Reutilization and Commercialization of Stormwater Pond Sediments

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Abstract

Sediment accumulation in stormwater ponds will gradually degrade their hydraulic and water quality functions. Landfill disposal is a common way of removing accumulated sediments. However, given the vast number of deposits created, it is unsustainable. This research aims to find possible applications for stormwater pond sediments and sludge via an examination of sediment reuse studies and practices conducted across the globe. Several journal articles, regulations, and guideline papers in Canada, the United States, Singapore, China, and Europe were reviewed. Numerous essential issues are discussed, including sediment characterization, reuse possibilities, potential sustainability, economic advantages, risk-based contamination levels for different case studies, the significance of the regulatory framework, and treatment procedures. After a series of lab tests, multiple beneficial compounds in noticeable concentrations were found in the pond sediment from industrial and residential areas. They have pharmacological, commercial, and industrial applications, including the manufacture of insulating materials or medicinally and as an antioxidant in vegetable oils. Besides, biotech products have also been developed and prepared with tremendous potential for use in the field.

Furthermore, multiple leachable heavy metals are in sediments from stormwater ponds. A few request attention and amendment before applying to land recovery(e.g. wastelands of the mining industry). Ultimately, in developing construction materials, gypsum is an appropriate binder, and the optimal percentage of gypsum in Bio-concrete is 10%. The strongest Bio-concrete should have an unconfined compressive strength of 35.93MPa. This article will give an overview of sediment reuse strategies in Auckland Council as a first step towards expanding current knowledge on the subject to achieve the goal of zero landfills by 2040.

Key Words: Reutilization, Commercialization, Sediments, Stormwater Pond

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1 INTRODUCTION

The ongoing process of urbanisation leads to a corresponding escalation in the impacts on natural cycles and the ecosystem. Urban stormwater management encompasses a range of strategies aimed at alleviating the adverse effects on the receiving environment resulting from modifications in land use, which have the potential to disrupt the natural flow patterns in watercourses located downstream. The incorporation of stormwater devices, such as ponds, wetlands, overland flow routes, and pipe systems, is a widely accepted approach in the construction and management of stormwater networks. These devices serve multiple functions, including the temporary storage of excess water (detention), the long-term storage of water (retention), and the improvement of water quality. These devices are designed to collect runoff that transports various forms of debris, sediments, and contaminants originating from catchment areas located upstream.

In order to uphold the optimal performance of these gadgets, it is necessary to do routine maintenance. The financial burden of stormwater pond upkeep can be significant, particularly in densely populated urban areas with extensive development, where a substantial amount of silt removal is required. There are currently around 600 stormwater ponds in operation within the Auckland region.

The sediments obtained from ponds are then disposed of in landfills, thereby occupying important land resources. The aforementioned justifications serve as the driving force behind

the initiation of this pilot study, which aims to examine the possible commercial applications of sediments obtained from stormwater ponds located in the Auckland Region of New Zealand.

While government policy directions may undergo changes over time, it is imperative that the rules remain consistent with the prevailing government policy directions. The comprehensive delineation of government waste policy may be found in the New Zealand Waste Strategy, as outlined by the Ministry for the Environment in 2007. The aforementioned plan was established by the Ministry of the Environment in conjunction with the Local Government of the United Kingdom.

This study is centred around the examination of the enhanced worth of sediments, encompassing the retrieval of diverse constituents and the development of commercially viable goods. This study examines the production of several value-added commodities, including soil amendments, building aggregates, fuels, biotechnology products, biosorbents, and other items. This study examined journal articles, regulations, and guideline papers from several regions, including Canada, the United States of America, Singapore, China, and Europe. The paper addresses several important topics, such as the characterization of sediment, potential for reuse, establishment of risk-based contamination thresholds for various case studies, the importance of regulatory frameworks, and treatment methodologies.

The present study examines the advantages and disadvantages of existing sediment treatment methodologies in order to address notable issues such as limitations in production techniques, advantages, and practical applications in the field. Additionally, it offers a wider range of opportunities for the future usage and commercialization of sediments. The study examines the advantages and disadvantages of existing sediment treatment approaches in order to address notable issues, including limitations in production techniques, potential benefits, and practical uses in the field. Additionally, it offers a wider range of opportunities for the future usage and commercialization of sediments.

1.1 Objectives

The study aims to identify previous sediment reuse methods to develop a commercial sediment reuse mechanism for the Auckland Council.

To determine the potential reuses, the chemical compositions in the sediment must be first determined. Several samples were taken from ponds in Auckland and analysed in a

commercial laboratory. These ponds were selected to give a general representation of residential, industrial, and commercial land uses. The current legislation that may apply to recycling different types of products, possible solutions/treatment options, and provide recommendations were also assessed.

1.2 Research Questions

To achieve the research objectives, the following research questions have been defined to guide the study:

1. Can the sediment extracted from sediment ponds in Auckland be reused commercially?

2. What are the key factors that drive sediment reuse?

3. Are there any lessons from sediment reuse overseas that can benefit Auckland?

4. What are the main benefits of sediment reuse?

2 LITERATURE REVIEW

The available research publications regarding sediment reuse available in the literature were reviewed. The literature review in this section focuses on sediment treatment technology and current sediment re-utilisation technologies. Furthermore, the literature review covers the current approach to dealing with sediments in stormwater ponds, maintenance, landfills, etc. Also, the current approach's advantages and disadvantages (e.g., cost). Then what are the other methods tried around the world regarding sediment reuse?

Sediments and sludge from water treatment plants comprise inorganic particles, organic matter, bacteria, and water. Nitrogen, phosphorus, and potassium can be degradable, but there are also harmful non-biodegradable substances, such as heavy metals (such as copper, chromium, zinc), pathogens, parasites, polychlorinated biphenyls (PCBs), etc. (Tchobanoglous,2003)

2.1 Sediment treatment technology from urban sewage treatment plants and sediment ponds

There are various sediment treatment technologies available and currently in use. These techniques are discussed in the following sub-sections.

2.1.1 High-temperature composting

Composting is the best use of bacteria and fungi in the natural environment. The process is a microbial process that effectively allows these components to control biodegradable organic matter in sediments. (Anne Oppliger, Philippe Duquenne, 2016) Treated sediment can be directly transported to fields and other places as fertiliser. The air exhausted by the fan can be exhausted directly after deodorisation. The advantage of high-temperature composting is that high-temperature composting can treat a large amount of wastewater. However, the disadvantage is that composting covers a large area and causes severe soil pollution. (Andreoli, 2007).

2.1.2 Landfilling

Landfilling is the most direct and efficient way to deal with the sediments produced while treating a large quantity of urban wastewater. After the dehydration process of the sediment, then it is transported to landfills located away from cities and water sources with low soil permeability and then covered with an impermeable layer of high-density polyethylene. The advantage of this method is speed and low cost. However, the disadvantage is that less and less alternative land will be available for filling, and the treatment plant will continue to generate sediment. (Andreoli, 2007). However, landfilling has many disadvantages, such as heavy metal leaching and occupying land space.

2.1.3 Incineration

To begin with, incineration treatment can kill harmful organic matter in sediments and has little impact on the surrounding environment. Before incineration, the sediments must be dehydrated and dried. The incineration process efficiently removes feed organics, creates minimal odour (assuming that the flue gas treatment is good), and yields a stable and primarily recyclable solid ash product. The produced heat may be collected from the flue gas stream and either used directly or transformed into electricity.

However, incineration has some shortcomings. With sludge moisture concentrations below 33%, incineration becomes auto-thermal (i.e., supported by the heat produced by the combustion process). In addition, while most incinerators are energy-positive, the necessity to heat dry the sludge before burning affects the operation's overall energy balance. Incineration plants must be specially built near treatment plants. (Andreoli, 2007)

2.1.4   Thickening and dehydration

Thickening refers to reducing the water content of sediments and the volume of sediments. Compared to gravity concentration, mechanical concentration has the advantages of a small footprint, low initial cost, and no anaerobic phosphorus. Dewatering is the separation of adsorbed and capillary water in sediments into semi-solid or solid clays. After dehydration, the water content of the sediment will drop rapidly. Further drying is called sediment drying, and the moisture content of sediment after drying is less than 1/10. Dehydration methods mainly include mechanical dehydration, natural drying, and granulation.

Polymers such as flocculants and superabsorbent polymers (SAP) can speed up the dewatering process. Polyacrylamide (PAM) is a common chemical used to flocculate moist sediment. Moreover, anionic PAM is preferred because of its low toxicity and higher efficacy (Rocha & Van Seters, 2013). Sodium polyacrylate is one of the most often employed SAPs;

Figure 2-1*: SAP Powder Applied to Wet Sediment (TRCA &*
CH2M Ltd, 2016)

it absorbs water from wet sediment slurry to produce a gel-like substance that increases the volume of the material. According to TRCA & CH2M Ltd. (2016), both types of polymers may be utilized with mechanical excavation, whereas flocculant polymers can be utilized in line with hydraulic dredging, which drains sediment via geotextile bags. Both Figure 2.1 and Figure 2.2 depict SAP and PAM applications.

