0.11±0.005, with the mean value of 0.07±0.003 in the mine site and also ranged from 0.08±0.002 to 0.14±0.004, with the mean value of 0.10±0.002, which is higher compared with the worldwide effective dose of 0.07mSv/y by UNSCEAR, (2000). The acceptable annual effective dose for members of the public without constraint should be 1.0 mSv/y for safety purposes ICRP, (1990). As such the radiological constraints for an adequate protection of potential users of 0.5mSv/y in which the results obtained from this study were compared with as recommended by EC report, (1999).

The radium equivalent activity (Ra_{eq}), the external hazard index (H_{ex}), gamma index (I) and excess lifetime cancer risk (ELCR) were also evaluated and presented in Table 4.19. The results of the radium equivalent activity (Bq/kg) of the soil and mill tailing samples ranged from 51.51 ± 3.23 to 186.35 ± 10.66, with a mean of 122.57 ± 5.71 in the mine site and 144.01 ± 6.23 to 251.35 ± 17.36, with a mean of 185.29 ± 11.73 in the mill tailing site, respectively. From the results obtained from this study, it can be observed that the Ra_{eq} values were lower than the accepted safety limit value of 370Bq/kg as recommended by

the Organization for Economic Cooperation and development (OECD, 1979, UNSCEAR, 1999, Belivermis, *et al.,* 2009).The radium equivalent of 370 Bq/kg corresponds to the dose limit of 1 mSv for the general population (Jibiril, *et al*., 2009). As such the use of the soil and mill tailings as raw material for building for human inhibition in the area may not pose health risk to the population. The results of the external hazard index (H_{ex}) of the soil and mill tailing samples ranged from 0.14 ± 0.01 to 0.50 ± 0.03 with the mean value of 0.33 ± 0.02, in the mine site and ranged from 0.39 ± 0.01 to 0.68 ± 0.04 with a mean value of 0.50 ± 0.03, in the mill tailing site, respectively. The obtained results of the external hazard index (H_{ex}) shows that the soil and mill tailing samples are below the limit of unity, however, the radiation dose is within the permissible limit of 1 mSv/y as recommended by (ICRP, 1990). The results of the gamma index (I) of the soil and mill tailing samples ranged from 0.19 ± 0.01 to 0.67 ± 0.04 with the mean value of 0.43 ± 0.02 in the mine site and ranged from 0.53 ± 0.01 to 0.90 ± 0.04, with a mean value of 0.67 ± 0.04, in the mill tailing site, respectively. The results of the excess lifetime cancer risk (ELCR) of the soil and mill tailing samples ranged from (0.05 ± 0.01) × 10^{-3} to (0.37 ± 0.02) × 10^{-3} with the mean value of (0.23 ± 0.01) × 10^{-3} in the mine site and ranged from (0.29 ± 0.01) × 10^{-3} to (0.49 ± 0.04) × 10^{-3} with a mean value of (0.36 ± 0.02) × 10^{-3}, in the mill tailing site, respectively.

Table 4.19 The mean absorbed dose rate (D), annual effective dose equivalent (AEDE), radium equivalent activity (Ra_{eq}), external hazard index (H_{ex}), gamma index (I) and excess lifetime cancer risk (ELCR) obtained from all the soil and mill tailing samples with that of control and worldwide

Sample	**D(nGy/h)**	**AEDE(mSv/y)**	**Ra_{eq}(Bq/kg)**	**H_{ex}**	**I**	**ELCR ×10^{-3}**
Mine site	53.94±2.84	0.07±0.003	122.65±5.71	0.33±0.01	0.43±0.02	0.23±0.01
Mill tailing site	84.39±5.21	0.10±0.006	185.29±11.73	0.50±0.03	0.67±0.04	0.36±0.02
Control area	27.60±1.11	0.03±0.001	60.76±2.45	0.16±0.01	0.22±0.01	0.12±0.01
Worldwide mean	**59**	**0.07**	**< 370**	**≤ 1**	**≤1**	**0.29**

