ASSESSMENT OF CARCINOGENIC RISK ASSOCIATED WITH INGESTION EXPOSURE TO HEAVY METAL IN SOIL AROUND SAND MINE

BY

Mary lfeoma EBERE (Miss)
LSC2010021

DEPARTMENT OF SCIENCE LABORATORY TECHNOLOGY
FACULTY OF LIFE SCIENCES
UNIVERSITY OF BENIN
BENIN CITY

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A PROJECT WORK SUBMITTED TO THE DEPARTMENT OF SCIENCE LABORATORY TECHNOLOGY, FACULTY OF LIFE SCIENCES, UNIVERSITY OF BENIN, BENIN CITY, IN PARTIAL FULFILLMENT OF THE REQUIREMENT FOR THE AWARD OF BACHELOR OF SCIENCE DEGREE (B.Sc) IN SCIENCE LABORATORY TECHNOLOGY (GEOLOGY & MINING TECHNIQUES OPTION).

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CERTIFICATION

This is to certify that this undergraduate project work titled **ASSESSMENT OF CARCINOGENIC RISK ASSOCIATED WITH INGESTION EXPOSURE TO HEAVY METAL IN SOIL AROUND SAND MINE** was submitted and presented by **Mary Ifeoma EBERE** with matriculation number LSC2010021 in the Department of Science Laboratory Technology, Faculty of Life Sciences, University of Benin, Benin City

Dr. K. OJEAGA **Date**
(Project Supervisor)

DR. P. O. ALONGE **Date**
(Project Coordinator)

PROF. J. O. OSARUMWENSE **Date**
(Head of Department, SLT)

External Examiner **Date**

DEDICATION

This project is dedicated to Almighty God, whose infinite wisdom, strength, and guidance have been my source of Inspiration throughout this academic journey. His grace has seen me through Challenges and moment of uncertainty and for that l am forever grateful.

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ABSTRACT

This study assessed heavy metal contamination and associated health risks in soils around a sand mining site in Ikpeshi, Edo State, Nigeria. Soil samples were collected at varying distances from the mine and analyzed for Fe, Zn, Cu, Pb, Cd, Mn, Ni, Cr, and Co. Results showed that most metals were below U.S. EPA limits, except Cobalt (Co). Pollution indices indicated low contamination levels overall, though Cadmium (Cd) showed moderate contamination linked to mining activities. The ecological risk index (RI = 79.11) revealed low ecological threat. Health risk assessment found negligible non-carcinogenic effects, while the Total Carcinogenic Risk (TCR) from Pb, Cd, Ni, and Cr was within the U.S. EPA acceptable range (10^{-6} to 10^{-4}) for both adults and children. However, Cd and Cr contributed most to potential cancer risk. Continuous monitoring and stricter mining regulations are recommended to prevent future health hazards.

CHAPTER ONE

INTRODUCTION

1.1. BACKGROUND OF STUDY

Sand mining is the extraction of sand from diverse natural environments such as rivers, coastal areas, and inland dune systems. This practice has gained global environmental concern due to its pervasive nature and the ecological stress it imposes. The surge in demand for sand, primarily propelled by rapid urbanization and infrastructural expansion, has led to unsustainable extraction practices that severely strain natural systems (Bayram and Önsoy, 2015). Among the most adversely impacted components of the environment are groundwater resources, which are indispensable for ecological sustainability, agriculture, and potable water supply. Sand mining interferes significantly with the natural hydrological cycle, particularly groundwater recharge processes. Typically, groundwater is replenished through the infiltration of surface water, a process facilitated by the permeability of soil and sediment layers. However, sand extraction, especially from fluvial and coastal settings, compromises the permeability of these systems, reducing their water retention capacity and impairing groundwater recharge (Barman *et al.,* 2019).

This disruption often results in the morphological alteration of riverbeds and aquifers, manifesting as channel deepening and widening. Such modifications increase flow velocity, thereby shortening the residence time required for infiltration, and consequently diminishing recharge efficiency. Furthermore, the destabilization of riverbanks caused by sand removal can enhance sediment transport and deposition downstream. These sediments may clog aquifer pore spaces, further limiting groundwater infiltration (Rezagama, 2019). In extreme cases, excessive sand mining may increase hydraulic connectivity between surface water and aquifers. While this may temporarily augment groundwater flow, it frequently leads to unsustainable extraction rates where aquifers are depleted more rapidly than they are recharged (Yen and Rohasliney, 2013).

A notable hydrological Impact of sand mining is the depression of the water table. As sand is excavated from hydrologically active zones, groundwater surfaces may become exposed, leading to increased evaporative losses and a subsequent decline in groundwater levels. This is particularly concerning in arid and semi-arid regions, where groundwater serves as a critical resource for both domestic and agricultural use (Brown *et al.,* 1998). In addition to quantitative reductions, sand mining can degrade groundwater quality. Natural sediment layers often serve as filters, trapping contaminants and preventing their entry into aquifers. The disruption of these filters facilitates the leaching of pollutants including heavy metals into the groundwater system. Moreover, sediment-laden water from destabilized riverbeds can transport fine particulates into aquifers, leading to clogging of boreholes and a decline in water quality (Donald and Blessing, 2019).

In coastal environments, sand extraction can also trigger saltwater intrusion. The removal of protective sand barriers allows seawater to encroach into freshwater aquifers, rendering groundwater saline and unsuitable for most agricultural and domestic purposes (Zamroni *et al.,* 2021). For many rural communities that rely heavily on groundwater as their primary drinking source, the degradation of groundwater both in quantity and quality due to sand mining may result in water scarcity and increased dependence on alternative, costlier water sources. This not only places financial stress on vulnerable populations but also poses serious health risks where access to clean water is limited (Sreebha and Padmalal, 2011).

Over recent decades, the intensification of industrialization, urban growth, and infrastructure development has dramatically increased the demand for sand and gravel (Bayram and Önsoy, 2015). These materials are foundational to diverse construction activities, including the creation of artificial islands, coastal reinforcements, roads, bridges, airports, and septic systems (Ashraf *et al.,* 2011). The escalating demand has led to the proliferation of sand mining activities, particularly within riverine and watershed areas (Lusiagustin and Kusratmoko, 2017). While economically beneficial especially for developing regions these activities pose considerable environmental risks. Environmental consequences become particularly severe when the rate of extraction surpasses the natural replenishment of sand and gravel deposits (Ashraf *et al.,* 2011; Rachmawati and Zamroni, 2020). The disruptive nature of sand mining modifies river morphology by intensifying sediment mobilization and altering flow regimes. The resulting turbulence accelerates bed load transportation, which further erodes the river's structural stability (Barman *et al.,* 2019). Concurrently, rising populations, economic advancement, and increased urbanization have sustained an ever-growing demand for construction-grade sand, primarily sourced from river systems due to their accessibility and material quality (Nabegu, 2013).

Heavy metals, although naturally occurring in trace amounts, become hazardous when they accumulate to toxic levels (Shakya and Agarwal, 2020). Their persistence and tendency to bioaccumulate in ecosystems necessitate rigorous monitoring and public awareness, particularly in areas near mining operations where contamination risks are elevated. Heavy metals such as cadmium, lead, chromium, and arsenic pose significant carcinogenic and toxic threats to humans and ecosystems (Wu *et al.,* 2016). The intensification of industrial activities, urban sprawl, and modern agricultural practices are major contributors to soil contamination, especially with heavy metals and metalloids (Zorpas *et al.,* 2021). Emissions from industries, mining operations, improper waste disposal, and the extensive use of agrochemicals elevate the deposition of these hazardous substances in soils, further complicating environmental and public health challenges (Gabarron *et al.,* 2017).

The environmental consequences of sand mining manifest both in the short and long term, leading to the alteration of natural landscapes, soil compaction, increased erosion, and a decline in biodiversity. While sand mining yields considerable economic advantages, the environmental costs, including pollution of air and water and destruction of natural habitats, are frequently overlooked. The rising global concern regarding heavy metal contamination is largely attributed to their increasing application in response to escalating population demands. Both natural processes and anthropogenic activities contribute to environmental pollution, often surpassing the assimilative capacity of ecosystems. This imbalance disrupts ecological networks, affects food chains, and introduces significant public health hazards. Although certain heavy metals are essential micronutrients, their elevated concentrations can induce toxicity, leading to various forms of poisoning (Egorova and Ananikov, 2017).

Heavy metals interact with biological molecules by forming metal cations, which interfere with the structure and function of critical macromolecules. Their toxicological effects are widespread, impairing major organ systems such as the gastrointestinal tract, kidneys, nervous system, and immune system. Conditions such as congenital anomalies and malignancies have also been linked to heavy metal exposure. The health risks are often exacerbated by simultaneous exposure to multiple metals. For example, concurrent exposure to mercury and lead can result in severe outcomes such as renal failure (Morais *et al.,* 2012).