Figure 2-2: Left to Right - Flocculant Pumping System, Geotextile Bags, Sediment after Several Weeks (TRCA & CH2M Ltd, 2016)

2.1.5 Anaerobic digestion

Anaerobic sediment digestion is a commonly employed process for the degradation of sediment, wherein organic matter is broken down into carbon dioxide, methane, and water under anaerobic circumstances. One significant technological benefit associated with anaerobic digestion is the utilisation of biogas for power generation. This application has the potential to effectively decrease the concentration of organic matter in sediment and the amount of solid sediment. Enhance the sediment's dewatering capacity, mitigate its odoriferous nature, and decrease the expenses associated with subsequent treatment processes. Elevated temperatures have the potential to eradicate pests and diseases. The processing method is complicated, the anaerobic digestion time is extended, the area is large, the initial investment is high, and the operating cost is high. (Andreoli, 2007).

2.1.6 Aerobic fermentation

Aerobic sediment digestion, also known as aerobic composting, is a metabolic process by which aerobic thermophilic bacteria break down organic matter. Add oxygen to the sediment, which is oxidised and broken down into water, carbon dioxide, ammonia, etc. The breakdown of organic matter results in the production of heat, which can be utilised to effectively eliminate microorganisms, including harmful bacteria and parasite eggs, due to the high temperatures involved. Research conducted in the United States has demonstrated that the implementation of the Aerobic Thermophilic (AT) and Anaerobic Mesophilic (AM) systems, along with the use of a controlled anaerobic digester operating under the Solids Retention Time (SRT) system for a duration of 15 days, can successfully attain a volatile solids (VS) reduction exceeding 38%, hence satisfying the standards for volatility reduction. The allure of

mud-like vectors. The chemical oxygen demand (COD) of the supernatant resulting from the digestion of sludge is a substantial component of the organic load present in the influent of conventional wastewater treatment facilities. The implementation of the AT system in conjunction with mechanical dewatering of the digested sludge resulted in a decrease in the chemical oxygen demand (COD) present in the liquid portion of the digested sludge. This reduction corresponded to a decrease of roughly 10% in the COD load of the influent.

The reduction in Chemical Oxygen Demand (COD) was observed to be 55-59% in the sludge treated with the Anaerobic Membrane (AM) system, whereas the control anaerobic digester exhibited a reduction of 39-40%. It is noteworthy that these reductions were achieved when the identical feed sludge was utilised for all three systems. The mean concentration of chemical oxygen demand (COD) in the supernatant relative to the feed was determined to be 22,500 µg/L.

2.2 Issues of Current Reuse of Sludge

The utilisation of sludge on agricultural fields poses challenges as a result of the existence of potentially harmful metals, bacteria, and the potential for nitrate or phosphate contamination. The subsequent sections provide a concise discussion of these matters.

2.2.1 Toxic Metals

The utilisation of wastewater sludge in agricultural practises has the potential to result in soil contamination, phytotoxicity, and the accumulation of harmful metals within the food chain. The severity of the issue is contingent upon the interconnectedness of multiple factors, such as the composition of sludge in relation to its solid content and characteristics, the rate and regularity of treatments, the features of the soil, and the specific plant species involved. The availability of metal for plant absorption is determined by the chemical forms of metal, which in turn are affected by the kind of sludge and the interactions between sludge and soil. However, it should be noted that many factors such as different plant species and soil characteristics might have an impact on the uptake and accumulation of elements in agricultural crops. According to Chaney (1980),

There are three defensive mechanisms within the plant-soil system that serve to limit the presence of potentially harmful trace elements, namely metals, in the aboveground portions of plants. These processes play a crucial role in mitigating health hazards for both animals

and humans. Table 3.3 presents a comprehensive compilation of the anticipated surface sub-layers together with their corresponding pertinent features. The protective mechanism in question is commonly referred to as the "soil-plant barrier." According to Chaney (1980), The categorization of elements can be made based on their behaviour in soil and their accumulation in plants. These categories include:

1. Elements that exhibit insolubility in soil and do not accumulate in plants. This category encompasses elements such as lead (Pb), mercury (Hg), chromium (Cr), fluorine (F), silver (Ag), gold (Au), titanium (Ti), tin (Sn), silicon (Si), and zirconium (Zr).

2. Elements that are absorbed by plant roots but exhibit insolubility within the root or have limited translocation to the shoot, resulting in negligible accumulation in plants.

3. Elements that, when applied excessively, lead to phytotoxicity upon accumulation. Consequently, plants containing these elements cannot be safely consumed by humans or domestic animals. This category includes elements such as zinc (Zn), copper (Cu), nickel (Ni), cobalt (Co), manganese (Mn), arsenic (As), and boron (B).

Not all constituents included in wastewater sludges may be classified within the three categories. The potential concerns associated with the land application of wastewater sludges were noted by the Council of Agricultural Science and Technology of the United States Environmental Protection Agency in 1976. These risks include the presence of Cd, Cu, Mo, Ni, and Zn. These metallic elements have a propensity to accumulate within the tissues of plants, leading to reduced crop yields or potential health concerns for animals or humans who ingest these plants. According to Chaney (1980),

The levels of harmful metals present in sludges exhibit considerable variation throughout different cities, treatment plants within the same city, and even within a single treatment plant over different seasons (Metcalf & Eddy, 2003). Municipal sewage sludges contain a diverse range of metallic elements. The shape of the sludge and the metal is determined by their chemical features and the underlying chemistry, as discussed by Bradford et al. (1975). A progressive extraction technique is employed to fractionate the metals.

There is a scarcity of data regarding the interactions between metals present in wastewater sludge and soils. The existence of several soil qualities, including cation exchange capacity, pH levels, organic matter content, sesquioxide content, redox potential, texture, and the presence of other elements, has been found to have an impact on the uptake, solubility, and mobility of metals by plants (Su et al., 2004). In order for plants to utilise metals, it is necessary for them to undergo a process of conversion into forms that are soluble in water, capable of being exchanged, or able to be absorbed. The uptake of active metals in plants is influenced by several plant variables, such as uptake capacity. Consequently, these metals tend to accumulate in distinct plant sections, including the stem, leaves, roots, and fruits. The distribution of these metals in specific plant parts is determined by genetic regulation, as evidenced by studies conducted by Schmidt (1997) and Snyman et al. (1998).

2.2.2 Pathogens

Despite the absence of documented cases of illnesses or outbreaks, the presence of bacteria, viruses, and parasites commonly found in wastewater sludges poses a potential hazard. According to Schipper and Sparling (2000), the potential risk to public health from sludge-borne bacteria and viruses in land application systems is not significant, as these pathogens are not competitive in the external environment outside of their respective hosts. The survival of these organisms in the soil is contingent upon various environmental variables, including temperature, sunlight exposure, moisture levels, organic matter availability, soil pH, soil composition, and the presence of toxic substances and competing species. The majority of bacterial and viral diseases exhibit limited persistence in the soil. The absence of historical documentation of disease outbreaks resulting from the utilisation of secondary activated, anaerobically digested, or chemically treated sewage waste suggests that these microorganisms do not retain their ability to cause disease and sustain infectious populations within the soil ecosystem.

The parasite species that elicit the most substantial apprehension in the land application of municipal sludges can be identified based on the following distinguishing characteristics:

1) The generation of cysts or eggs (Hays, 1977); 2) The necessity for a duration of maturation in the surrounding environment (in the case of soil-transmitted parasites) or an intermediary host lacking specificity (Pahren et al., 1979); and 3) A direct transmission route involving the ingestion of faecal matter.

In the context of regulated land application systems, it is quite unlikely for bacteria or viruses to be transmitted to animals due to the inability of these microorganisms to survive outside their host for prolonged periods of time. The egg or cyst stage of certain nematodes and cestodes exhibits the greatest capacity for prolonged viability in an extrahost environment. Although sewage sludge may include certain pathogens, their presence does not necessarily imply the transmission of diseases to animals. Additional research is necessary to determine the pathogenicity of these organisms and assess the potential for a disease outbreak resulting from the controlled land application of treated municipal sewage sludges harbouring these organisms.