CHAPTER 5: CONCLUSION AND RECOMMENDATION

5.1 Research Summary

The study was carried out to determine the heavy metal accumulation and its pollution status using Contamination Factor (CF), Pollution load Index (PLI), Degree of Contamination (DC) Index, Geo-accumulation Index (I-geo), Hazard Quotient (HQ) and Potential Ecological Risk index (ERF). Soil and mill tailing samples were analyzed using Flame Atomic Absorption Spectroscopy (FAAS) to obtain the composition and concentration. Angwan Maigero Gold mining site is located between (Latitude $10^{o}1^{1}29^{o}$N to $10^{o}1^{1}$ 30N and longtitude $6^{o}28^{1}30^{''}$ E to $6^{o}28^{1}$31"E) in Niger State, Nigeria. Samples were collected from the mine and mill tailing sites, respectively. The study area was observed to be contaminated with Pb, Fe, Mn, Cu, Zn, As, Cr, Co, Ni, and Cd. The mean concentrations (ppm) of the studied heavy metals in Mine Site was observed to be 319.77, 1707.63, 181.61, 90.95, 37.41, 17.99, 93.19, 47.94, 22.28 and 3.23 respectively, Also the mean concentrations (ppm) of heavy metals of interest in the mill tailing site was observed to be 846.87, 1914.99, 177.70, 739.33, 100.41, 43.52, 163.28, 53.48, 45.59, 5.83. The Highest concentration corresponds to Fe and the lowest corresponds to Cd. The increasing trend was in the order: Cd <As <Ni <Zn <Co<Cu <Cr <Mn <Pb <Fe. The concentrations of all the elements in the mill tailing site were observed to be higher than that obtained in the mine site with greater variation in relative abundance of most of the metals. These variations can be attributed to the variable dumping of mill tailing deposit. The mean concentrations (ppm) of the heavy metals in the control area were 21.40, 1331.00, 68.10, 35.72, 11.91, 5.73, 47.85, 18.17, 14.26, and 0.23 respectively. Statistical significant difference was observed between the mean of heavy metal

concentration in the mine and mill tailing sites, respectively which suggested effect of anthropogenic inputs. The contamination factors, pollution load index and degree of contamination indices were computed. The pollution load index revealed that the mine and mill tailing sites was polluted with all the observed heavy toxic metal. (Mean PLI=2.712 and 5.784 respectively) and the degree of contamination index indicated that the mine and mill tailing sites was highly contaminated (Mean DC=47.84 and 115.09 respectively). The extremely high contents of the heavy metals generally recorded at the study area are alarming in terms of environmental pollution. Therefore, the inhabitants and the numerous workers, who reside and work at the environment, are at serious risk of heavy metal toxicity and awareness needs to be created. As such, immediate actions needs be taken up by the government and relevant authorities against the toxicological risk involved in the mining activities of the area and a strategic planning should be devised for the protection of human health and environment. Similarly, the soil and mill tailing samples were analyzed for natural radioactivity using high resolution gamma ray spectrometry with NaI(TI) detector for the activity concentration for K-40, Ra-226 and Th-232. Similar study was also carried out at about 3km away from the study area to serve as the control.

5.2 Conclusion

The study was carried out to ascertain the levels of environmental impacts caused by gold mining and ore processing activities at the study area, so as to give an insight to the possible level of radiological and toxicological problems that could be poised by the mining activities in the area. Flame atomic absorption spectroscopy (FAAS) was used to obtain the composition and concentration of heavy metal pollutants: Pb, As, Mn, Fe, Cr, Ni, Co, Cd, Cu, and Zn from the study area. The obtained concentrations (ppm) were used

in the computations of heavy metal hazard indices using geo-accumulation index, ecological risk index, pollution load index, degree of contamination index, hazard quotient and contamination factors. On the basis of these indices, it has been established that the soil and mill tailing samples of study area has been highly affected by accumulation of toxic and heavy metals. These heavy metals are thus terribly alarming and can cause environmental problems in the ecosystem of the area. As such, it is suggested that these sites should be monitored for health related problems so as to minimize the extent of the accumulated pollution level in the area. Similarly, the activity concentrations for ^{226}Ra, ^{232}Th and ^{40}K were computed. The mean activity concentration for ^{226}Ra lies within the worldwide mean, but for ^{232}Th and ^{40}K the mean activity concentrations were higher than the worldwide mean. The obtained results indicated that the elevated activity concentrations of ^{232}Th and ^{40}K recorded in the study area may be as a result of the variation in geological structure or the invariable disposal of the mill tailing waste deposited at the mill tailing site due to the impact of the mining activity in the area. The radiation hazard indices were estimated for soil and mill tailing samples collected from the study area. The results shows that all the hazard indices (H_{ex} and I) and excess lifetime cancer risk calculated have values less than unity. In conformity with the values of the hazard indices, the radium equivalent activity has a mean value of 122.57Bq/kg and 185.29 Bq/kg in the mine and mill tailing sites, respectively which is lower than the maximum permissible value of 370Bq/kg which implies that members of the public will be subjected to an effective dose lower than 1mSv.Thus, the values of the radiological hazard indices are within the safety limits. It can be observed that the artisanal gold mining activities in the studied area is associated with immediate and long term health risks to the exposed

populations, therefore gold mining in the area is not safe toxicologically, but should be controlled and monitored regularly in order to avoid any radiological impact to the population.

5.3 Recommendation

Assessments of heavy metal sources as well as contamination status are important environmental scientific issues. In view of the observed events relative to the mine site and mill tailing site in terms of variation in Lead as well as the resultant hazards, further research work need to be conducted to strongly establish the mobility and solubility strength of the stated element.

Similar work should be carried out on the Acid mine drainage that must have resulted over length of time and corresponding possible uptake of this metals by crops planted in the area both during the dry season by irrigation as well as during the raining season and consequently compare the result to similar output from regions outside the mine and mill tailing sites so as to calculate the transfer factor.