The entry of heavy metals into the food chain occurs primarily through microbial and plant uptake, ultimately affecting animals and humans. Metals such as lead and zinc, even in minimal concentrations, can impair nearly all organ systems, and their accumulation at higher levels can lead to significant disruptions in cellular and physiological functions. Chronic exposure to metals such as nickel has been associated with serious health outcomes, including carcinogenesis. A thorough understanding of the pathways through which heavy metals induce cancer, particularly those involving aberrant gene expression and DNA damage, is essential (Itam *et al.,* 2024). Accurate health risk assessments must identify the sources of heavy metal contamination in the environment and evaluate their quantitative impacts on human well-being. Toxic substances naturally embedded in soils, rocks, and minerals are often released during mining operations in the form of dust, gases, and airborne particulates. These pollutants can settle on terrestrial surfaces and vegetation or be transported to aquatic ecosystems, contributing to environmental toxicity (Anand, 2006; Lameed and Ayodele, 2010).

Assessing human health risks involves evaluating potential negative health outcomes resulting from environmental exposure to hazardous substances. These risks are typically categorized as non-carcinogenic or carcinogenic (USEPA, 1989). According to the United States Environmental Protection Agency (USEPA), non-carcinogenic risks are assessed using indicators such as chronic daily intake (CDI), hazard quotient (HQ), and hazard index (HI). In contrast, carcinogenic risks are evaluated using CDI in conjunction with the cancer slope factor (CSF). The threshold reference dose (RfD) is employed in non-carcinogenic risk assessments to denote the exposure level below which no adverse effects are anticipated. However, in the case of carcinogenic substances, no exposure level is considered entirely safe, implying that even minimal contact may entail some risk of cancer development (USEPA, 1986).

1.2 AIM AND OBJECTIVES

The aim of this study is to assess the carcinogenic risk associated with ingestion exposure to heavy metals in soils surrounding sand mining areas.

The objectives are:

- To determine the concentrations of selected heavy metals in soil samples collected around sand mining sites.
- To evaluate the ingestion exposure levels
- To assess the carcinogenic risks posed by ingestion of contaminated soil using standard risk assessment models
- To compare the concentrations and determine the level of public health concern.

1.3 STATEMENT OF PROBLEM

Sand mining is a critical activity in the construction industry that is often associated with significant environmental degradation, particularly through the release of toxic heavy metals into the surrounding ecosystem. During the excavation and processing of sand deposits, naturally occurring heavy metals such as cadmium (Cd), lead (Pb), chromium (Cr), cobalt (Co), nickel (Ni), and zinc (Zn) can be mobilized and accumulate in surface soils. These contaminants pose substantial risks to human health, especially in communities residing near mining sites, where prolonged exposure is common. One of the primary pathways of human exposure is through the inadvertent ingestion of contaminated soil particles, either directly, especially among children, or indirectly through the consumption of crops grown in such soils. Among the various health effects associated with heavy metal exposure, carcinogenicity remains a paramount concern. Chronic ingestion of heavy metals has been linked to increased incidences of various cancers, including those of the liver, kidneys, lungs, and gastrointestinal tract. Despite this growing concern, there is a scarcity of localized data assessing the carcinogenic risks posed by ingestion exposure to heavy metals in soils surrounding sand mines. Therefore, a detailed assessment of carcinogenic risk associated with ingestion of heavy metal-contaminated soils around sand mining areas is urgent, in order to elucidate the extent of the hazard so as to make informed decisions.

1.4. JUSTIFICATION OF STUDY

This study is imperative in light of the potential health implications associated with the ingestion of heavy metal-contaminated soils in sand mining environments. By evaluating the carcinogenic risks linked to specific heavy metals in the vicinity of sand mining operations, this research will provide critical insights into the magnitude of public health threats faced by nearby communities. It will help identify the most hazardous contaminants and assess their concentration levels relative to international safety standards. Additionally, the study will contribute to the understanding of exposure pathways and risk characterization for ingestion, which remains one of the least examined routes in environmental health research, especially in low- and middle-income settings. The results will offer a scientific basis for environmental monitoring and control measures around sand mining sites. In addition to its public health relevance, this research will also highlight the environmental consequences of sand mining on soil quality, and bring to light, best practices for mitigating contamination.

CHAPTER TWO

LITERATURE REVIEW

2.1. CONCEPTUAL OVERVIEW

mining refers to the extraction of sand from natural reserves such as rivers, coastal plains, and terrestrial deposits. This activity serves as a backbone to many industrial operations, particularly in the construction sector, where sand constitutes a primary ingredient in the manufacture of concrete, asphalt, and glass (Peduzzi, 2014). As global urbanization accelerates and infrastructure projects expand, the demand for sand has surged, propelling sand mining into one of the most extensive extractive industries globally (Koehnken and Rintoul, 2018). However, the intensification of sand mining has provoked substantial concern regarding its environmental sustainability and ecological ramifications.

When poorly regulated, sand extraction can trigger extensive environmental degradation. Notable consequences include the disruption of landforms, loss of biological diversity, and disturbance of both terrestrial and aquatic ecosystems (Saviour, 2012). One of the most acute environmental threats associated with this activity is its adverse effect on soil systems. Vegetative cover is often removed during mining operations, exposing soils to accelerated erosion and structural deterioration (UNEP, 2019). In addition, the process can result in the accumulation of harmful substances, particularly heavy metals, in the soil matrix, thereby heightening environmental and public health risks (Bhat and Ullal, 2014).

Soils are foundational to ecosystem regulation, agricultural productivity, and hydrological filtration. Disturbances arising from sand mining alter the physical structure and chemical balance of soils, reducing fertility, undermining soil cohesion, and increasing susceptibility to erosion (Erskine, 1990). When these processes are left unchecked, they can initiate a cascade of degradation, particularly in dry regions where sand mining contributes to desertification and agricultural decline (Schandl *et al.,* 2016). These impacts can jeopardize regional food systems and ecological resilience.

A major environmental implication is the introduction of heavy metals into soils during mining. Toxic elements such as lead (Pb), cadmium (Cd), mercury (Hg), and arsenic (As) pose long-term risks, even in trace amounts (Chakraborty *et al.,* 2017). These metals may originate from both the geological substrate and from anthropogenic sources such as machinery and transportation activities. Once embedded in the soil, heavy metals exhibit persistence over extended periods, depending on the metal's chemical nature and the soil's retention capacity (Nagajyoti *et al.,* 2010).

Quantifying soil pollution in mining regions is critical for assessing ecological harm. Numerous studies have reported elevated concentrations of heavy metals in soils adjacent to sand mining sites, often surpassing environmental safety thresholds (Aigberua *et al.,* 2018). Researchers frequently utilize indices like the Geo-accumulation Index (Igeo), the Pollution Load Index (PLI), and the Enrichment Factor (EF) to evaluate contamination levels and determine ecological risks (Müller, 1969; Harikumar and Jisha, 2010).

The ecological toxicity of heavy metals is especially concerning, given their potential to be absorbed by vegetation and transmitted through the food chain (Alloway, 2013). Their mobility and bioavailability are influenced by parameters such as pH, organic matter, and mineral composition of the soil (Kabata-Pendias, 2011). In heavily contaminated soils, bioaccumulation in plants and animals may result in long-term ecological stress and significant human health implications (Sarma, 2011).

2.2 ENVIRONMENTAL IMPACTS OF SAND MINING

Sand mining has emerged as a significant environmental challenge due to its profound and widespread ecological implications. While essential for infrastructure and construction, uncontrolled sand extraction often leads to ecological degradation, particularly in terms of soil quality, landscape disruption, and ecosystem stability. The removal of sand from natural settings such as riverbeds, coastlines, and inland deposits frequently alters geomorphological structures and enhances susceptibility to erosion (Bhattacharyya *et al.,* 2019).

The removal of the topsoil, essential for vegetation growth and soil function, often results in severe land degradation. In riverine environments, the excavation of sand reduces the riverbed's depth, destabilizes banks, and increases sediment transport downstream (Padmalal and Maya, 2014). These disturbances lead to agricultural losses, impaired drainage systems, and heightened flood risks. In coastal zones, sand removal exacerbates saltwater intrusion into freshwater aquifers, causing soil salinization and a decline in agricultural viability (Kondolf, 1997). In many developing regions, especially in parts of Africa and Southeast Asia, such unregulated mining has rendered formerly arable land infertile and ecologically degraded (UNEP, 2019). These environmental alterations also foster desertification, particularly in dry climates where vegetation loss and soil destabilization are already pressing concerns (Saviour, 2012).