The issue of contamination by phosphate and nitrate compounds When nitrogen-rich sludge is applied to the soil, the soluble NH_4+ undergoes rapid reduction to NO_3-. The process of nitrification is hindered when the sludge contains a significant proportion of zinc, cadmium, or lead, whether present separately or in combination. According to Deleris et al. (2002), the presence of industrial sludge has a negative impact on the process of nitrification, but residential sludge has a negligible effect on nitrification. Meanwhile, alongside the utilisation of nitrification techniques, denitrification also presents itself as a viable alternative. From an agricultural perspective, denitrification is considered undesirable due to the loss of nitrogen (N). Nevertheless, denitrification offers environmental benefits due to the potential for NO_3- to infiltrate the soil profile and contaminate groundwater. When there is a deficiency of free oxygen but the existence of nitrate ions (NO_3-) and an energy source such as glucose or accessible carbon from raw sewage, certain microorganisms have the ability to utilise coupled oxygen, namely nitrate ions (NO_3-), as demonstrated by Metcalf and Eddy (2003).

$$C_6H_{12}O_6 + 4NO_3 = 6CO_2 + 6H_2O + 2N_2 + 4e-$$

The possibility for water pollution arises from the leaching of nitrate ions (NO_3-) when sludge is applied, which is a matter of significant concern. A concentration of 10 mg-N/L of NO_3 - is regarded as a potential hazard to human health. According to a study conducted by Correa et al. (2006), it has been demonstrated that the utilisation of compost on sewage sludge has led to a 50% decrease in the emission of mineral-N in sandy Spodosol soil and a 30% decrease in clayey Oxisol soil. Consequently, the levels of NO_3 and N in soils subjected to composted sludge incubation were determined to be less than 50% of the quantities recorded in soils incubated with digested sludge. A study was done to estimate the maximum monthly leaching of nitrate from the four different combinations of soil and sludge treatment.

The application of digested sludge at a higher nitrogen application rate resulted in a greater likelihood of nitrate leaching when compared to the application of the compost product.

2.2.3 Agricultural Application Studies

Numerous studies have investigated the phenomenon of plant uptake of metals in soils that have been modified with sludge. In a study conducted by Couillard and Grenier (1989), it was discovered that a notable and positive association exists between the quantity of phosphorus and nitrogen present in the tissues of a seedling and its growth rate. According to Wong's (1996) research, the presence of limestone and sewage sludge in soil has the potential to enhance the dry weight yields of Agropyron elongatum, while concurrently diminishing the uptake of zinc, copper, and cadmium by the soil. According to Miller et al. (1995), the incorporation of lime into acidic soils leads to a significant reduction in the levels of cadmium and zinc found in the plant tissue of lucerne.

According to the research conducted by Brallier et al. (1994), the application of lime resulted in a decrease in the uptake of cadmium, nickel, and zinc in most crops. In a similar vein, Chu and Poon (1999) conducted a study to examine the feasibility of producing plants using chemically modified sludge from a prototype chemically assisted primary treatment plant and anaerobically digested sludge from a secondary sewage treatment facility that has undergone stabilisation processes. Both forms of sludge underwent stabilisation through the process of anaerobic digestion. In this study, the plant species Agropyron elongatum, commonly referred to as tall wheatgrass, was employed. Regarding the chemically treated sludge samples, it was observed that the concentrations of metal pollutants and nutritional levels in both the shoot and root tissues of crops grown in the stabilised amended soil were significantly higher compared to those discovered in the control group. Nevertheless, the quantities of trace metals detected in the crops were well below the thresholds deemed hazardous. The results of the research investigations suggest that cultivating crops on a combination of native soil and treated sewage sludge is a viable option, given that the application rate is carefully managed and regulated.

Hansson and Frederiksson (2004) conducted a distinct investigation wherein they capitalised on the extensive aquatic presence of Phragmites australis in Sweden. The present study developed and evaluated appropriate technical and logistical approaches for the collection of

reed biomass from the lake, with the aim of utilising it as a nutrient source in organic crop cultivation. The action was undertaken in order to achieve the objectives outlined in the project. One examined approach was the fragmentation of the gathered material, followed by its direct application to crops. The potential utilisation of harvested biomass and wastewater treatment plant sludge as feedstock for biogas production, with the resulting byproduct being applied as fertiliser on agricultural fields, is worth considering. This represents the third option. The energy balances for the three distinct systems were calculated to be -4.05, -4.43, and +4.05 MJ/kg of harvested dry matter, respectively. The biogas production process has the potential to generate significant quantities of energy in the form of gas, while also providing easily absorbable nutrients to agricultural plants. The analysis revealed a caveat pertaining to the biogas option, namely that although its energy balance was reasonably favourable, the system's economics were susceptible to fluctuations in gas production income and reductions in operational expenditures.

2.2.4   Organic sediments reuse and possible commercialization

If the sediment in the pond (probably from a municipal sludge pond) is relatively organic, it could be dredged, dewatered, and then reutilised for agriculture, to improve soil and compost, to be used as fuel and biological products.
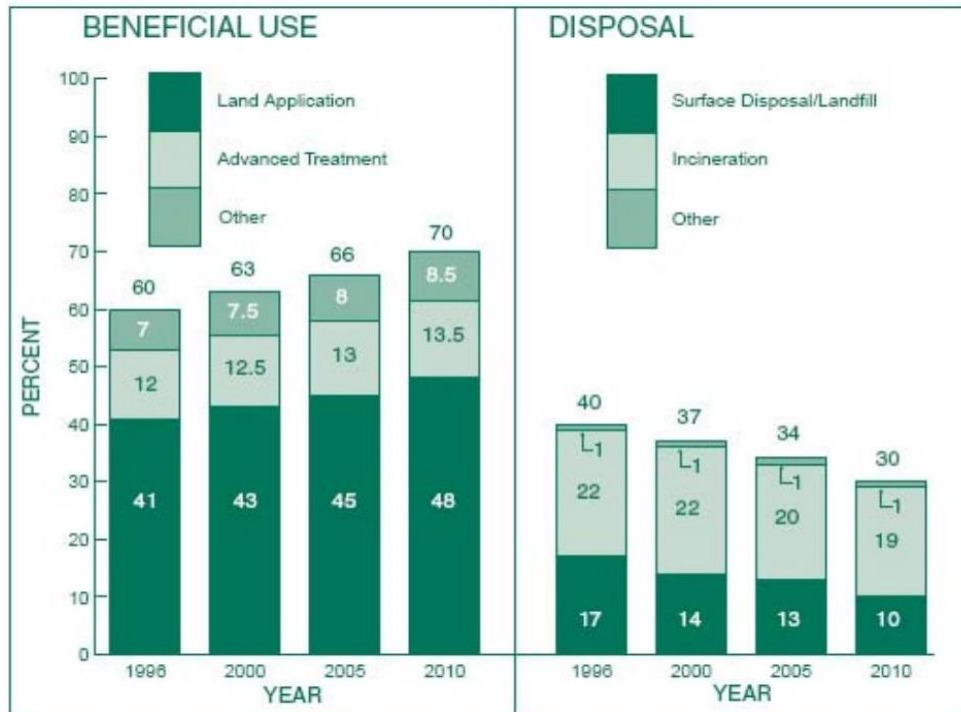
Figure 2-3: Projected production of biosolids used or disposed of in 2010 (EPA, 1999).

2.2.5   Agroforestry use of sediments

Table 2.1*:Exposure assessment routes. (Korentajer, 1991)*

Pathway Numbers	Pathway
1	Sludge-soil-plant-human
2	Sludge-soil-plant-home gardener
3	Sludge-soil-child
4	Sludge-soil-plant-animal-human
5	Sludge-soil-animal-human
6	Sludge-soil-plant-animal
7	Sludge-soil-animal
8	Sludge-soil-plant
9	Sludge-soil-soil biota
10	Sludge-soil-soil biota-predator of soil biota
11	Sludge-soil-airborne dust-human
12	Sludge-soil-surface water-fish-human

The existing practice of applying sludge to agricultural land is problematic owing to the presence of dangerous metals, microbes, and nitrate or phosphate contamination, as stated in Section 1.1.

The utilisation of sludge derived from sewage treatment plants has been observed in the agricultural sector, as well as in reforestation efforts, with a special emphasis on its application in nurseries. Nevertheless, the use of this application is limited as a result of the possible existence of hazardous compounds. The pollutants in question encompass heavy metals and persistent organic compounds, specifically organochlorine insecticides, polychlorinated biphenyls (PCBs), phenols, and hydrocarbons. Cadmium, lead, and mercury are among the heavy metals that can be classified as such (McGrath et al., 1994). Furthermore, it should be noted that sewage sludge has the potential to include a diverse array of pathogenic microorganisms, which have the ability to induce various diseases. Due to its high content of essential phytonutrients like as nitrogen, phosphorus, and other micronutrients, sewage sludge can be utilised as a fertiliser in agricultural contexts. Furthermore, sludge has the potential to serve as an organic modifier and a lime modifier, in addition to its other applications.

Pathogens can be eliminated by composting, although the process has a limited impact on hazardous heavy metals. The high organic matter content of the sludge ought to have a good impact on the physical qualities of the soil (Korentajer, 1991), and it is anticipated that this

would increase the populations of microorganisms and invertebrates. On the other hand, pollution load suggests long-term soil function and the possibility of dangers to soil nutrient webs through a variety of channels, as illustrated in Table 2.1.