Wider area study needs to be carried out so as to provide more accurate data. In similar vein, further works need to be carried out to study the effects of mining in the wells of the neighboring villages to the mining sites.

Health screening should be carried out periodically on the local miners and inhabitants to check for some symptoms of heavy metals inhalation.

Government should sensitize the people on the dangers of farming on the land and also the ground water within or close to the mining sites should be monitored

Residential houses should be far away from the mining site and farming on nearby land should be prohibited so that members of the community who are less protected will not be exposed to mining hazards unduly.

The mine workers should adhere strictly to the use of protective facemask to prevent undue inhalation of toxic elements that may be present in the dust during mining operation so as to reduce radiation exposure and contamination.

In view of the possibility of radiological and toxicological risks involved in the mining activities of the area observed in this research work, similar work should be carried out on all the mines in Nigeria especially where little or none has been done.

REFERENCES

Adewole, A. T. (2009). Waste Management Towards Sustainable Development in Nigeria: A case Study of Lagos State. *International Non Governmental Organisation Journal, 4*, 173-179.

Agency for Toxic Substances and Disease Registry. (2007). *Toxicology Profile for Lead.* Public Health Services, U S Department of Health and Human services.

Ahmad, K. K., Zi, A., & Yasmin, S. (2014). Assement of Heavy Metal and Metalloid Levels in Spinach (Spinacia oleracae) grown in waste soil of Sarghoda, Pakistan. *Pakistan Journal of Botany, 46* (5), 1805-1810.

Ajayi, I. O., & Ajayi, I. R. (1999). A Survey of Environmental Gamma Radiation Levels of Rocks in Some Areas of Ekiti and Ondo States, Southwestern Nigeria. *Nigerian Journal of Physics, 232-210.*

Ajayi, T. R., Torto, N., Tchokossa, P., & Akinlua, A. (2009). Natural radioactivity and trace metals in crude oils: implication for health. *Environmental Geochemistry and Health, 31* (1), 61-69.

Akpan, I. O., & William, E. S. (2014). Assessment of Elemental Concentration of Roadside Soil in relation to Traffic density in Calabar, Nigeria. *International Journal of scientific and technology Research*. Volume 3, Issue 9, September 2014 ISSN 2277 – 8616.

Amasi, I. A., Mtei, K. M., Ijumba, J. N., Jodlowski, P., & Dinh, N. C. (2014). Natural Radioactivity in Tanzania cements and their raw Materials.

Arogunjo, A. M. (2009). Determination of activity concentration in soils and mineral sands from Bisichi, Jos, Nigeria. *Journal of Environmental Radioactivity, 100*, 232-240.

Attia, T. E., Shendi, E. H., & Shehata, M. A. (2014). Assessment of natural and artificial Radioactivity levels of Radiation hazards and their relation to heavy metals in the industrial area of Port Said City, Egypt. *Journal Earth Science and Climate Change, 5* (4), 188.

Audu, D. A., Zakari, Y. I., & Ibeanu, I. G. (2013). Radiological effects of iron ore mining and processing at Itakpe, Kogi state. *Journal of Natural Sciences Research, 3* (15).

Augustine, A. K., Ayo, I., Babalola, F. O., & Alabi. (2014). Assessments of natural radioactivity and determination of heavy metals in soil around industrial dumpsites in Sango - Ota, Ogun state, Nigeria. *Journal of Medical Physics, 39* (2), 106-111.

Augustine, K. A. ,& Aku, M. O. (2013). Gamma Radiation Measurement and Dose rates in Cements used in Nigeria. *Nigerian Journal of Physics,* 24: 91-95.

Avwiri, G. O., & Esi, E. O. (2014). Evaluation of background ionization radiation level in selected dumpsites in Delta State Nigeria. *International Journal of Science and Technology, 3* (10), 663-673.

Bafandeh, H. R. (2004). Alternative Approaches to Internal Ionizing Radiation Protection. Massachusetts, USA.

Bako, S. P., Odowo, J. E., & Ezealor, A. U. (2008). Plant Diversity in Relation To Anthropogenic Trade Metals in Soils of Selected Sites in Nigeria's Guinea Savanna. *International Journal Of environment and Pollution, 33* (2/3), 185-194.

Bashir, M., Zakari, Y. I., Ibeanu, I. G., & Sadiq, U. (2013). Assessment of radiological risk in Flooded soil samples of Kudenda, Kaduna state Nigeria. *International Journal of Engineering science invention, 2* (10), 69-74.

Beliverimis, M., Kilic, O., Coytuk, Y., & Topcough, S. (2009). The effects of physiochemical properties of gamma emitting natural radionuclides level in the soil profile of Istanbul. *Environmental Monitoring and Assessment, 163* (1-4), 15-26.

Bertrand, V.J., Monroy, M.G., & Lawrence. (2000). Weathering Characteristics of Cemented paste backfill: Mineralogy and Solid phase Chemistry. Proceedings of the 5th International Conference on Acid Rock Drainage (ICARD) pp: 863-876.