2.2.1 Alteration of Soil Physical and Chemical Properties

Sand mining exerts significant influence on the physicochemical characteristics of soils. The extraction process depletes topsoil layers that are rich in nutrients and organic matter, resulting in diminished soil fertility and reduced water-holding capacity (Kondolf *et al.,* 2002). These conditions impair microbial activity and lower overall soil productivity. Additionally, pollutants introduced during the mining process, especially heavy metals from machinery and transportation, further compromise soil health (Bhat and Ullal, 2014).

Such heavy metals tend to persist in the environment due to their non-biodegradable nature, leading to chronic contamination. The presence of elements like Pb, Cd, Hg, and As in soil poses lasting threats to both ecosystems and human populations, as they can enter the food chain through plant uptake (Nagajyoti *et al.,* 2010).

2.2.2 Loss of Biodiversity and Ecosystem Disruption

Sand extraction disrupts both terrestrial and aquatic biodiversity. Terrestrial mining often leads to habitat loss, especially for species reliant on stable vegetation and undisturbed soils (Bhattacharyya *et al.,* 2019). In aquatic systems, sand removal modifies sediment dynamics, destroying breeding habitats for fish and invertebrates (Padmalal *et al.,* 2008; Koehnken and Rintoul, 2018). Such disruptions reduce ecological resilience and compromise ecosystem services like nutrient cycling, water purification, and climate regulation (Schandl *et al.,* 2016). In coastal regions, the destruction of buffer systems such as mangroves further exposes communities to natural hazards including erosion and storm surges (Peduzzi, 2014).

2.3 HEAVY METAL CONTAMINATION AND SOIL POLLUTION

One of the gravest outcomes of sand mining is the contamination of soils with heavy metals. These pollutants may be released through the mining of mineral-rich geological layers or from anthropogenic activities related to extraction. Elements such as lead, cadmium, mercury, arsenic, and chromium are frequently detected in excess in mining zones (Chakraborty *et al.,* 2017). Over time, these contaminants accumulate and alter soil chemistry, affecting fertility and posing hazards to health and ecological systems (Aigberua *et al.,* 2018). Their persistence, influenced by soil texture, pH, and moisture content, allows for their transmission through runoff or biological uptake, thereby amplifying environmental damage (Kabata-Pendias, 2011; Alloway, 2013).

2.4 HUMAN HEALTH RISKS ASSOCIATED WITH SAND MINING

The consequences of sand mining are not limited to ecological damage. Human populations residing near mining sites are at risk of exposure to soil-borne pollutants. Heavy metals can enter the human body through the consumption of contaminated crops, ingestion of polluted water, or direct contact with contaminated soils (Chakraborty *et al.,* 2017). Chronic exposure to metals such as lead and cadmium has been linked to various health conditions, including neurological impairments, renal dysfunction, and carcinogenesis (Sarma, 2011). Furthermore, soil degradation resulting from sand mining can reduce agricultural productivity, thereby threatening food security in vulnerable communities (UNEP, 2019). The loss of ecosystem services further compounds the risks, weakening natural defenses against environmental shocks (Schandl *et al.,* 2016).

2.5 ASSESSMENT OF SOIL POLLUTION IN SAND MINING AREAS

The environmental degradation associated with sand mining necessitates a thorough investigation of soil pollution. In particular, the risk of contamination by heavy metals has emerged as a critical concern. Sand mining activities can expose and mobilize toxic elements like Pb, Cd, As, and Hg, fundamentally altering soil chemistry and posing long-term ecological threats (Ramachandra *et al.,* 2018). Such metals may originate from both geological strata and operational inputs such as machinery or chemicals (Akande *et al.,* 2021). Comparative analyses have shown that soils in proximity to mining operations exhibit elevated metal concentrations, often exceeding international safety thresholds (Manjunatha *et al.,* 2016; Olatunji and Osibanjo, 2018).

The removal of nutrient-rich topsoil leads to decreased porosity and organic content, impairing water infiltration and root development. In Brazil, research revealed a 30% decline in soil porosity in mined areas, coupled with reduced vegetative support (Silva *et al.,* 2019). Similarly, studies in Nigeria have highlighted the increased erosion of soils during rainy seasons, often resulting in sedimentation and further environmental degradation (Adekola *et al.,* 2020). The depletion of essential nutrients also affects agricultural productivity. Loss of nitrogen, phosphorus, and potassium; commonly concentrated in upper soil horizons, diminishes crop yields. In Kenya, sand mining sites exhibited significant nutrient loss compared to unaffected farmlands (Njeru *et al.,* 2018).

2.6. BIOAVAILABILITY OF POLLUTANTS AND TOXICITY

The ecological risk posed by heavy metal contamination is further complicated by the bioavailability of these elements. Pollutants in the soil can be absorbed by plants and enter the food chain, threatening both wildlife and human health. South Asian studies have confirmed increased metal uptake in crops grown on contaminated soils, particularly cadmium and lead, which are linked to renal and neurological disorders (Saha *et al.,* 2017). In addition, chemicals used during mining operations such as lubricants, fuels, and explosives can leach into the soil, intensifying its toxicity (Ojelede and Okonofua, 2019).

2.7 IMPACTS OF SAND MINING ON SOIL PROPERTIES

Sand mining substantially alters the physical, chemical, and biological characteristics of soil, which are foundational to ecosystem functionality, agricultural viability, and environmental sustainability (Saviour, 2012).

2.8 PHYSICAL AND BIOLOGICAL PROPERTIES OF SOIL IN SAND MINING AREAS

2.8.1. Alteration of Physical Soil Properties

One of the most immediate consequences of sand mining is the disturbance of the physical composition of soil. This typically begins with the removal of the topsoil, which contains a high proportion of organic matter and nutrients essential for plant growth and soil stability. The exposure of subsoil layers, which are often structurally and texturally distinct, results in diminished soil cohesion and increased susceptibility to degradation (Saviour, 2012). Furthermore, heavy machinery used during sand mining compacts the remaining soil layers, reducing porosity and increasing bulk density. These changes hinder root penetration and restrict vegetative development (Kondolf *et al.,* 2014). Empirical evidence from Padmalal *et al.* (2014) in India indicates that sand-mined soils exhibit a 20–30% reduction in porosity relative to undisturbed sites, which adversely affects water infiltration and enhances runoff, erosion, and flood risk.

2.8.2. Soil Structure and Aggregate Stability

Soil structure, defined by the spatial arrangement of particles into aggregates, is vital for air circulation, water movement, and root proliferation. Sand mining frequently disrupts this structure by disaggregating soil particles, especially in sandy soils. The breakdown of aggregates not only reduces structural integrity but also increases erosion potential under wind and water action (Kondolf, 2017). Findings by Ali and Hayati (2019) revealed a marked decline in soil aggregation in mining-affected areas in Southeast Asia, corresponding with a 40% rise in erosion rates. The disintegration of soil structure leads to nutrient losses through leaching and sediment displacement, thereby impairing the land's agricultural viability.

2.8.3 Soil Moisture Retention and Hydrological Properties

The extraction of sand alters the hydrological properties of soil, particularly its capacity to retain moisture. Mining tends to remove finer soil components such as clay and silt, which are critical for water retention. As a result, mined soils become dominated by coarse sand particles with low water-holding capacity, making them vulnerable to desiccation and unsuitable for cultivation (Adekola *et al.,* 2020). A study by Mabiza *et al.* (2018) in South Africa demonstrated that soil moisture content in sand-mined areas was approximately 50% lower than in comparable undisturbed locations, primarily due to the reduction of fine-textured particles.

2.7.4. Soil Nutrient Content and Fertility Decline

The extraction of sand severely affects the nutrient profile of soil. The topsoil, which is abundant in key macronutrients such as nitrogen (N), phosphorus (P), and potassium (K), is typically removed during mining, leaving behind nutrient-deficient subsoils. This nutrient loss leads to a sharp decline in soil fertility, with implications for crop production and ecological resilience (Oladipo *et al.,* 2019). Research in Kenya by Njeru *et al.* (2018) found that sand-mined soils contained significantly lower levels of N and P compared to nearby undisturbed lands. The study reported a 60% decline in fertility, rendering the soil unsuitable for both agricultural use and natural regeneration. Similar long-term fertility reductions have been observed in Nigerian mining regions lacking proper restoration measures (Adekola *et al.,* 2020).