Since the organic matter has not been entirely broken down, spreading raw sludge in agricultural or forestry practices is not suggested because it does not fulfil the requirements for fertilisation. Also, one must take into consideration whether or not the fresh sludge contains any bacteria that may be regarded as harmful. Since it is difficult to dispose of sludge, there is a need for the provision of temporary storage facilitieBothoth the European directive and the American EPA directive on the reuse of sewage sludge in agriculture require the stabilisation of sewage sludge prior to applying it to agriculture. These directives also place restrictions on several metal contaminants that can be present in the sludge (EPA, 1999). The neutralisation value that it brings to sewage sludge, specifically calcium and/or magnesium, allows for an important and effective treatment method that stabilises sludge and significantly increases the agronomic value of sewage sludge. This is accomplished thanks to the fact that it brings calcium and/or magnesium to the sewage sludge. Nevertheless, due to the fact that the number of applications (0-5) that may be made with these products is restricted due to agricultural cycles, weather conditions, and restrictions, they often need for extended periods of time in treatment plants (up to 8 months) in the encircling space (Aarab et al., 2006).

Powdered fuel ash (PFA) is an additional material that can be employed for the purpose of stabilising sludge prior to its recycling. PFA possesses significant value as a material for various applications, including land reclamation, soil conditioning, landfill cover, and landfill engineering. Lime has been identified as the predominant material employed for this particular objective, as evidenced by studies conducted by Wong (1995), Dirk (1996), Wong and Fang (2000), Chan et al. (2002), and Wong and Selvan (2006). Particulate Fly Ash (PFA) is a solid residual substance generated as a byproduct in power plants that utilise coal as a primary fuel source. According to Poon and Boos (1996), the inclusion of PFA in sewage sludge, following adjustments for CaO, exhibits a noteworthy capacity to reduce total coliform levels and mitigate the leaching of heavy metals in the stabilised sludge.

Sediments can be used to improve soil and compost. Soil depletion due to over-planting or abuse of mineral fertilisers should be of special concern due to the rising number of problems afflicting the planet since it destroys non-renewable resources that are becoming rare today. This challenge, however, may be alleviated by making proper use of the many types of organic leftovers created daily by human activities, which are otherwise difficult to handle in an ecologically friendly manner.

2.3   Use of sediments as fuel

There are various uses of sediment, such as oil, and gas. These techniques are discussed in the following sub-sections.

2.3.1   Oils

The amount of oil recovered in secondary pollution incineration operations is small. When compared to incineration, this alternative method demonstrated enhanced efficiency, lower operating costs, reduced emissions of NOx and SOx, and operated at lower temperatures (Avenell et al., 1996). Typically, the pyrolysis of sewage sludge is conducted within a range of moderate temperatures, specifically between 300 to 600°C. This process takes place in fluidised, stationary, and revolving reactors, with varying gas residence durations ranging from 1.5 to 3.5 seconds. The study conducted by Cassidy et al. (1998) documented the process of pyrolysis applied to dewatered and dried sludge. The non-reactive component of the sludge underwent a conversion into a substance resembling coke when subjected to temperatures ranging from 400 to 700 °C in an oxygen-depleted environment. Subsequently, the condensed converted substance derived from the organic component can likewise be employed to produce elecfuel, as well as serving as a viable source material for the manufacturing of activated carbon. According to scholarly research, sludge is a type of fuel that undergoes thermal transformation, resulting in both liquid and solid forms. The process of converting organic compounds derived from sludge into incinerator oil by the use of solvents, air pressure, and temperatures ranging from 200 to 300°C is a widely recognised procedure referred to as "Sludge Fuel" (STF) (Milot et al., 1989). In conjunction with an elevated temperature of around 10 MPa, as reported by Itoh et al. (1994) and Boon & Thomas (1996). 'Enersludge', a product developed by Environmental Solutions International Ltd. located in Burswood, Australia, is derived from the process of dewatering sludge, followed

by its subsequent drying. In a subsequent reactor, the catalytic conversion of organic molecules into hydrocarbons takes place within the sludge.

The oil-to-oil process (OFS) was developed and implemented in 1982 at the Canadian Centre for Wastewater Technology, which is affiliated with Environment Canada. The origin of the OFS can be traced back to a research study conducted at the University of Tübingen in Germany. The overall process of the oxidative fusion synthesis (OFS) consists of two distinct stages, namely solution heating and catalytic conversion. According to Kyriakos (1990) and Hudson & Lowe (1996), charcoal is the substance in which oil is formed. When the Tubingen process is carried out at temperatures as low as around 300 degrees Celsius, it results in an oil content ranging from 20 to 30 percent per kilogramme of dry sludge. Based on the findings of multiple experiments, it has been concluded that the oils obtained from the pyrolysis of sewage sludge demonstrate favourable attributes for direct application in diesel engines. The research undertaken by Campbell (1990) and Werther and Ogada (1999) provides evidence that these oils exhibit a level of quality similar to that of lower-grade petroleum distillates produced by commercial refineries.

2.3.2 Gas

Traditionally, lignocellulosic biomass has been used to produce environmentally friendly, low-yield biofuels. However, a recent study pointed out that using untreated sewage sludge as microbial seeds could facilitate the production of biohydrogen from lignocellulose solid waste, resulting in improved recovery rates.

2.4 Use of biological products and possible commercialization

There are various uses of biological products and possible commercialization available and currently in use, such as bioplastics, biosurfactants, biosorbents, and biopesticides and enzymes. These techniques are discussed in the following sub-section

2.4.1 Bioplastics

An further approach to the recovery of sludge involves the utilisation of activated sludge for the production of bioplastics. Polyhydroxyalkanoates (PHA) refer to polymers composed of hydroxy alkanoate units, which are known to accumulate as carbon/energy or serve as materials for energy storage in many microorganisms. In the presence of alternating periods of nutrient abundance and scarcity, bacterial populations adopt a survival mechanism wherein

they accumulate a significant portion of their soluble substrates, if available, in the form of storage polymers, such as polyacid-beta-hydroxybutyric (PHB). In situations characterised by uneven growth, the aforementioned mechanism functions as a means of regulating the redox status of heterotrophic cells through the spillover of nicotinamide adenine dinucleotide (Senior and Dawes, 1973; Van Niel et al., 1995). Biopolymers have been identified as potential materials for the production of bioplastics (Kessler et al., 2001; Nonato et al., 2001; Kim & Lenz, 2001). The primary factor hindering the widespread adoption of PHAs in the plastics sector is the escalated cost associated with the selection of microorganisms and substrates (Anderson & Dawes, 1990). Most processes for generating polyhydroxyalkanoates (PHA) include the use of pure cultures of bacteria, such as Ralstonia eutropha, in a synthetic medium that lacks particular nutrients (Doi et al., 1987; Lee, 1996a). Nevertheless, more cost-effective alternatives have replaced the previous method of utilising synthetic media. According to previous studies conducted by Satoh et al. (1998) and Takabatake et al. (2002), it has been demonstrated that the PHA content of activated sludge may be increased up to 62% by the implementation of a cost-effective anaerobic sludge technology. The utilisation of mixed culture in this technology renders it more economically efficient compared to conventional pure systems. Numerous supplementary studies have investigated the utilisation of industrial food waste as a viable nutrient source for microorganisms. Bench tests have demonstrated that microorganisms present in municipal activated sludge effectively engage in the manufacture of polyhydroxyalkanoates (PHAs) by utilising nutrients derived from malt and soy waste. According to the study conducted by Yu et al. (1999). The advantages of manufacturing PHA in mixed cultures over pure cultures (greater than 88 per cent dry cell weight) (Lee, 1996b) would include better economics, easier process control, no requirement for single treatment sterilisation, and improved waste management (Satoh et al., 1998). PHB builds up in vast volumes during bacterial development, although it develops relatively slowly in comparison to the time it takes for bacteria to proliferate. The addition of agro-industrial sewage to municipal sludge increases the formation of bioplastics. PHB may also be used as an economical carbon substrate to produce bioplastics and as a beneficial carbon source for low C: N effluent (Third et al., 2003). When photosynthetic bacteria were utilised in anaerobic activated sludge, Sawayama et al. (1999) found that PHB production on acetate medium and PHB concentration on dry biomass was around 6.6-14 µg/L/d and 15.1-25.3%, respectively. PHB is preserved in the presence of ethanol but not glutamate, according to research (Beccari et al., 2002).

Bioplastics can be derived from activated sludge through the introduction of basic organic acids, such as acetic, propionic, and lactic acids, into sequential batch reactors in a cyclical manner characterised by alternating periods of high and low nutrient availability (Donisi et al., 2001). The prompt describes the measurement of the biomass's swift response to an excessive substrate, quantified as chemical oxygen demand (COD). The results indicate that the biomass produced 649 mg of polyhydroxyalkanoates (PHA) per gramme of biomass (as COD), and 0.45 mg of PHA (as COD) per milligram of substrate and removing substances (as COD-1), respectively. Poly (-hydroxybutyrate/-hydroxy valerate) copolymer with 31 per cent hydroxy valerate monomer was generated from the substrate combination for the PHA composition (Donisi et al., 2004). Although this method shows promise in terms of scalability, yields might be enhanced further by enriching and generating biomass at greater organic loading than those explored in most research, making the process more economically feasible. The optimisation of low organic loading rates and the economic reduction of yields are the key technological hurdles in the manufacturing of bioplastics from sewage sludge. This will necessitate substantial research into increasing the organic load and efficiency of mixed medium vs mixed crops. Because of cheaper substrates and non-sterile reactors with limited process control, the overall cost of these processes will be greatly lowered. Furthermore, the intrinsic stability of the culture, along with the use of open fermentation, avoids the bottleneck associated with typical continuous bacterial fermentation. Bioplastics in sludge may be a new option with expanded research on applicability options and their potential beneficial effects and sequelae, if any.