Bertin, E. P. (1985). *Introduction to X-Ray Spectrometric Analysis.* NY: Premium Press.

Bertka, J., & Matthew, P. J. (1995). Natural radioactivity of Australine Bulding Materials Industrial Waste and by- products. *Health Physics, 48*, 87-95.

Boadu, T. M. (2014). Heavy Metals Contaminations of soil and water at Agbogbloshie Scrap Market, Accra, Unpublished Thesis. Kwame Nkrumah University of Science and Technology. M.Sc Thesis May, 2014.

Brouwer, P. (2003). *Theory of XRF.* Netherlands: PAN Analytical B. V., ISBN: 90-9016758-7.

Canfield, R.L., Henderson, R. J., Cory-Slechta, D. A., Cox, C., Jusko, T.A., Lanphear, B. P. (2003). Intellectual Impairment in Children with Blood, lead concentration below 10ug per deciliter N. EnglJ.med 48 (16): 1577-1526.

CDC. (2002). Managing Elevated Blood Lead Levels Among Young Children. Recommendations from the Advisory Committee on Childhood Lead Poisoning Prevention, Centers for Diseases Control and Prevention. available: http://www.cdc.gov/nceh/casemanagement/casemanagemain.htm.

CDC.(2010). Notes from the Field: Outbreak of Acute lead poisoning Among Children Aged ≤5 years,Centers for Diseases Control and Prevention. Zamfara, Nigeria. Pp.59(27):846.

Cember, H. (1996).*Introduction to health physics*. Mc Graw Hill. Health Professional. Davis New York.

Charke, E. G., & Clarke, L. M. (1967). *Vertinary Toxicology* (3rd ed.). United States of America: Williams and Wilkins Company.

Diabi, H. M., Nouh, A., Hamdy, S.A., & El-Fiki. (2008). Evaluation of Natural Radioactivity in a cultivated area around a Fertilizer Factory. *Journal of Nuclear and Radiation Physics*, vol. 3, No.1, pp53-62.

Draghici, C., Coman, G., Jelescu, C., Dima, C., & Chirila, E. (2010). Heavy metals determination in environmental and biological Samples. *Environmental Heavy Metal Pollution and Effects On Child Mental Development- Risk Assessment and Prevention Strategies.* Sofia, Bulgaria: NATO Advanced Research Workshop.

Dooyema, C.A., Neri, A., Lo, Y.C., Durant, J.,Dargon, P.I., &Swathout, Y. (2012). Outbreak of Fatal Childhood Lead Poisoning related to Artisanal Gold Mining in North Western Nigeria, 2010 *Environmental Health Prospect* 120: 601-607.

EC. (1999). *Radiation Protection Principles Concerning the Natural Radioactivity in Building Materials Environmental Nuclear Safety and Civil Protection*, Directorate General, European Commission.

Elaigwu, S. E., Ajibola, V. O., & Folaranmi, F. M. (2007). Studies on the impact of municipal waste dumps on surrounding soil and air quality of two cities in northern Nigeria. *Journal of Applied Science, 7* (3), 421-425.

Eisler, R. (2004). Arsenic Hazards to Humans, Plants and Animals for Gold Mining" Rev. Environ Contamination Toxicology.

Evalyn, I.A. (2004). *Background Concentration of Trace elements in Soils and Rocks of Georgia piedmont.* A Thesis Submitted to the Graduate Faculty of the University of Georgia in Partial Fulfillment of the Degree of Master of Science. Athens, Georgia.

Fasae, K. P. (2013). Natural radioactivity in locally produced building materials in Ekiti state, South Western Nigeria. *Civil and Environmental Research, 3* (11).

Faweya, E.B., &Babalola, A.I. (2010). Radiological Safety Assessment and Occurrence of Heavy Metals in Soil from Designated Waste Dumpsites used for Building and Composting in South-Western Nigeria. Arabs.s. sci.eng. 5:89-101.

Faweya E. B., & Oniya E. O. (2012). Radiological Safety Assessment and Physico-chemical Characterization of soil mixed with mine tailings used as building materials from Oke – Kusa mining sites in Ijero, Nigeria. *Nature and science*2012.10(5):64-71 ISSN: 1545 – 0740 http://www.sciencepub.net/nature.

Ferner, D.J. (2001). Toxicity of Heavy Metals, eMed. J.2 (5):1.

Ferreira, S. E., Zhang, C., Serrano, P. L., Patinha, C., & Reis, P. (2004). Hazard Assessment of Arsenic and Lead in Soils of Castromil Gold Mining Area. *Applied Geochemistry*, *19* (6), 887-898.

Fidelis, N. (2015, May, 15). Lead poisoning Outbreak. The Sun. Retrieved from http:// www.sunnews.com.ng/sun/index.