2.8. BIOLOGICAL SOIL FUNCTIONS AND MICROBIAL ACTIVITY

2.8.1. Impacts on Soil Microorganisms

Soil biological activity, particularly the function of microorganisms and fauna, is essential for nutrient cycling, organic matter decomposition, and overall ecosystem health. Sand mining disturbs these biological processes by degrading the environmental conditions required for microbial viability. The removal of organic-rich topsoil and changes in moisture and texture lead to reduced microbial biomass and diversity (Blaikie and Brookfield, 2015). Raju *et al.* (2019) reported significantly lower enzymatic activity and microbial biomass in soils from sand mining sites, indicating compromised biological functionality. Such declines in microbial health impair the soil's capacity to recover and sustain vegetative life following disturbance.

2.9. ECOTOXICOLOGICAL IMPLICATIONS OF HEAVY METAL CONTAMINATION

2.9.1 Introduction and Mobilization of Heavy Metals

Sand mining contributes to the mobilization of toxic heavy metals such as lead (Pb), cadmium (Cd), mercury (Hg), chromium (Cr), and arsenic (As) into the environment. These metals may originate from the mined geological materials or be introduced through mechanical and chemical processes during mining operations (Zhang *et al*., 2019; Tang *et al.,* 2020). Once mobilized, heavy metals interact with various soil components, with their mobility and toxicity influenced by pH, redox potential, and mineralogical composition (Alloway, 2013). Mining activities expose deeper soil layers and disrupt stable mineral forms, increasing metal bioavailability and distribution across the landscape (Wang *et al.,* 2020).

2.9.2 Effects on Microbial Activity and Plant Growth

Heavy metals exert toxic effects on soil microbial communities, inhibiting vital enzymatic functions and altering population structure. Elements such as Cd, Pb, and Cr are known to interfere with enzymatic processes, particularly those related to dehydrogenase and urease activities, which are crucial for soil metabolism and fertility (Rajapaksha *et al.,* 2012; Giller et al., 2009). In plants, these metals disrupt nutrient uptake, impair photosynthesis, and hinder root development. Cadmium, for example, competes with calcium and magnesium absorption, affecting chlorophyll synthesis and overall plant health (Liu *et al.,* 2019). Reduced vegetation cover due to phytotoxicity further exacerbates soil erosion and habitat loss.

2.9.3. Bioaccumulation and Biomagnification

The toxic metals introduced through mining are not confined to soil systems; they bioaccumulate in flora and fauna and biomagnify through trophic levels. Bioaccumulation refers to the progressive accumulation of metals in organisms, while biomagnification describes the increase in metal concentrations along the food chain (Chibuike and Obiora, 2014). Earthworms in contaminated soils, for instance, accumulate heavy metals which are then transferred to higher predators such as birds and mammals (Sizmur and Hodson, 2009). In aquatic environments, leaching and runoff from mining zones contribute to metal uptake by fish and aquatic invertebrates, ultimately impacting human populations through consumption of contaminated food sources (Zhou *et al.,* 2008).

2.9.4. Soil Fauna and Ecosystem Impairment

Heavy metals adversely affect soil-dwelling organisms, including earthworms, nematodes, and arthropods, which are integral to maintaining soil structure and nutrient turnover. Exposure to toxic elements results in reduced reproduction, developmental anomalies, and increased mortality in these organisms (Yeung *et al.,* 2017). Earthworms, often used as bioindicators, have demonstrated sensitivity to Pb and Cd, showing signs of oxidative stress, DNA damage, and reduced metabolic efficiency (Spurgeon *et al.,* 2005). The decline in soil fauna diversity compromises ecological functions and perpetuates a cycle of soil degradation.

2.10. PUBLIC HEALTH IMPLICATIONS

The contamination of soil and water resources with heavy metals presents serious risks to human health. Individuals living near sand mining sites may be exposed to toxic metals through ingestion of contaminated crops and water, inhalation of dust, or dermal contact (Naseem and Tahir, 2001). Lead, mercury, and cadmium are particularly hazardous, associated with neurological disorders, renal failure, cardiovascular complications, and cancer (Tchounwou *et al.,* 2012). In regions where mining is poorly regulated, arsenic exposure via drinking water has been linked to carcinogenesis and dermatological conditions (Singh *et al.,* 2015).

2.11. SPATIAL DISTRIBUTION OF HEAVY METALS IN MINING ZONES

2.11.1. Patterns of Contaminant Dispersion

Understanding the spatial variation in heavy metal concentrations is crucial for evaluating the environmental impact of sand mining. Studies have shown that metal concentrations are typically highest near the source of mining activities and decline with distance due to dilution and dispersion (Tchounwou *et al.,* 2012). The spatial pattern is shaped by environmental variables such as topography, soil type, and hydrology. For instance, low-lying areas may serve as sinks for heavy metals transported through surface runoff and erosion (Das *et al.,* 2023), while higher terrains may exhibit reduced concentrations due to enhanced leaching.

2.11.2 Use of Spatial Tools in Environmental Assessment

Geographic Information Systems (GIS) and spatial modeling have proven valuable for mapping contamination, identifying pollution hotspots, and guiding remediation efforts. These techniques allow for precise visualization of heavy metal dispersion and facilitate environmental risk assessments (Qin *et al.,* 2021). A case study in Nigeria illustrated strong spatial correlation between mining operations and elevated heavy metal levels in agricultural soils, highlighting the pervasive nature of pollution and its implications for food safety (Abrahim and Parker, 2008).

2.12. FACTORS INFLUENCING METAL DISTRIBUTION

The retention and mobility of heavy metals in soil are regulated by characteristics such as texture, pH, organic content, and moisture. Metals tend to bind to clay particles and organic matter, which may reduce immediate bioavailability but influence long-term distribution patterns (Alloway, 2013; Golia, 2023). Hydrological events, particularly heavy rainfall, can accelerate metal transport through leaching and runoff, expanding the area affected by contamination (Wuana and Okieimen, 2011).

2.13. IMPLICATIONS FOR ECOSYSTEMS AND HUMAN POPULATIONS

The uneven distribution of heavy metals has important implications for environmental quality and public health. Soils with high contaminant loads can impair microbial and plant communities, reducing fertility and ecosystem productivity (Rajapaksha *et al.,* 2012). Metals absorbed by crops grown in contaminated areas may enter the human food chain, posing direct threats to health (Wuana and Okieimen, 2011). Spatial distribution analyses are therefore instrumental in prioritizing remediation strategies such as phytoremediation and soil amendments, which can be targeted to the most affected zones (Salt *et al.,* 1998).

2.14. HEAVY METAL ANALYSIS IN SAND MINING AREAS

The assessment of heavy metal concentrations is a fundamental aspect of environmental monitoring in regions affected by sand mining. Analytical evaluation of toxic elements such as Cd, Pb, Hg, and As helps determine pollution severity, trace sources of contamination, and evaluate the success of mitigation efforts. Given their persistence, toxicity, and capacity for bioaccumulation, these metals serve as key indicators of soil and ecosystem health. Systematic heavy metal analysis informs risk management decisions and supports the development of strategies to restore degraded environments (Tchounwou *et al.,* 2012).

2.15. CARCINOGENIC ASSESSMENT OF HEAVY METALS

Heavy metals (HMs) represent a significant hazard to human health, particularly due to their tendency to bioaccumulate within biological systems over time. Chronic or prolonged exposure can lead to toxicological effects that compromise multiple physiological systems (Huang *et al.,* 2018; Zhang *et al.,* 2020). Once absorbed, HMs can accumulate in vital organs and tissues, disrupting the functions of the immune, neurological, endocrine, cardiovascular, and skeletal systems (Doležalová *et al.,* 2019).

Contaminated sediments serve as another reservoir for human exposure, with ingestion, skin contact, and inhalation identified as the primary exposure mechanisms (Miletic *et al.,* 2023). Individuals may come into contact with such sediments during recreational or occupational activities, or simply by residing near contaminated zones. Aquatic organisms inhabiting polluted environments can bioaccumulate HMs, thus representing another route of human exposure through seafood consumption (Miletic *et al.,* 2023). Heavy metals are generally categorized into essential and non-essential types. Essential trace elements such as chromium (Cr), copper (Cu), nickel (Ni), and zinc (Zn) are required for normal physiological function but become toxic when present in excess. In contrast, non-essential elements including arsenic (As), cadmium (Cd), mercury (Hg), and lead (Pb) are intrinsically harmful even at low concentrations (Pecina *et al.,* 2021). Both categories are capable of inducing cellular toxicity by disrupting redox homeostasis, impairing enzymatic activity, and altering signal transduction pathways once inside human cells (Miletic *et al.,* 2023).