2.4.2 Biosurfactants

Extracellular macromolecules are classed as metabolic byproducts in the microbial metabolism of organic substrates. Biodegradable substances possessing surfactant-like characteristics have been identified, exhibiting potential for diverse applications such as food, personal care products, and household/laundry detergents (Khaled et al., 1992; Kosaric et al., 1992). Gallery and Winter (2002) conducted a comprehensive examination of sewage sludge as a potential source for biosurfactant production. Their study focused on the utilisation of Bacillus licheniformis and Bacillus subtilis in various ways. The investigation into

biosurfactants commenced in batch reactors, employing various substrates such as molasses, waste vegetable oils, and olive oil. The media was exclusively supplemented with wastewater derived from the food industry as the carbon source. In the study conducted by Vipul Nandan and Ren (2000), saline was employed as the sole carbon source in the media.

Subsequently, activated sludge has been effectively employed to produce biosurfactants from diverse organic substrates under non-sterile conditions, utilising a wide array of carbon sources. The study conducted by Nitschke and Pastore (2003) demonstrates that biosurfactants formed from activated sludge use wastewater as a carbon source exhibit comparable performance to biosurfactants derived from a pure culture utilising costly pure organic substrates, when evaluated in sterile environments. The investigation focused on examining the biosurfactant-producing capabilities of filamentous actinomycetes found in foam-activated sludge treatment facilities, as indicated by recent Gordonia study findings. According to Pagilla et al. (2002), the use of hexadecane as the exclusive carbon source resulted in biosurfactants exhibiting a critical micelle concentration (CMC) that was three times higher compared to a combination of hexadecane and acetate, and this CMC value was three times higher than that observed when acetate was used as the sole carbon source. Conversely, the pathogenic nature of certain microbial strains has historically hindered the extensive manufacturing of biosurfactants. They offer a great deal of potential in the bioremediation of nonpolar substances, but more research into the fate and transit of diverse molecules is required. A closer look at microorganisms is therefore required in the synthesis of biosurfactants, as certain bacterial taxa may offer an increased risk to public health when used in this process. Biosurfactants can also be examined as novel sludge management alternatives, with non-pathogenic species such as Bacillus licheniformis, Bacillus subtilis, and others being used to create the surfactants.

2.4.3   Biopesticides and enzymes

In a study conducted by Jung et al. (2001), the researchers utilised a dynamometer to extract enzymes, including proteases, amylases, glucosidases, lipases, and dehydrogenases, for the degradation of excess sludge derived from municipal wastewater treatment plants. According to Jung et al. (2002), the protease activity shown a 69% increase compared to the other

enzymes that were examined. This finding implies that utilising protease could potentially enhance the process of in situ protein hydrolysis in wastewater treatment.

In a similar vein, the use of dehydrated and dried powdered sludge as a substitute feed for yeast extract (YE) in order to enhance the thermophilic toxicity (TOP) process for lipid molecule breakdown resulted in an efficiency of 82.9%, surpassing that of YE. The rate of deterioration shown an efficiency of 68.3 percent. (Matsumura & Nakano, 2002) Historically, the practise of utilising anaerobic digestion/electricity production and biosolids land application has been employed as a means of extracting energy and nutrients from sewage sludge. However, according to Bridle and Pritchard (2004), pyrolysis has been shown to offer advantages in terms of nutrient recovery and the generation of oil (energy) from pyrolysis carbon. An eight-week laboratory soil incubation study was conducted in Australia to investigate the availability of nutrients by utilising charcoal. According to Bridle and Pritchard (2004), while the nitrogen content in char is not soluble, the findings indicate that plants can still obtain phosphorus. The utilisation of various types of sewage sludge has been observed as a consequence of the research findings, namely for the purpose of cultivating rhizobium (nitrogen-fixing) inoculants (Ben Rabah et al., 2002d). Research findings have demonstrated that sludge-based biofertilisers exhibit superior effectiveness compared to existing biofertilisers when applied to crops such as soybeans and lucerne. Moreover, these sludge-based biofertilisers exhibit equivalent symbiosis efficacy in terms of nodulation and plant growth, as observed with sludge-based inoculants. The yeast broth utilised in this study was supplemented with mannitol. Following inoculation, there was a notable increase in the nodule index (modulation capacity) from a range of 4-6 to 8-12. Additionally, the quantity of rhizobia in the soil showed a significant rise, with uninoculated soil containing approximately 103 cells/gram, whereas inoculated soil exhibited a range of 106-107 cells/gram. According to Ben Rabah et al. (2002a), the salinity and heavy metal concentration of the sludge did not have any detrimental effects on soil fertility. According to Rebah et al. (2001), the application of acid-base treatment to sludge leads to the generation of bacterial growth substrates that are less complex. This, in turn, enhances the production of rhizobium and increases the effectiveness of sludge utilisation, particularly when higher solids levels are present.

According to a study conducted by Ben Rabah et al. (2002b), the use of supplements such as yeast extract and glycerol resulted in an increase in cell production. Moreover, as comparison to conventional media, sewage sludge offers enhanced safeguarding for the survival of

rhizobia during freeze-thaw cycles at a temperature of -20°C. Additionally, sludge serves as a valuable medium for maintaining a substantial population of nodules in soil. The storage conditions for microorganisms were investigated by Ben Rabah et al. in 2002c. Consequently, sludge can be utilised as an innovative substrate and formulation aid (in its dry state) for the production of biofertilizers that enhance soil fertility and promote plant growth.

In contrast to artificial fertilisers, which can be customised to optimise their nutritional properties for specific applications, treated sludge is applied at an agronomic rate, which corresponds to the amount of nitrogen present in the soil where the crop is grown, in order to meet the crop's nutrient needs. Various nutritional requirements can lead to an overabundance or insufficiency of said nutrients. In addition, it should be noted that crops can readily access nitrogen in organic forms such as nitrate and ammonium. However, it is important to highlight that inorganic forms of nitrogen necessitate first conversion into organic forms by microbial activity. Nitrogen leaching may arise as a consequence of inadequate consideration of organic nitrogen mineralisation, leading to the occurrence of over-fertilisation. In order to overcome the obstacles that may hinder the process of commercialising nutritious sources, it is necessary to allocate significant and continuous efforts. There is a need for extensive research to investigate the long-term potential of toxic chemicals and heavy metals leaching from soils that contain biofertilisers.

2.4.4   Biosorbants

Sewage sludge contains a high concentration of organic matter and is predominantly carbonaceous in composition. As a result, it can transform into activated carbon by controlled pyrolysis or chemical treatment. This process has the potential to significantly reduce sludge volume while also generating usable adsorbents at a lower cost than commercially activated carbons (Jeyaseelan & Lu, 1996; Martin et al., 1996). Historically, impregnation with chemically activated H2SO4 followed by pyrolysis has been employed to make adsorbent materials from sewage sludge. After oven-drying to a constant weight at 105°C, the mother sludge was chemically activated by impregnation with H2SO4. After pyrolysing the activated sludge with inert nitrogen, it was washed with a 10% mass dilute HCl solution. These sorbent particles are ground to the desired particle size and surface area, resulting in a particle with increased porosity and surface area (Otero et al., 2003). The samples burned at 950°C had the greatest capacity for removing H2S. (Bagreev and Bandosz 2002). Although sludge-based adsorbents have a lower specific surface area than commercial activated carbons, they appear

to perform equally in experiments (Hagstorm et al., 1997). Sludge from sewage treatment plants can be utilised to create pyrolysis and chemically activated adsorbents for the removal of dyes, organic pollutants, and a range of other adsorbates. Additional research is needed to determine the synergistic/antagonistic effects of contaminants and the molecular mechanisms by which they adsorb. Otero et al., 2001 Additionally, adsorbents can be synthesised by combining them with organic polymers to form stable products known as composite adsorbents.

According to one study, methylene blue adsorbs faster than safranin and adsorbs better in a binary solution (Rozada et al., 2003). In analogous settings, other studies have discovered that dyes are eliminated more rapidly than organics. Adsorption of crystal violet was reported to be stronger and more rapid than that of indigo carmine or phenol (Otero et al., 2003). Annadurai et al. carried out a second survey. In 2003, the adsorption capability of Rhodamine 6G, a synthetic dye adsorbent produced from microwave-treated activated sludge, was examined. The energy cost analysis in this work demonstrates the viability of adopting the recommended microwave technology to fabricate sludge-based adsorbents.