Fuad, A. A. (2008). *Measurements of Naturally Occurring Radioactive Materials (NORMs) in Environmental Sample.* MSc Thesis, Department of Physics, Faculty of Engineering and Physical Science, University of Surrey.

Garbarino, J. R., Hayes, H., Roth. D., Antweider, R., Brinton, J. I. & Taylor, H. (1995). Contaminants in the Mississippi River, U.S. Geological Survey Circular 1133, Virginia, USA. (www.pubs.usgs.gov/circ/circ1331).

Girigisu, S. (2012). *Assessment of radiological levels and elemental concentrations in soils from Awwal and Bagega gold mining sites in Kebbi and Zamfara states, Nigeria.* MSc. Thesis, Ahmadu Bello University, Physics Department, Zaria.

Girigisu, S., Ibeanu, I. G., Adeyemo, D. J., Onoja, R. A., Bappah, I. A., & Okoh, S. (2014). Assessment of Radiological Levels in Soils from Awwal Artisanal Gold Mining Exercises in Kebbi State, Nigeria. *Research Journal of Applied Sciences, Engineering and Technology*, *7* (14), 2899-2904.

Gordon, G. (2008). *Practical Gamma Ray Spectrometry* (2nd ed.). New Jersey, US: John Wiley and Sons.

Grolier Incorporated. (1985). *Encyclopedia America* (Vol. 1). Danbury, US.

Hakanson, L. (1980). Ecological Risk Index For Aquatic Pollution Control, A Sedimentological Approach Water Resources. *Journal of the International Water Association, 14* (8), 975-1001.

Hall, E. J., & Giacca, A. J. (2006). *Radiobiology for the Radiologist.* Philadelphia, US: Williams and Wilkins.

Harikumar, P. S., Nasir, U. P., & MajeebuRahma, M. P. (2009). Distribution Of Heavy Metals In The Core Sediments Of A Tropical Wetland System. *International Journal of Environmental Science and Technology, 6* (2), 225-232.

Hawkes, J.S. (1997). Heavy Metals in the Environment. *Journal of Chemical Education* 74 (11): 1374

Hayumbu, P., Zaman, M. B., Lubaba, N. C., Munsanje, S. S., Muleya, D. (1995). Natural Radioactivity in Zambian building materials collected from Lusaka. *Journal of Radioanalytical Nuclear Chemistry* Letter 11 , 299.

Health Canada. (2009, May 13). *Draft Guidance on Heavy metal impurities in cosmetics.* Retrieved June 4, 2015, from ConsumerProductSafety.org: cafeoa/documents/draftguidanceonheavymetalsimpuritiesincosmetics.htm

Hostyneck, J. J. (1998). Toxic Potentials from Metals Absorbed through the Skin. *Cosmetics and Toiletries Magazine, 113*, 33-42.

IAEA. (1989). Measurement of Radionuclides in Food and Environment, Aguide book, Technical Report Series No. 295, IAEA, Vienna

Ibeanu, I. G. (1999). *Assessment of radiological Effects of Tin Mining Activities in Jos and its Environs.* PhD Thesis, Ahmadu Bello University, Physics Department, Zaria.

ICRP. (1991). Recommendations of the International commission on Radiological protection. Annals of the ICRP 21 (1-3). ICRP Publication 60. Pergamon Press, Oxford.

Innocent, A. J., John, O., Ali, H., Onimisi, M. Y., & Jonah, S. A. (2014). Radiological safety assesment of some mine sites at Gusau and environs, Nigeria. *Advancement of Scientific and Engineering Research, 2* (2), 23-28.

Institute of Environmental Conservation & Research. (2000). Position Paper against Mining in Rapu-Rapu, Published by INECAR, Ateneo de Naga University, Philippines (www.adnu.edu.ph/institutes/inecar/pospaper1.asp).

Iyaka, Y. A., & Kakulu, S. E. (2011). Heavy Metal Concentrations In Top Agricultural Soils Around Ceramic And Pharmaceutical Industrials Sites In Niger State, Nigeria. *Research Journal Of Environmental And Earth Science, 4* (2), 171-176.

Jibiri, N. N., Farai, I. P., & Alausa, S. K. (2007). Estimation of Annual Effective Dose due to Natural Radioactive Elements in Ingestion of Foodstuffs in Tin Mining Area of Jos-Plateau, Nigeria. *Journal of Environmental Radioactivity*, 94, 31-40.

Jibiri, N. N., & Biere, P. E. (2011). Activity concentrations of 226Ra, 228Th, and 40K in different food crops from a high background radiation area in Bitsichi, Jos Plateau, Nigeria. *Iranian Journal of Radiation Research, 8* (4), 201-206.

Jibiril, N. N. (2009). Radiological Hazard Indices due to Activity Concentration of Natural Radionuclides in Farm Soils from Two High Background Radioation Areas in Nigeria. *International Journal of Low Radiation, 6* (2), 79-95.