Numerous health conditions have been directly linked to specific HMs. Chromium, for example, is associated with respiratory tract malignancies; arsenic exposure has been implicated in neurodevelopmental deficits in children and disorders of the skin and liver (Fan *et al.,* 2022). Cadmium is known to contribute to renal dysfunction, pulmonary toxicity, and bone demineralization, while excessive nickel exposure has been linked to respiratory ailments such as asthma and fibrosis, as well as dermatological conditions like contact dermatitis. Lead, one of the most studied toxic metals, is notorious for its neurotoxicity in children and its adverse effects on the immune, reproductive, and skeletal systems (Bai *et al.,* 2020). Elevated copper intake, though less common, can lead to hematological disturbances such as anemia and gastrointestinal distress (Miletic *et al.,* 2023).

2.16. CARCINOGENIC RISK

The carcinogenic potential of heavy metals present in soils and sediments is quantified to evaluate their ability to induce cancer in exposed human populations. This evaluation typically relies on two indices: carcinogenic risk (CR) and total carcinogenic risk (TCR). The TCR metric captures the aggregate probability of cancer development resulting from prolonged exposure to multiple carcinogenic substances. According to widely accepted environmental health standards, a TCR or CR value below 1×10^{-6} is regarded as negligible, while values exceeding 1×10^{-4} indicate an elevated and potentially unacceptable health risk (Zhou *et al.,* 2022).

Despite the general acceptance of the terms CR and TCR in environmental toxicology literature, various nomenclatures exist across studies. Some researchers refer exclusively to CR, while others incorporate alternative or supplementary terms such as CRt, LCR, TR, ILCR, ILCRt, RI, and even cancer risk or R to describe similar constructs (Miletic *et al.,* 2023; Karimi *et al.,* 2020).

2.17. ANALYTICAL TECHNIQUES FOR METAL QUANTIFICATION

The determination of heavy metal concentrations in extracted solutions relies on advanced instrumental techniques, each with specific strengths and limitations: Atomic Absorption Spectroscopy (AAS) is widely regarded for its precision and sensitivity in detecting individual metals at trace levels. While limited to single-element analysis per run, its reliability makes it a staple in environmental monitoring laboratories (Naseem and Tahir, 2001).

Inductively Coupled Plasma Mass Spectrometry (ICP-MS) offers superior sensitivity and the capacity for simultaneous multi-element analysis, enabling the detection of ultra-trace metal concentrations across diverse environmental matrices. Its high throughput and broad dynamic range render it ideal for comprehensive studies (Mao *et al.,* 2017).

X-ray Fluorescence (XRF) is a non-destructive and rapid screening tool that facilitates the insitu identification of heavy metals in solid samples. Although less sensitive for certain trace metals compared to AAS or ICP-MS, XRF is particularly useful for preliminary field investigations and spatial mapping of contamination (Marguí *et al.,* 2022).

2.18. RELATED LITERATURE

(Kilavi *et al.*, 2023) conducted a comprehensive assessment of heavy metal contamination in soils surrounding a heavy mineral sands (HMS) mining site along Kenya’s southern coast. Utilizing inductively coupled plasma mass spectrometry (ICP-MS), specifically the Agilent 7700s, the study quantified concentrations of various heavy metals and employed standard pollution assessment indices such as the geo-accumulation index (Igeo), enrichment factor (EF), and pollution index (PI) to determine the extent of soil contamination. Their findings revealed moderate pollution levels attributed to elements like titanium (Ti), manganese (Mn), zinc (Zn), and niobium (Nb), whereas chromium (Cr) and arsenic (As) exhibited considerably higher contamination levels. Cr concentrations in the topsoil surpassed the maximum allowable concentration (MAC) and trigger value (TV) by factors of 135 and 61, respectively. Likewise, As levels exceeded the MAC and TV by 8 and 2 times. The calculated enrichment factors further emphasized the extent of contamination, with Cr and As showing EF values approximately 200 and 25 times greater than the baseline threshold of 1.5. Health risk assessments, including non-carcinogenic risk (NCR) and carcinogenic risk (CR), indicated that As posed significant non-carcinogenic health concerns, while both Cr and As presented heightened carcinogenic risks. Notably, the CR values for these two elements surpassed the acceptable threshold of 10^{-4}, signaling a significant threat to human health via soil ingestion.

(Moghadam *et al.*, 2024) investigated the spatial distribution and contamination levels of heavy metals in agricultural soils in the northern region of Ahvaz, southwest Iran, particularly focusing on the Weiss and Arab Assad areas. Soil samples were systematically collected from agricultural farms and riverside areas along the Karun River. A total of nine farms were selected in each study zone, with ten samples obtained from each farm. Geographic Information System (GIS) techniques were employed to generate spatial distribution (zoning) maps for heavy metal concentrations. The contamination assessment utilized several established indices, including the contamination factor (CF), enrichment factor (EF), and geo-accumulation index (Igeo). Cadmium (Cd) exhibited the highest contamination indices among the analyzed metals, with values of 7.84 (CF), 73.92 (EF), and 2.38 (Igeo), signaling severe enrichment and pollution. In contrast, chromium (Cr) recorded the lowest contamination levels, with corresponding values of 0.21, 1.98, and −2.82. The findings revealed that Cd was the most toxic heavy metal in the agricultural soils examined, posing significant environmental and health risks. Further analysis of the GIS-based zoning maps demonstrated that while elements like cobalt (Co), copper (Cu), lead (Pb), and chromium (Cr) did not significantly contaminate wheat cultivation fields, cadmium (Cd) and zinc (Zn) were found at elevated and concerning levels. The spatial distribution of nickel (Ni) suggested that its presence stemmed from both natural geological formations and anthropogenic sources. The high EF values recorded for Cd, Cu, Cr, Pb, Ni, Fe, Mn, Co, and Zn were attributed to extensive fertilizer application (particularly phosphate and nitrate), as well as other agricultural, industrial, and human-related activities in the region.

(Rehman *et al.*, 2018) examined the extent of toxic metal contamination in soils and surface waters near the Sewakht mining region in Pakistan, with a focus on spatial distribution, enrichment, potential ecological risk index (PERI), and the associated human health risks. The study analyzed 54 soil samples from various agricultural fields and 38 surface water samples, targeting a range of heavy metals including cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe), manganese (Mn), lead (Pb), nickel (Ni), zinc (Zn), and molybdenum (Mo). To evaluate soil contamination, several pollution indices were employed, such as the geo-accumulation index (Igeo), contamination factor (CF), degree of contamination (CD), enrichment factor (EF), and PERI. The CF values suggested moderate contamination levels with Cd, Co, Fe, and Mo, while Igeo pointed to moderate accumulation particularly of Cu. Notably, Cd displayed EF values greater than 1.5 in agricultural soils, indicating significant anthropogenic enrichment. The ecological risk assessment revealed alarming findings; PERI values exceeded 380 at 4% of the sampled locations, indicating very high ecological risk, while 7.4% of sites posed considerable risk. In terms of human health, non-carcinogenic risk (Hazard Index, HI) from soil exposure to Fe was found to be above the acceptable threshold (HI > 1) for both children and adults. Carcinogenic risk (CR) estimates for children exceeded acceptable limits for Cd, Cr, Co, and Ni, while for adults, Co was the only metal exceeding the CR threshold (10^{-4}) through soil ingestion. Moreover, heavy metal exposure through surface water also posed serious health concerns. For children, non-carcinogenic risk due to Cd, Co, and Mo surpassed the safe limits (HQ > 1), whereas adults were at higher risk from Cr, Cd, Cu, Pb, Co, and Mo. Carcinogenic risk estimates from surface water exposure to Co, Cd, Cr, and Ni (excluding Pb) also exceeded acceptable limits in both adults and children. Interestingly, lead (Pb) concentrations in both soil and water were not found to pose a significant carcinogenic threat to the local population (Rehman et al., 2018).

(Belle *et al.*, 2024) conducted an in-depth evaluation of the health risks posed by heavy metal exposure in Matjhabeng Local Municipality, a prominent gold mining area in South Africa. The study applied a deterministic source–pathway–receptor model to quantify both non-carcinogenic and carcinogenic risks arising from metal-laden soil. By incorporating a broad spectrum of exposure pathways including ingestion, dermal contact, and inhalation; the study offered a holistic risk assessment framework particularly relevant for mining-affected populations. Soil samples were analyzed for multiple toxic elements, including arsenic (As), cadmium (Cd), lead (Pb), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe), nickel (Ni), selenium (Se), and zinc (Zn). Average concentrations for key metals were reported as 3.2 mg/kg for As, 2.5 mg/kg for Cd, 45 mg/kg for Pb, 17 mg/kg for Co, and 25 mg/kg for Cr. Non-carcinogenic risk via ingestion was found to be minimal across all demographic groups, with hazard quotient (HQ) values below the critical threshold (HQ < 1). However, dermal exposure revealed significant health concerns, particularly in adults, with HQ values exceeding 1 for Cd, Pb, Co, and Cr, suggesting elevated non-carcinogenic risks. Inhalation emerged as the most critical exposure pathway, with HQ values ranging between 5 and 15 and chronic hazard index (CHI) values far surpassing the safe benchmark (CHI > 5), thereby indicating serious long-term health implications. Carcinogenic risk estimates through inhalation were particularly concerning. Lifetime cancer risk values for As, Cd, Cr, Co, and

Ni significantly exceeded the U.S. Environmental Protection Agency's acceptable limits (1×10^{-4} to 1×10^{-6}) in both adult and juvenile populations.