Although both have limited applications, sewage sludge has been used to cure malodorous gases via adsorption and flue gases via desulphurisation (Krogmann et al., 1997). Numerous studies have demonstrated that digested sludge is a good adsorbent for removing colours from water, with a specific surface area of 82 to 150 m2/g. 2001; Weng et al. 2003; Weng et al. 2004. (Kargi and Ozmihci, 2004) Investigated the viability of stripping powdered activated sludge (PAS) for the adsorption removal of six distinct colours using shake-flask tests, similar to those used with powdered activated carbons. Except for one dye (Direct Yellow 12), all adsorption curves followed the Freundlich isotherm. PAS was shown to have a high capacity for the adsorption of large molecular weight compounds but a low capacity for the removal of tiny molecules such as phenol (Martin et al., 2004).

Numerous bio sorbents suffer from organic and metal leaching problems. However, by combining sodium alginate and calcium alginate immobilised sludges for the same purpose, this issue can be somewhat alleviated (Chen et al., 2002a). On the other hand, all of these metal removal studies ignore the impacts of competitive inhibition and intricate antagonistic interactions, and the bulk of them rely on laboratory batch testing. While activated carbon derived from sewage sludge may be beneficial for eliminating colours from water, it is still unknown which type of sorbent can manage all wastes/pollutants (e.g., exhaust gas flows).

Even though numerous feasibility studies and analyses have been conducted on the production of sorbents from sludge, this value-added model produces inconsistent results when it comes to organics and other pollutants, particularly when it comes to waste stream disposal; the majority of the results come from batch studies, and its practical application is limited. As a result, additional research is necessary to better understand each pollutant's synergistic/antagonistic interaction with the adsorbent, as well as its environmental impact (secondary disposal/reuse possibilities). While the composite adsorbent has excellent adsorption capability, the cost factor is significant. Cost savings are possible with biological adsorbents.

2.5 INORGANIC SEDIMENTS REUSE AND POSSIBLE COMMERCIALIZATION

Suppose the sediment in the pond (probably from a stormwater retention pond) is relatively inorganic. In that case, it could be dredged, dewatered, and then reutilised as building materials such as bricks and cement.

Meanwhile, SSA (incinerated sewage sludge ash) has several effects on the finished brick product: ashes act as opening agents (Valls & Vazquez, 2001). In spite of laborious efforts to improve the leachability of the product, there is a restriction placed on the product's applicability over a prolonged period of time.

Even though the produced bricks outperformed regular bricks in terms of strength and mechanical qualities, they suffered from a variety of secondary issues, such as the development of moss, the icing on the surface, and whitening. Additionally, the produced bricks were a lighter colour than regular bricks (Okuno et al., 1997b; Wiebusch & Seyfried, 1997). Flame melting is presently the preferred method for creating ceramic material as a consequence of several improvements made to the existing process of producing ceramic material.

Furthermore, it has been documented that sludge has been utilised in the manufacturing processes of slag, pumice, and even Portland cement (Onaka, 1999). The utilisation of sludge aggregate, composed entirely of sludge, as a substitute for granite, does not compromise the compressive strength of the material. Additionally, compounds resembling sludge-derived cement can be employed as alternatives to conventional cement.

Up to twenty percent of the overall mass might be composed of Portland cement (Tay et al., 2002; Paya et al., 2002). After 150 days of use in concrete, mixtures, including sludge, marine clay, and pelletised aggregates, demonstrated insignificant levels of leachate, indicating that there was no significant pollution of the surrounding environment (Tay et al., 2004).

O'Conor et al. (2001) suggest that there exists a potential utilisation of sludge that has been supplemented with glass aggregate in various industrial applications, such as the manufacturing of abrasives, asphalt paving, roofing shingle granules, and floor tiles. Nevertheless, these bricks exhibited certain limitations, including the need for anticum agents, resulting in elevated production costs, and the heightened demand for tempering water, potentially leading to greater burning expenses. Both of these elements have the potential to increase the cost. As a result of this, additional research must be undertaken to ascertain the feasibility and actual efficacy of products based on SSA in practical applications.

The utilisation of mechanical grinding of the SSA has the potential to aid in surmounting these constraints. Nevertheless, it is vital to examine the expenses involved with mechanical grinding. The persistent accumulation of metals and toxins over time remains a subject of concern, necessitating a considerable level of effort to promote the use of construction aggregates. This level of effort can be likened to that necessary for the promotion of composts derived from sewage sludge. The level of exertion necessary to promote composts derived from sewage sludge would be comparable. Furthermore, the energy requirements associated with the manufacturing of aggregates might be influenced by the building procedures and technologies employed. Consequently, it is imperative to perform an analysis that encompasses energy, materials, products, and benefits to ascertain the pricing of the end product and the availability of conventional materials in certain geographic areas. Such considerations may impose limitations on overall utilisation.

2.6   Other uses

Some urban sewage treatment plants and thermal power plants use dehydrated sediment to replace part of the coal to obtain better efficiency. During the incineration process, sediments can be safely burned to ash, which also reduces the cost of heat generation.

3 METHODOLOGY

The methodology used in this study is discussed in this section. The literature review analyzed journal articles, regulations, and guideline papers in Canada, the United States of America, Singapore, China, and Europe. As can be seen from the flow chart of *Figure 3-1*, the key tasks of the methodology, including instruments, site protocols and laboratory testing are discussed in the following sub-sections.

Figure 3-1: *Sub-Section of Methodology*
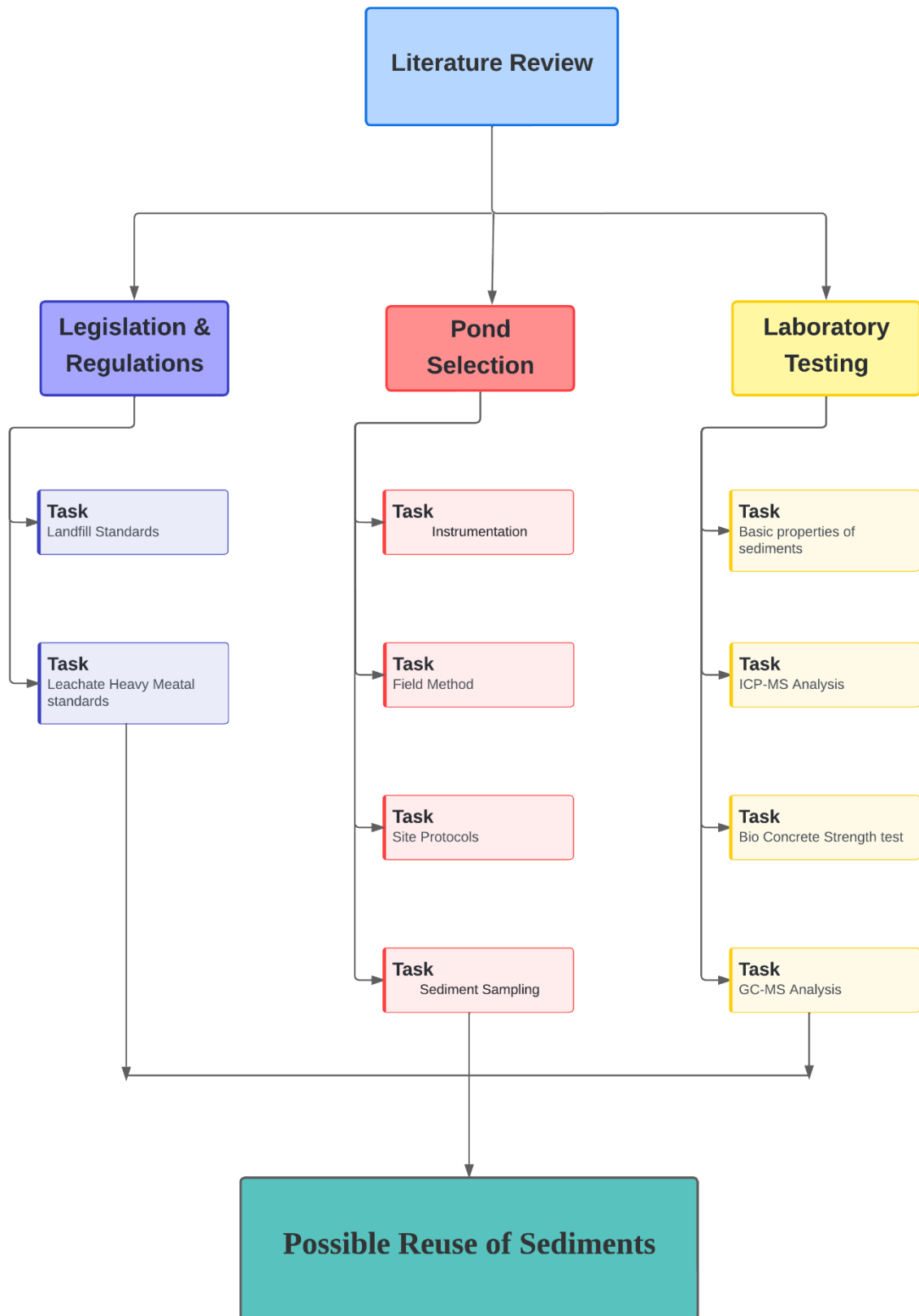
3.1 Instrumentation

Samples may be collected immediately using the En Core® Sampler or a syringe if it is predicted that the total VOC level in the sediment would be less than 200 g/kg. When using syringes, the sample must be promptly put in the sample container, which is a pre-prepared vial containing 40 millilitres, to reduce losses due to volatilisation. The unpreserved samples should be placed in vials that are 40 millilitres in size and contain either 10 millilitres of organic-free water or 10 millilitres of organic-free water together with a preservative.