Kalagbor, I. A., Barisere, V., Barivule, G., Barile, S., & Bassey, C. (2014). Investigation of the presence of some heavy metals in four edible vegetables Bitter leaf(Vernonia amygdalina), scent leaf (Ocimum gratissimum), water leaf (Talinum triangulare) and fluted pumpkin (Telfairia occidentalis) from cottage farm in port Harcourt. *Journal of Environmental and health sciences, 6* (1), 18-24.

Kodom. K., Preko. K., & Boamah. D. (2008). X-Ray Fluorescence (XRF) Analysis of Soil Heavy Metal Pollution from an Industrial Area- A case study of "Suame – Magazine" in Kumasi, Ghana.

Kowsar, M. D., Shyamal, R. C., Adulkhaer, M. R., Arunkumar, D. M., Mantazul, I. S., & Suza, M. D. (2012). Measurement of Physicochemical Parameters and Determination of the Level of Radiological Threat to the Population Associated With the Karnaphuli River Sediment containing municipal and industrial wastes of Chittagong city in Bangladesh. *Oxford Journals, Radiation Protection Dosimetry*, Issue: July 2012 165 (1-4).

Kilic, E. (2009). Heavy Metal Pollution in water. Health effects of heavy metal Hacettepe University. Cag-Kimya, Turkey.

Krieger, R. (1981). Radioactivity of construction materials. *Betonwerk Fertigteil Technology, 47*, 468-473.

Lee, E. M., Menezes, & Flinch, E. C. (2004). National Radioactivity in Building Materials in the Republic of Ireland. *Health Physics, 86*, 378-388.

Lee, G. F., & Lee, J. A. (2005). Municipal Solid Waste landfills – Water Quality Issues. *Water Encyclopedia, Water Quality and Resource Development* , 163-169.

Lenntech Water Treatment and Air Purification. (2004). Water Treatment Published by *Lenntech, Rotter dam Sewage*, Netherlands. (www.excelwater.com/thp/filters/Water-Purification.htm.

Lilisa, S. E., Jolanki, R., Hakala, E., & Kanarv, L. (2000). Metals and Arsenic in Eye Shadows. *Contact Dermatitis, 42* (3), 5-10.

Liu, O. M., Salmijah, S., Ismail, B. S., & Aminah, S. (2007). The Impact Of Traffic Causing Lead Exposure To Malaysian School Children. *Global Journal of Environmental Research* , 43-48.

Lokeshappa, B., Kandarp, S., Vivek, T., & Anil, K. D. (2012). Assessment of Toxic Metals in Agricultural Produce. *Journal of Scientific & Academic Publishing*, Food and Public Health 2012 DOI: 10.5923/j.fph.20120201. 05.

Marfo, B. T. (2014). *Heavy Metal Contaminations of Soil and Water at Agbogbloshie Scrap Market, Accra.* MSc Thesis, Kwame Nkrumah University of Science and Technology, Kumasi.

Martinez, T. J., Lartigue, P., Avilaperez, M., Navarrete, G., Zarazau, C., Lopez, L., et al. (2001). Metallic Pollutants in Mexico Valley. *16th International Conference On Application of Accelerators in Research and Industry, At American Institute of Physics*, (pp. 512-515). NY.

Mathews-Amene, O. G., & Kukulu, S. (2013). Investigation of Heavy Metals levels in Roadsides Agricultural Soil And Plant Samples In Adogo, Nigeria. *Academic Journal Of Environmental Science, 1* (2), 31-35.

McCluggage, D. (1991). Heavy Metal Poisoning, NCS Magazine. Published by the Bird Hospital, Co. USA. (www.cockatiels.org/articles/Diseases/metals.html).

Meyer, P.A., Brown, M.J., & Falk, H. (2008). *Global Approach to Reducing Lead Exposure and Poisoning*, Mutation res 659 (1-2), 166-175.

Montes, M. I., Taylor, M. A., Mercader, R.C., Sinves, F. R., & Desimoni, J. (2010). Hyperfine and Radiological Characterization of Soils of the Province of Buenos aires, Argentina. International Conference on the Applications of the Mossbauer Effect (ICAME). *Journal of Physics, Conference Series* 217 (2010) 012058.

Mousa, I. (1997). *Assessing the Impacts from Sulphide Mine Waste on Surface Water Resources for Environmmental Management; The Case of Mathiatis Sha Mining District Republic of Cyprus.* Enschede, Netherlands: International Institute for Geo-Information Science and Earth Observation.

Mull, E. J. (2005). *Approaches Towards Sustainable Urban Solid Waste Management:Sahakaranagar Layout.* MSc Thesis, Lund University, Environmental Science, Lund.

Muller, G. (1969). Index of Geo-accumulation in sediments of the Rhine River. *Geological Journal, 2* (3), 108-118.

NAS/NRC. (1999).*Arsenic in drinking water*, National Academy of Sciences/National Research Council Washington, D.C, pp. 251 – 257.

Nadal, M., Schuhmacher, M., & Domingo, J. L. (2004). Metal pollution of soils and vegetation in an area with petrochemical industry. *Science of the Total Environmrnt, 321*, 59-69.