CHAPTER THREE

MATERIALS AND METHOD

3.1. REGIONAL GEOLOGY

Ikpeshi lies within the Akoko-Edo Local Government Area of Edo State. Regionally, Akoko-Edo Local Government Area is geologically positioned in the southwestern portion of the Nigerian Basement Complex, a segment of the larger Pan-African mobile belt. This belt represents a tectonic domain that emerged from the collision and suturing of the West African and Congo cratons (Rahaman, 1988). The area's geology comprises a diverse assemblage of Precambrian crystalline basement rocks, which are emblematic of the Nigerian Basement Complex, as well as Cretaceous sedimentary sequences. The basement complex is chiefly made up of migmatite-gneissic formations, quartzites, schists, and granitic bodies that resulted from multiple tectonothermal episodes (Rahaman, 1988). These lithologies have experienced repeated cycles of metamorphism, structural deformation, and magmatic activity, notably associated with the Pan-African orogenic processes that occurred around 600 million years ago. In the Ikpeshi locality, the geological framework is defined by the interplay of ancient Precambrian rocks and younger Cretaceous sediments. The basement rocks are predominantly schists, gneisses, and granites, whereas the sedimentary units linked to the Anambra Basin are mainly composed of limestone, marble, and shale deposits (Sanni *et al.,* 2023).

3.2. LOCAL GEOLOGY

The study area, Ikpeshi, lies within the southwestern segment of the Nigerian Basement Complex. Geographically, it is located around latitude 7°07′59″N and longitude 6°12′25″E, and forms part of the Igarra Schist Belt in southwestern Nigeria. The lithological composition of the region is dominated by basement rocks such as migmatite complexes, gneisses, and schists. In certain locations within Ikpeshi, these crystalline basement rocks are overlain by sedimentary formations, notably limestone and marble, which constitute significant mineral deposits in the area. Additional mineral resources found in the region include quartz, feldspar, and various clay minerals (Sanni *et al.,* 2023).

3.3. LITHOLOGIC UNITS

3.3.1. Migmatite-Gneiss Complex

The Migmatite-Gneiss Complex represents the oldest and most widely distributed lithological unit in the Ikpeshi region. Similar to other parts of southwestern Nigeria, this complex comprises predominantly banded gneisses and migmatites, distinguished by alternating layers of felsic and mafic minerals. These rocks originated from high-grade metamorphic processes and partial anatexis, largely driven by the Pan-African tectonothermal events (Rahaman, 1988).

3.3.2. Granites and Granitic Gneisses

Granitic intrusions are frequently encountered within the migmatite-gneiss complex in the Ikpeshi area, forming prominent outcrops in several localities. These granites are typically coarse-grained and composed of quartz, feldspar, and biotite, with occasional hornblende. Their emplacement is associated with tectonomagmatic activities during the Pan-African orogeny, which played a significant role in the remobilization and injection of granitic bodies throughout the Nigerian Basement Complex (Oyewole and Ofuyah, 2017).

3.3.3. Schists Formations

Schist units in the study area belong to the metasedimentary series and are predominantly composed of muscovite and biotite schists. These rocks are medium-grade metamorphosed equivalents of shale and siltstone, transformed under conditions of regional metamorphism linked to the Pan-African orogenic cycle. In addition to their geological significance, these schists host economically valuable quartz veins, which have been subjected to artisanal and industrial extraction (Oyewole and Ofuyah, 2017).

3.3.4. Cretaceous Sedimentary Units

Certain sections of Ikpeshi are underlain by sedimentary deposits overlying the crystalline basement. These sedimentary rocks, primarily comprising limestone and marble, represent the southern reach of the Anambra Basin. The Cretaceous limestone and marble deposits in the area are of notable quality and have become central to the development of local cement production and marble processing industries (Sanni *et al.,* 2023).

3.4. MATERIALS

The materials and equipment used in the study included soil sampling equipment, a GPS device, sample containers, deionized water, personal protective equipment (PPE), acid-washed polyethylene bags, laboratory reagents (nitric acid and hydrogen peroxide), an atomic absorption spectrophotometer (AAS), a pH meter, an analytical balance, a digestion apparatus, data analysis software, statistical software (SPSS).

3.5. METHODS

3.5.1 Soil Sample Collection and Preparation

Soil samples were collected from ten (10) strategically selected points within the vicinity of the mining area. The sampling points were designated as SL1, SL2, SL3, SL4, SL5, SL6, SL7, SL8, SL9, and SL10. These points were positioned at successive distances of 10m, 20m, 30m, 40m, 50m, 60m, 70m, 80m, 90m, and 100m, respectively, from the mining site. At each of these locations, composite soil samples were carefully obtained for analysis., approximately 50g (grams) of soil were collected using a hand auger at a depth of 0–15cm. The samples were then sealed in clean polyethylene bags to prevent contamination and stored for subsequent laboratory analysis. In the laboratory, the soil samples underwent air-drying and were subsequently sieved through a mesh with an aperture size of less than 0.25mm. To eliminate residual moisture, the sieved samples were then heated in an electric oven at approximately 40°C for 30 minutes. The resulting fine powdered samples were stored in desiccators prior to heavy metal assessment.

3.5.2 Determination of Heavy Metal Concentrations

The concentrations of selected heavy metals; Lead (Pb), Chromium (Cr), Copper (Cu), Cadmium (Cd), Cobalt (Co), Nickel (Ni), Vanadium (V), Zinc (Zn), Selenium (Se), and Arsenic (As) were determined using atomic absorption spectrophotometry (AAS), following the methodology described by (Itam *et al.,* 2024). In addition to quantifying the metal concentrations, several pollution assessment indices were employed to evaluate contamination levels. Furthermore, both carcinogenic and non-carcinogenic health risks associated with the presence of these metals were examined using established risk assessment models.

3.5.3 Pollution Assessment Indices

To comprehensively evaluate the extent of heavy metal contamination in the soil, the study employed a suite of pollution indices. These included the Contamination Factor (Cf), Degree of Contamination (Cd), Geo-accumulation Index (Igeo), and Pollution Load Index (PLI). Each of these indicators provides a different dimension of assessment, ranging from individual metal impact to cumulative pollution load.

3.5.4 Contamination Factor (Cf)

The contamination factor (Cf), which quantifies the level of contamination for each individual metal relative to background concentrations, was calculated using the method proposed by (Hakanson, 1980). This index serves as a critical component in identifying metals with the highest ecological and health risk potential at the study sites.

$$Cf = C_{metal} / C_{background}$$

Where: Cf = contamination factor, $Cf < 1$ = low contamination factor, $Cf \leq 1 < 3$ = Moderate contamination factor, $3 \leq Cf < 6$ = considerable contamination factor, $Cf \geq 6$ = Very high contamination factor.

3.5.5. Degree of contamination (Cd)

The degree of contamination (Cd) is calculated following the method proposed by (Hakanson, 1980), using the equation provided below;

$Cd = \sum C^{fI}$

Where: $Cd<7$ = low degree of contamination, $7 \leq Cd < 14$ = moderate degree of contamination, $14 \leq Cd < 28$ = considerable degree of contamination, $Cd \geq 28$ = very high degree of contamination.

3.5.6. Pollution Load Index (PLI)

The pollution load index (PLI) for heavy metal analysis is determined using the formula described by (Tomlinson *et al.*, 1980), as shown below.

$PLI = (CF_1 \times CF_2 \times CF_3 \times ....CF_n)^{1/n}$

Where: $PLI<1$ = No metal pollution, $PLI = 1$ = Baseline levels of pollutants, $PLI >1$= indicates a polluted condition.

3.5.7. Geo-accumulation Index (Igeo)

The geo-accumulation index (Igeo) was computed following the method defined by (Muller, 1969).