While using an En Core® Sampler, the sample has to be stored in a plastic bag that is hermetically sealed, secured, and protected.

Using a syringe, transfer about 3.7 ccs, or approximately 5 grammes, of the sample material into the containers that hold 40 ml and have been prepared in advance. Put each container inside its own plastic bag. Instead of applying a custodial seal to the container, apply it to the plastic bag so that it may be easily transported. While utilising syringes, it is imperative that there is no possibility of air being trapped behind the sample prior to its evacuation, since this would cause irreparable harm to the specimen.

Using laboratory spatulas made of stainless steel, transfer between 4.5 and 5.5 grammes of sample material to containers with a capacity of 40 millilitres. Put each container inside its own individual plastic bag. Instead of applying a custodial seal to the container, apply it to the plastic bag so that it may be easily transported. (Georgia, 2020)

3.2 Site Protocols and Pond Selection

1. Occupational Health and Safety (OSH) criteria were met during every visit to the location designated by the NGA.

2. The field crew made sure to check in with a site engineer (or with workers from the security business on the weekends) and then followed the route to the research site as specified. As the earthwork and building operations advanced during the course of the research, the route to take and the location of a good site to park changed.

3. Before departing the location, staff members made sure to check out with the NGA or the security contact.

Two ponds from the residential area and two ponds from the industrial area were selected. Then, one sediment sample from the outlet and another sample from the inlet was taken from each pond.

Here are the locations of the four ponds:

1) Link Drive Stormwater Pond - 13 Link Drive, Wairau Valley, 0627, Auckland, New Zealand

2) Stormwater Pond – Silverdale, 0932, Auckland, New Zealand

3) Unsworth Heights Stormwater Quality Pond - Unsworth Reserve, Unsworth Heights, Auckland 0632

4) Millwater Sportsground Stormwater Pond - Silverdale 0932 Auckland, New Zealand

3.3 Field methods

It is recommended that sediments from ponds be collected at a depth of at least 30 centimetres below the water's surface whenever it is practicable. The person taking the sample should wade into the pond until they reach the necessary depth, at which point they should either use a pan made of stainless steel or scoop sediments straight into sampling jars.

It is necessary to repeat the t-collection process to collect a sufficient amount of silt in order to fill the sample jars.

It is essential that you include a note in the "notes" section of the field data collection form on the required quantity of grabs at each sample site.

On a mixing tray, use the spoon made of stainless steel to fully blend the sediment sample into a single, uniform sample until you have reached the desired consistency.

Using the mixing spoon, transfer some representative subsamples of the homogenous mixture to the clean sample jars. To the fullest extent feasible, fill the container with the sample combination.

Be sure that the name of the project, the sampling site number, the date of the sampling, the kind of sample (sediment), and that it is properly tied to the jar all appear on the label that is attached to the container. Wrap the label with water-resistant transparent plastic tape to make it more durable in the rain.

3.4   Analytical Methodology

Extraction: Solids - Volatile Organic Compounds

Solids are converted into volatile organic compounds by the extraction process.

use as our foundation the US-EPA procedures 8260 and 624. A representative subsample of the samples was obtained by coring a large number of holes in the samples. To extract the samples from the methanol, sonication was used. The extraction process and subsequent analysis were delayed while the methanol was stored in a container at a temperature of 4 degrees Celsius. Together with each methanol extraction worksheet, there were twenty blank worksheets included. In the last moments before doing the analysis, an aliquot of the methanol extract was mixed with an aliquot of water. The final extraction and analysis were carried out with the assistance of an automated Purge and Trap system.

Extraction: Liquids – VOCs

Use as our foundation the US-EPA procedures 8260 and 624. An automated Purge and Trap System was used for the extraction process. This system utilised a stream of helium for the purpose of removing volatile compounds from the aqueous phase and placing them into a VOCARB4000 trap. A pre-extraction addition of 10 micrograms per litre of equivalent surrogates was added to the liquid.

Analytical Instrumentation – VOCs

In accordance with the procedure number 3545 of the USEPA. The samples were homogenised with sodium sulphate, and rapid solvent extraction was performed using dichloromethane extraction (ASE). The homogenised material was then supplemented with 333 and 667 g/kg (solid) of surrogates prior to the extraction process.

Solids - SVOCs: Extraction

According to USEPA procedure 3545. The samples were homogenized with sodium sulphate and extracted with dichloromethane using accelerated solvent extraction (ASE). Before extraction, 333 and 667 g/kg (solid) of surrogates was added to the homogenized sample.

Extraction: Liquids - SVOCs

In accordance with the procedure 3520A of the USEPA. After that, samples were acidified, and a process known as continuous liquid/liquid extraction was used to remove

dichloromethane from the mixture. A surrogate was added to the sample at a concentration of 12.5 and 25 g/L (the equivalent in liquid volume) before the extraction process.

Extract Clean-up - SVOCs

The technique of gel permeation chromatography was used to clean up a part of each extract (USEPA method 3640).

SVOCs - Instrumental Analysis

Using a comprehensive GC-MS scan, SVOC compounds were located and identified (based on USEPA method 8270). A four-point calibration curve and internal standards were used to determine the concentration of each of the target compounds.

Tentatively Identified Compounds(TIC) Analysis

Throughout the process of performing the standard volatile organic compound (VOC) and semivolatile organic compound (SVOC) analytical procedures used to determine the levels of the organic compounds that are discussed in this article, a significant amount of additional information is produced.

The mass spectrometer generates a series of spectral lines for each molecule that are distinct from one another. When analysing a material as complex as sewage sludge, a large number of spectral lines for chemicals other than those in the standard suite are expected to be present. A computer analyses the collected data from the various mass spectral library recordings over time to determine what they are. The library records include information on compounds that are only seldom included in analytical suites such as the one that was utilised for this work. This means that mass spectra were recorded for a significant number of chemicals for which there was no standard included. Because of this, their identity has not been conclusively established; these substances are only presumed to be recognised (TICs).

The mass spectra obtained from this survey were evaluated, and TICs were found in the following list of chemical classes and individual compounds: There is no guarantee that the compounds have been correctly identified, and it is quite probable that the amounts that have been cited should be viewed as just being semi-quantitative. Existing information may be

useful for comparison purposes in the future, when sludge samples are analysed, and when standard analytical suites may be increased. Both of these events may take place in the future.

3.5   Particle size analysis

The Environmental Lab at the University of Auckland will use a Galai WCIS-100 particle size analyser in order to get an accurate measurement of the particle size. This is a "time-of-flight" apparatus, which measures the distance that a particle travels via a laser beam in order to establish its dimensions and form (Weiner et al., 1998). In each sample, millions of particles are counted and quantified, and data on the frequency of occurrence of particles in various size bands are recorded. The number of particles, their total surface area, and their total volume is used to report the frequency.

3.6   Dissolved metals -ICP-MS

In order to examine the dissolved aluminium in each filtered sample, a sub-sample of 14 millilitres was collected from the filtrate. Once the sub-sample was placed in the acid-washed or sterilised plastic vial of 15 millilitres capacity, 1 millilitre of nitric acid was added to it to make it more acidic. Evaluation of the Efficiency of a Sediment Retention Pond That Receives Chemical Treatment 18 These sub-samples are going to be sent to the Environmental Lab at the University of Auckland so that dissolved aluminium may be determined using inductively coupled plasma mass spectrometry (ICP-MS) in accordance with the APHA 3125B technique.

3.7   Bio-Concrete moulding preparation and strength test

In this experiment, Bio-Concrete specimens were created by combining different proportions of sediments and binders, as shown in Table 2. In Table 2, the specimen was denoted by the initials of the binder's name and the sediments, which were then followed by their respective mixing ratios (in grammes per gramme). As an example, the notation S15-L5-G0-PC80 indicates that 15% of sediments were combined with 80% by weight of Portland cement and 5% by weight of lime. It was determined via a battery of mechanical and chemical testing which binder would work best, as well as the ideal proportions of binder to other ingredients. The specimens were partitioned into the following four categories:

The first group, which included samples 1-8, included PC, lime, and gypsum as binders. The quantity of binder that was applied was altered, but the sediment stayed the same throughout

the experiment. The goal was to determine which kind of binders were most effective and which types would work best for the project at hand.

The binders for the remaining three groups (samples 5-16) were prepared using PC and gypsum. The ratio of sediment remained the same, however, the ratio of gypsum varied across all of the subgroups (sample 5-8, sample, 9-12, and sample 13-16). Measurements were taken for unconfined compression strength, moisture on the tent, the volume change of sewage sludge, leaching of heavy metals (Cu, Zn, and Pb), chemical oxygen demand (COD), and pH values as part of the mechanical and chemical experiments.