NBIRR. (2003). Nigeria Basic Ionizing Radiation Regulations.

Needleman H, (2004). Lead Poisoning Annul Rev Med 55 (1) 209-222.

Nguelem, E. J., Darko, E. O., Nduntchueng, M. M., Schandorf, C., Akiti, T. T., & Muhulo, A. P. (2013). Assessment of Natural Radioactivity level in ground water from selected areas in Accra metropolis. *Journal of Environmental and Health Sciences, 5* (2), 85-93.

Nunan, F. (2000). Urban Organic Waste Markets: Responding to Change in Hubli-Dharwad, India. *Habitat International, 24*, 347-360.

Nyle, C. B., & Ray, R. N. (1999). *The Nature and Properties of soils* (12th ed.). USA.

Odunaike, R. K., Alausa, S. K., Fasunwon, O. O., Orunsolu, B. A., & Ijeoma, G. C. (2009). Radiation dose survey of refuse dumpsites in Abeokuta in Ogun State in the Southwestern zone of Nigeria. *Research Journal of Environmental Sciences, 3* (2), 262-266.

Odunaiya, C. O. (2002). Repositioning solid waste management practices in cosmopolitan Lagos State. *Lagos State Polytechnic Inaugural lecture series, 6*, p. 66. Lagos.

OECD. (1979). *Exposure to radiation from natural radioactivity in building materials.* Report by Group of Experts of Organisation for Economic Cooperation, Nuclear Energy Agency, Paris, France.

Ogwuegbu, M. O., & Ijioma, M. A. (2003). *Effects of Certain Heavy Metals on the population due to Mineral Exploitation. International Conference on Scientific and Environmental Issues in the population, Environment and Sustainable Development in Nigeria*. University of Ado Ekiti, Ekiti State, Nigerian. Pp.8-10.

Ogwuegbu, M.O., & Muhanga, W. (2005). Investigation of Lead Concentration in the Blood of people in the Copper belt Province of Zambia. *Journal of Environmental Science*. (1): 66-75.

Okeyode, I. C., & Akanni, A. O. (2009). Determination of some physical parameters of Olumo Rock, Abeokuta, Ogun State, Nigeria. *Indian Journal of Science and Technology, 2* (7), 6-10.

Okeyode, I. C., & Rufai, A. A. (2011). Determination of elemental composition of soil samples from some selected dumpsites in Abeokuta, Ogun State Nigeria, Using Atomic Absorption Spectrophotometer. *International Journals of Engineering & Sciences, 11* (6), 55-70.

Oladapo, O. O., Oni, E. A., Olawoyin, A. A., Akerele, O. O., & Tijani, S. A. (2012). Assessment of Natural Radionuclides Level in Wasteland Soils around Olusosun Dumpsite Lagos, Nigeria. *Institute of Sound Recording Journal of Applied Physics, 2* (3), 38-43.

Olayiwola, O. A. (2013). Accumulation and Contamination of Heavy Metals In Soils and Vegetation from Industrial Area of Ikirun, Osun, Nigeria. *Global Journal of Pure of Applied Chemistry Research, 1* (1), 25-34.

Onibokun, A. G., Adadipe, N. O., & Sridlier, M. K. (2000). Affordable Technology and Statement in African Lesson and Experience. *Centre for African Settlement Studies and Development,13*,134.

Oyedele, J. A., Sitoka, S., & Davids, I. (2008). *Radionuclide concentrations in soils of northern Namibia.* South Africa: Oxford University Press.

PHAC. (2008). Nak'azdli Aboriginal Interests and Use of Study on the Proposed Milligan Copper-Gold Mine at Shus Nadloh. Public Health Agency of Canada. Retrieved on July, 20, 2015 from http://www.phac-aspc.gc.cal.

Piccard, R. D. (2005). Radiation Damage and Dose Measurement. Notes on Modern Physics and Ionizing Radiation. Chapter 6. Ohio. Retrieved on July 5, 2015. from http://oak.cats.ohiou.edu/piccard/radnotes/dose.html.

Piccolo, A., & Mbagulu, J. C. (1997, March/April 21-28). Exogenuos humic substances as conditions for the rehabilitation of degraed soil agro Foods Industry. *Hi Tech* .

Popoola, O. E., Bangbose, O., Okonkwo, O. J., Arowolo, T. A., Odukoya, & Popoola, A. O. (2012). Heavy metals content in playground topsoil of some public primary schools in metropolitan Lagos, Nigeria. *Journal of Environmental and Earth Sciences, 4* (4), 434-439.

Psichoudaki, M., & Papefthymiou. (2008). Natural Radioactivity Measurement in the City of Ptolemias (Northern Greece). *Journal of Environmental Radioactivity*, vol. 99, issue 7, 1011-1017.

Radwan, M.A., and Salama, A.K., (2006) Market Basket Survey for some Heavy Metals in Egyptian Fruits and Vegetables, Food. *Journal of Chemical toxicological* 44, 127-1278.