$I_{geo} = \log_2 (C_n/1.5 \times B_n)$

C_n denotes the observed concentration of a specific metal n in the soil, whereas B_n represents the corresponding reference or geochemical background concentration of that metal, expressed in milligrams per kilogram (mg/kg). To accommodate potential fluctuations in natural background levels and slight anthropogenic contributions, a constant factor of 1.5 is incorporated into the calculation. The geo-accumulation index (*Igeo*) is interpreted according to the following classification scheme:

Class 0: Igeo < 0 = practically uncontaminated, Class 1: 0 < Igeo < 1 = uncontaminated to moderately contaminated, Class 2: 1 < Igeo < 2 = moderately contaminated, Class 3: 2 < Igeo < 3 = moderately to heavily contaminated, Class 4: 3 < Igeo < 4 = heavily contaminated, Class 5: 4 < Igeo < 5 = heavily to extremely contaminated, Class 6: Igeo > 5 = extremely contaminated.

3.5.8. Ecological Risk Factors and Index

The ecological risk factor for both the control soil (collected from outside the mining site) and the soils sampled at distances of 20m – 90m from the mining site was determined using the methodology outlined by (Hakanson, 1980).

$Er = TR \times CF$

$RI = \sum E_r$

Where: Er = ecological risk factor, TR = toxicity of heavy metals, CF = pollution factor, RI = ecological risk index, RI< 150 <300 = low risk, 150 ≤ RI = moderate Risk, 300≤ RI <600 = considerable risk, IR ≥600 = very high risk.

3.5.9. Determination of the Average Daily Dose

The average daily dose was estimated by considering the various human exposure pathways, namely ingestion, inhalation, and dermal contact. This evaluation was designed to assess both carcinogenic and non-carcinogenic health risks resulting from exposure to heavy metals, based on the approach described by (Zglobicki and Telecka, 2021). The corresponding risk values were calculated using the following equations:

$$\mathrm{ADD}_{\mathrm{ingestion}} = \frac{\mathrm{C} \times \mathrm{R}_{\mathrm{ing}} \times \mathrm{EF} \times \mathrm{ED}}{\mathrm{BW} \times \mathrm{AT}} \times 10^{-6} \quad \text{.......} \quad (1)$$

$$\mathrm{ADD}_{\mathrm{inhalation}} = \frac{\mathrm{C} \times \mathrm{R}_{\mathrm{inh}} \times \mathrm{EF} \times \mathrm{ED}}{\mathrm{PEF} \times \mathrm{BW} \times \mathrm{AT}} \quad \text{.......} \quad (2)$$

$$\mathrm{ADD}_{\mathrm{dermal}} = \frac{\mathrm{C} \times \mathrm{SA} \times \mathrm{SL} \times \mathrm{ABF} \times \mathrm{EF} \times \mathrm{ED}}{\mathrm{BW} \times \mathrm{AT}} \times 10^{-6} \quad \text{.......} \quad (3)$$

Therefore, the total ADD can be evaluated by adding the ADD dermal +ADD inhalation + ADD ingestion.

Where C = the Concentration of metal (e.g. µg/kg, mg/kg); Ring = ingestion rate, Rinh = inhalation Rate, EF is Exposure Frequency (days/yr); SA = exposed skin area, ABF = the exposure duration (h/day), SL = the chemical-specific dermal permeability constant (cm/h) ED = Exposure Duration (yr); AT = Averaging Time (period over which exposure is averaged) (days); BW = Body Weight (kg).

3.5.10. Determination of Non-Carcinogenic Risk

This evaluation focuses on quantifying the potential health risks associated with exposure to non-carcinogenic substances. The hazard quotient (HQ) is determined by dividing the estimated exposure dose by the oral reference dose (RfD), expressed in mg/kg/day. An HQ value greater than one suggests a possible risk of adverse health effects due to exposure to the contaminant.

HQ = ADD/RFD

Furthermore, the Hazard Index (HI) is employed to assess the cumulative non-carcinogenic risk arising from simultaneous exposure to multiple heavy metals present in the analyzed samples. As outlined by the United States Environmental Protection Agency (USEPA), the HI is derived by summing the individual Hazard Quotients (HQs) for each metal. The HQ itself quantifies the risk from a specific exposure pathway and is categorized as follows: HQ_{inh} for inhalation, HQ_{ing} for ingestion, HQ_{derm} for dermal absorption, and HQ_{t} representing the total hazard quotient from all exposure routes. The HI reflects the aggregated risk from all pathways and metals combined. An *HQ* value exceeding 1 indicates a potential for adverse health effects, whereas a value below 1 suggests negligible risk. Similarly, the interpretation of *HI* is as follows:

- $HI < 1$ implies an insignificant health risk,
- $1 \leq HI < 4$ suggests the possibility of risk, and
- $HI > 4$ indicates a high level of non-carcinogenic health risk.

3.5.11. Carcinogenic Risk Assessment

Carcinogenic risk (CR) was evaluated using the approach described by (Maeaba *et al.,* 2019) to estimate the probability of developing cancer over a lifetime as a result of exposure to carcinogenic heavy metals, including arsenic (As), nickel (Ni), lead (Pb), cadmium (Cd), cobalt (Co), and chromium (Cr). The following equations were applied to estimate carcinogenic risk across the different exposure pathways:

$CR = ADD \times SF$……………………………………………………………........(1)

Where CR= Cancer risk, ADD = Average daily dose and SF = Cancer slope factor

$TCR = \sum CR = CR_{ing} + CR_{inh} + CR_{derm}$………………………………..……………(2)

Where *TCR* = Total Carcinogenic Risk (unitless), *CR* = the individual Carcinogenic Risk (unitless). According to established guidelines, a *TCR* value within the range of 1×10^{-6} to 1×10^{-4} is considered acceptable, signifying no substantial threat to human health (Itam *et al.,* 2024). In contrast, values equal to or exceeding 1×10^{-3} may indicate a heightened probability of cancer development over a lifetime of exposure.

3.4.12 Statistical Analysis

The collected data, including the estimated parameters, were subjected to statistical analysis to determine their mean and standard deviation. This analysis was conducted using the Statistical Package for the Social Sciences (SPSS).

CHAPTER FOUR

RESULTS

The results of the heavy metal analysis conducted on ten (10) soil samples collected at a depth of 0-15 cm across the study area are presented in tables 4.0.1 to 4.0.8, which detail the mean concentrations, standard errors, minimum and maximum values, and associated health risk metrics. The concentrations of nine heavy metals; Iron (Fe), Zinc (Zn), Copper (Cu), Lead (Pb), Cadmium (Cd), Manganese (Mn), Nickel (Ni), Chromium (Cr), and Cobalt (Co) were quantified and evaluated. To assess the potential health implications of these contaminants, the measured concentrations were compared against EPA RSL standard. The analysis included the Contamination factor (CF), Degree of Contamination (Cd), Pollution Load Index (PLI), Geo-accumulation Index (Igeo), Ecological Risk Factor (Eri), Potential Ecological Risk Index, Average Daily Dose (ADD), Hazard Quotient (HQ), Lifetime Average Daily Dose (LADD), Carcinogenic Risk (CR) and Total Carcinogenic Risk (TCR).

Table 4.0.1: concentration of heavy metals in soil samples (mg/kg) and comparison with U.S. EPA standard

Metal	**Mean (mg/kg)**	**Std. Error (mg/kg)**	**Min (mg/kg)**	**Max (mg/kg)**	**EPA RSL (mg/kg)**
Fe	52.885	1.533	47.74	60.98	-
Zn	25.411	1.124	20.49	32.22	370
Cu	21.859	0.953	16.22	26.18	28
Pb	2.523	0.097	2.02	2.95	200
Cd	0.744	0.077	0.38	1.1	7.1
Mn	6.401	0.15	5.84	7.18	28
Ni	8.464	0.151	7.98	9.43	-
Cr	5.425	0.111	4.99	5.93	-
Co	2.413	0.04	2.14	2.54	0.27

Table 4.0.2: Contamination Factor (Cf) of mean concentrations

Metal	**Mean Concentration (mg/kg)**	**Cf (Mean/Bg)**
Fe	52.885	0.00112
Zn	25.411	0.26748
Cu	21.859	0.48576
Pb	2.523	0.12615
Cd	0.744	2.48000
Mn	6.401	0.00753
Ni	8.464	0.12447
Cr	5.425	0.06028
Co	2.413	0.12700

Table 4.0.3: Degree of Contamination (Cd) and Pollution Load Index (PLI)

Metric	**Value**
Degree of Contamination (Cd)	3.67979
Pollution Load Index (PLI)	0.08831481

Table 4.0.4: Geo-accumulation Index (Igeo) mean concentrations

Metal	**Mean Concentration (mg/kg)**	**Igeo**
Fe	52.885	-10.38668
Zn	25.411	-2.48744
Cu	21.859	-1.62666
Pb	2.523	-3.57175
Cd	0.744	0.72538
Mn	6.401	-7.63798
Ni	8.464	-3.59109
Cr	5.425	-4.63719
Co	2.413	-3.56206

Table 4.0.5: Ecological Risk Factor (Eri) per metal and Potential Ecological Risk Index (RI)

Metal	**Er^i**	**Tr (toxic-response factor)**
Fe	0.00112	1
Zn	0.26748	1
Cu	2.42878	5
Pb	0.63075	5
Cd	74.40000	30
Mn	0.00753	1
Ni	0.62235	5
Cr	0.12056	2
Co	0.63500	5
RI (sum of Eri)	79.11357	

Table 4.0.6: Average Daily Dose (ADD) and Hazard Quotient (HQ)

Metal	**Mean ADD children (mg/kg-day)**	**Mean HQ children**	**Mean ADD adults (mg/kg-day)**	**Mean HQ adults**
Fe	6.762×10^{-4}	0.000966	7.244×10^{-5}	0.000103
Zn	3.249×10^{-4}	0.001083	3.481×10^{-5}	0.000116
Cu	2.795×10^{-4}	0.006987	2.994×10^{-5}	0.000749
Pb	3.226×10^{-5}	0.009216	3.456×10^{-6}	0.000987
Cd	9.512×10^{-6}	0.009512	1.012×10^{-6}	0.001019
Mn	8.184×10^{-5}	0.000585	8.768×10^{-6}	0.000063
Ni	1.082×10^{-4}	0.005411	1.159×10^{-5}	0.000580
Cr	6.936×10^{-5}	0.000046	7.431×10^{-6}	0.000005
Co	3.085×10^{-5}	0.000717	3.305×10^{-6}	0.000077

Table 4.0.7: Lifetime Average Daily Dose (LADD) and Carcinogenic Risk (CR)

Metal	**Mean LADD children (mg/kg-day)**	**CSF (mg/kg-day)**	**Mean CR children**	**Mean CR adults**
Fe	5.796×10^{-5}	-	-	-
Zn	2.785×10^{-5}	-	-	-
Cu	2.396×10^{-5}	-	-	-
Pb	2.765×10^{-6}	-	2.350×10^{-8}	1.259×10^{-8}
Cd	8.153×10^{-7}	6.30000	5.137×10^{-6}	2.752×10^{-6}
Mn	7.015×10^{-6}	-	-	-
Ni	9.276×10^{-6}	0.00084	7.792×10^{-9}	4.174×10^{-9}
Cr	5.945×10^{-6}	0.50000	2.973×10^{-6}	1.592×10^{-6}
Co	2.644×10^{-6}	-	-	-

Table 4.0.8: Total Carcinogenic Risk (TCR) per sample

Sample	**TCR children**	**TCR adults**
S1	1.04×10^{-5}	5.597×10^{-6}
S2	8.926×10^{-6}	4.782×10^{-6}
S3	7.180×10^{-6}	3.846×10^{-6}
S4	7.530×10^{-6}	4.034×10^{-6}
S5	7.969×10^{-6}	4.269×10^{-6}
S6	6.651×10^{-6}	3.563×10^{-6}
S7	5.437×10^{-6}	2.912×10^{-6}
S8	6.697×10^{-6}	3.588×10^{-6}
S9	1.087×10^{-6}	5.823×10^{-6}
S10	9.700×10^{-6}	5.1964×10^{-6}

CHAPTER FIVE

DISCUSSION, CONCLUSION AND RECOMMENDATION

5.1 DISCUSSION

5.1.1. Carcinogenic Risk Assessment

Carcinogenic risk (CR) estimates the probability of a person developing cancer over a period due to exposure to carcinogenic contaminants (U.S. EPA, 2011). The results of the carcinogenic risk assessment for ingestion exposure to soil contaminants are presented in Table 4.0.7. The analysis focused on four recognized carcinogenic heavy metals lead (Pb), cadmium (Cd), nickel (Ni), and chromium (Cr), all of which are known to pose varying degrees of cancer risk to humans through oral exposure routes (Hu *et al.,* 2012). For children, the mean CR values ranged from 7.79×10^{-9} for Ni to 5.14×10^{-6} for Cd, while for adults, values ranged from 4.17×10^{-9} for Ni to 2.75×10^{-6} for Cd. Specifically, chromium (Cr) recorded CR values of 2.97×10^{-6} for children and 1.59×10^{-6} for adults, whereas lead (Pb) exhibited comparatively lower CR values of 2.35×10^{-8} for children and 1.26×10^{-8} for adults. When aggregated, the total carcinogenic risk (TCR) values were 8.16×10^{-6} for children and 4.61×10^{-6} for adults, respectively. According to the U.S. EPA (2011) guidelines, acceptable lifetime carcinogenic risk levels typically range between 1×10^{-6} and 1×10^{-4}. Consequently, the obtained TCR values for both adults and children fall within the permissible range, implying that while the overall cancer risk is not alarmingly high, long-term exposure could still present a potential health concern. Notably, both cadmium (Cd) and chromium (Cr) displayed CR values approaching the upper limit of the acceptable risk range, indicating these two metals as the dominant carcinogenic contributors in the studied soils.

The carcinogenic risk trend for ingestion exposure followed the order Cd > Cr > Pb > Ni. This trend highlights cadmium as the most influential carcinogenic agent, largely due to its high carcinogenic slope factor (CSF = 6.3 mg/kg-day^{-1}) and strong bioaccumulation tendency.

Chromium also showed carcinogenic potential. Lead and nickel, on the other hand, contributed minimally to the overall cancer burden, remaining well within the lower end of the risk spectrum (Thompson *et al.,* 2023). The slightly higher TCR observed in children compared to adults reflects greater susceptibility among children to carcinogenic effects (USEPA, 2011).

5.1.2. Ecological Risk Assessment

The ecological implications of heavy metal contamination were evaluated using the Ecological Risk Factor (Er^i) and the Potential Ecological Risk Index (RI), as presented in Table 4.0.5. The individual ecological risk factor (Er^i) integrates the contamination factor (Cf) with the metals toxic-response factor (Tr) to estimate potential ecological hazards. The Er^i values obtained for the metals followed the order Cd > Cu > Pb > Co > Ni > Cr > Zn > Mn > Fe, with cadmium showing a distinctly high-risk value ($Er^i = 74.4$), signifying a considerable ecological threat. In contrast, other metals exhibited Er^i values far below 40, representing low ecological risk levels (Tian *et al.,* 2020; Li *et al.,* 2025).

The cumulative ecological risk index (RI) obtained by summing all individual Er^i values was 79.11, which falls within the low ecological risk category (RI < 150) according to Hakanson (1980) classification. This indicates that the soil in the study area poses low potential ecological risk, although the elevated contribution of cadmium suggests localized hotspots or anthropogenic sources. The disproportionately high Er^i value of cadmium reflects its high toxicity coefficient (Tr = 30) and persistent bioaccumulation tendency in soil ecosystems (Tian *et al.,* 2020).

As presented in Table 4.0.4, the Igeo values for the metals ranged from –10.38668 (Fe) to 0.72538 (Cd). Among the analyzed metals, cadmium (Cd) is the only element with a positive Igeo value (0.725), indicating moderate contamination, while all other metals (Cr, Ni, Pb, Co, Cu, Zn, Mn, and Fe) exhibit negative Igeo values, signifying unpolluted to practically uncontaminated conditions (Navarrete-Rodríguez et al., 2020). This suggests that Cd enrichment is primarily of anthropogenic origin, potentially arising from activities associated with sand mining.

5.2. CONCLUSION

The carcinogenic and ecological risk assessments collectively reveal that while the soils in the study area exhibit generally low health and ecological risks, cadmium (Cd) and chromium (Cr) represent the dominant contributors to potential carcinogenic hazards. The total carcinogenic risk (TCR) values for both adults and children fall within the U.S. EPA's acceptable risk threshold, suggesting that cancer risks from long-term exposure remain tolerable but not negligible. The higher susceptibility observed in children reveals the heightened vulnerability of this group to heavy metal toxicity. The ecological risk index (RI) further indicates a low overall ecological threat. However, the elevated Er^i of cadmium points to localized contamination hotspots, likely influenced by sand mining.

5.3 RECOMMENDATIONS

- To mitigate potential human and ecological health impacts, periodic environmental monitoring should be implemented to track metal accumulation trends, especially for cadmium and chromium.
- Strict regulation of mining and waste disposal practices is essential to minimize further contamination.
- Phytoremediation and soil amendment strategies can be introduced to immobilize or extract toxic metals from affected soils.
- The public should be made aware of the potential exposure pathways, with special attention to protecting children and agricultural zones in proximity to mining areas.

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