Rahaman, M. A. (1988). Recent Advances in the Study of the Basement Complex of Nigeria. Precambrian: Geology of Nigeria. *Publication of the Geological Survey of Nigeria, 3*, 11-34.

Rahid, H., Seema, A. K., Muhammad, T. S., & Liaqat, A. (2015). MultiStatistical Approaches for Environmental Geochemical Assessment of Pollutants in Soils of Gadoon Amazi Industrial Estate, Pakistan. *Journal of Soils Sediments, 15*, 1119-1129.

Ramani, R.V. (2001). Environmental planning in the mining Industry. *Progress and Prospects*. IMM No. 41, pp. 5-9.

Ramasamy, V., Suresh, G., Meenakshisundaram, V., Gajendran, V., & Gajendran, V. (2009). Evaluation of Radionuclides Content in the River Sediments and Excess Lifetime Cancer Risk Due to Gamma Radioactivity. *Research Journal of Environmental and Earth Sciences*, Vol. 1, (1): Page No: 6-10.

Reference values for trace elements in soil, Monitorul Official al Romaniei, No.303 bis/6XUII 1997 (in Romanian). (1997).

Said, A. F., Salam, A. M., Hassan, S. F., & Moh, M. S. (2010). Assessment of the indices at Wadi Sahu Area, Sinai, Egypt. *Tenth Radiation Physics and Protection conference*, (pp. 149-155). Nass city-Cairo.

Satya, P. (2014). *Nuclear Physics* (3rd ed.). Pragati Prag.

Schmidt, C. W. (2009). Spheres of Influences/Nanotechnology Related EHS Research. *Environmental Health Perspectives, 117* (4), A159-A161.

Stihi, C., Radulescu, C., Busuioc, G., Popescu, I. V., Gheboianu, A., & Ene, A. (2011). Studies on Accumulation of Heavy metals from substrate to edible wild mushrooms. *Romania Journal of Physics,* Vol. 56, No. 1-2, pp. 257-264, Bucharest.

Tahawy, E. I., Hanningan, R., Nassef, M., & Sayed, E. I. (2006). Determination Of Some Heavy Metals In The Environment of Sadat Industrial City. *2nd Environmental Physics Conference of the Faculty Of Science, Cairo University Egypt*, (pp. 145-146, 151). Cairo.

Taskin, H., Karavus, M., Topuzogluu, A., Hindiroglu, S., & Karahan, G. (2009). Radionuclide Concentrations in Soil and Lifetime Cancer Risk due to the gamma radioactivity in Kirklareli, Turkey. *Journal of Environmental Radioactivity*. 100: 49-53.

Tchobanogluos, G., Theisen, H., Theisen, H., & Vigil, S. (1993). *Integrated Solid Waste Management; Engineering Principle and Management Issues.* (International ed.). Singapore: McGraw – Hill Book Co.

Tomlinson, D. L., Wilson, J. G., Hurris, C. R., & Jeffney, D. W. (1980). Problems In The Assessment Of Heavy Metal Levels In Estuaries And The Formation Of Pollution Index. *Springer for Research and Development, 33,* 566-572.

Udedi, S. S. (2003). From Guinea Worm Scourge to Metal Toxicity in Ebonyi State, Chemistry in Nigeria as the New Millennium unfolds; 2(2):13-14.

United Nations. (1993). *Sources and effects of Ionizing radiation.* United Nations General Assembly, United Nations Scientific Commitee on the Effects of Atomic Radiation. New York: United Nations.

UNSCEAR. (2000). *Sources of Ionizing Radiation.* Report to the General Assembly with Annexes, United Nations, United Nations Scientific Commitee on the Effects of Atomic Radiation, New York.

Vladimir, S., & Robert, E. M. (1990). Environmental Occurrence and Impacts of Arsenic at a Gold Mining Sitein the Western United States. *International Journal of Mine Water*, volume 9, number 1-4 page 181.

Walker, H.,& Sibly, P. (2001). Principles of Ecotoxicology. Taylor and Francis, London.

Wild, A. (1994). *Soils and The Enrichment: An Introduction.* New York: Cambridge University Press.S

World Health Organisation. (1996). World Health Organization Guidelines for Drinking Water Quality. *Health Criteria and Supporting Information, 2*.

Yu, K. N., Lau, B. M., & Nikezic, D. (2008). Assessment of Environmental Radon Hazard using Human Respiratory Tract Models. *Journal of Hazardous Materials*. P.132, pp.98-110

Zakari, Y. I., Nasiru, R., & Abdullahi, M. A. (2013). Determination of absorbed and annual effective doses around Birnin Gwari artisanal goldmine, Kaduna State, Nigeria. *Journal of Environmental and Health Science, 5* (5), 252-255.

Ziskind, M., Jones, R. N., & Weill, H. (1976). Silicosis- State of the Art. Am. Rev. Respiration Distribution. 113:643-665.