The mould preparing procedure follows NZS3112: Part 2: 1986:

This procedure describes the moulding and curing of concrete cylinders for compressive and tensile strength testing. The processes involve hand compaction, vibration compaction, and curing in the laboratory.

It must be possible to securely attach the base plates and, if necessary, the top plates to the moulds without distorting the plates. Assembled, the moulds must be correct circular cylinders that are suited for producing concrete specimens that meet the specifications of Before using the moulds, lightly treat them with a releasing agent and verify that they are substantially waterproof, as determined by their capacity to retain water poured into them.

(c) Tamping rod. A circular, straight rod of steel measuring around 16 mm in diameter and 600 mm in length, with one end tapering to a hemispherical point.

(d)Compaction by hand. Placing the moulds on a sturdy, level platform and pouring the concrete in three stages of about equal volume into the moulds. Move the scoop around the top border of the mould while putting each scoop of concrete to achieve a balanced distribution of the concrete inside the mould. Each layer should be tamped. Rod the depth of the first layer. The strokes of the following layers should penetrate, but not through, the layer immediately underneath them. Tap the sidewalls of the mould to fill any spaces left by the rod after compacting each layer. Ensure that the last layer's compaction leaves the mould overfilled.

(e)Consolidated horizontally Using a sliding and rotating action, work the top plates into the concrete until they fit tightly against the top of the moulds. Then, attach the top plates tightly to the moulds without distorting the plates. Place the moulds immediately on their sides with

their axes horizontal and retain this orientation until the specimens are removed from the moulds.

(f) Curing. Laboratory-produced specimens Immediately after casting the specimens put the moulds in the fog room maintained at 21±2 °C and do not disturb them. (NZS3112, 1986)

Figure 3-2: Mould Preparing Procedure

The testing procedure follows NZS3112: Part 2: 1986:

Figure 3-3:Testing Procedure

4   RESULTS AND DISCUSSIONS

After a series of lab tests, including basic properties of sediments, GC-MS (Gas chromatography–mass spectrometry) analysis, ICP-MS(Inductively coupled plasma mass spectrometry) analysis and Bio-Concrete strength testing, there could be various reuse options for the sediments from the retention pond.

4.1 Basic Properties of Sediments from Retention Pond

Table 4.1: Basic properties of Sediments from Retention Pond

Retention pond sediment	
Wet Unit weight (Kn/m^3)	12
Moisture content (%)	162
Undrained shear strength (kPa)	3
Organic matter (%)	31
Hydraulic conductivity (cm/s)	4.0×10^{-7}
Cu (mg=L)	1.15
Zn (mg=L)	7
Pb (mg=L)	1.5
COD (mg=L)	1282
pH	8.1

Table 4.1 provides a summary of the fundamental features of sewage sludge. The moisture content of the sludge, as indicated in the table, was 162%, which was within the normal norms for sewage sludge (represent an average of all ponds studied).

Normal dewatering procedure in the United States (NRC 1996) stipulates that the moisture content cannot exceed 400 per cent (NRC 1996). Therefore, the moisture content of the sewage sludge utilized in this study fulfilled the criterion for dewatering. The optimal range for sludge handling and landfill building is between 85% water content, the optimal water

content for conventional proctor compaction, and 95% water content, the sticky limit of the sludge material. (O'Kelly, 2005)
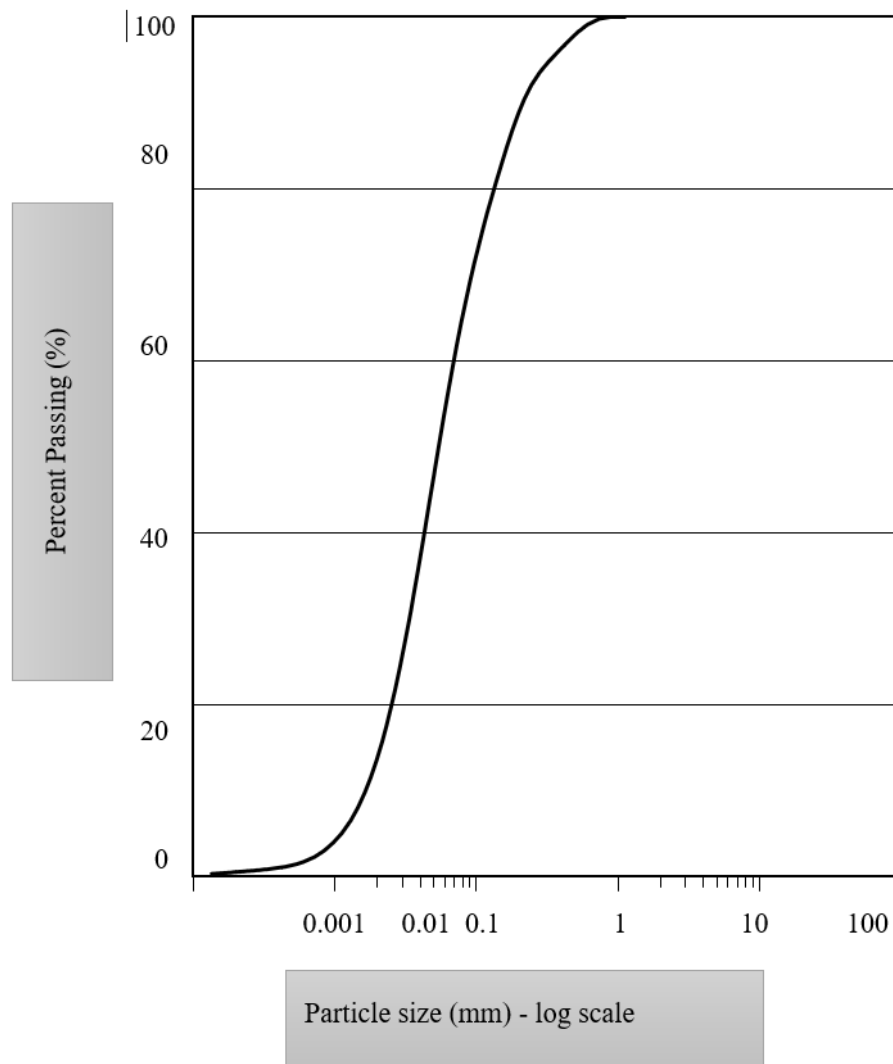
Figure 4-1 Grain size distribution curve of sediments from a retention pond (represent an average of all ponds studied)

According to **Figure 7**, over 70% of particles were smaller than 0.1 mm, which corresponded to the average range for sediments from the retention pond (Campbell et al. 1978).

The detected leachable contaminants in the sediments from the retention pond are within New Zealand requirements Waste Acceptance Criteria for Class A Landfills (Dolan, 2003), as shown in Table……. lists the fundamental characteristics and leachability of heavy metals. All heavy metals were below the hazardous material leaching standards (Cu 100 µg/L, Pb 5

µg/L, Cd 1 µg/L, and Cr 5 µg/L) (MEPC 2007). As a result, sediments from the retention ponds were not classed as hazardous substances, but rather as ordinary solid waste that is approved for beneficial reuse. The sediments from the retention pond consisted of pebbles and sand (88% by weight), glasses (9.1%), porcelain (2%), and a minor quantity of unburned organics (0.9%). Quartz, calcite, hydroxyapatite, gehlenite, and sanidine comprise most of the mineral makeup. As a result, sediments the from retention pond poses no threat to the surrounding environment and may not be further stabilized or utilized as a building material.

This study utilized Portland Cement (PC) 32.5 (unconfined compressive strength of 32.5 MPa), lime, and gypsum as binder materials.

4.2 ICP-MS analysis-heavy metal leaching

The following procedure was used in the preparation of the samples:

After placing the wet sample, which weighed about 0.5 grammes, into a Teflon tube with 80 millilitres of volume, chemicals were added. When the containers were sealed and put in a Maxi-44 rotor, they were digested in an Ethos-Up Microwave reaction system (Milestone SRL, Italy) for twenty minutes at a temperature of 180 degrees Celsius. After adding 44 mL of water of type 1, the digest was brought to a final concentration, and then the weight was measured. Before doing the analysis, we further diluted the solutions 20 times in 2% HN. In order to cut down on the amount of polyatomic interference, the solutions were subjected to a quantitative analysis for the necessary elements using an Agilent 7700 ICP-MS operating in He mode. In a matrix-matched solution, calibration standards were generated using single-element standards at a concentration of 1000 ppm (Inorganic Ventures, USA). The drift and matrix effects were monitored using a solution of Tb that contained 20 ppb. All of the data have been converted back to the WET sample's initial concentration using the back-calculation method.

The following are some of the reagents that were used in the analysis:

3.5 millilitres of 37% hydrochloric acid, Tracepur, manufactured by Merck

1.5 millilitres of 69% HNO_3-Tracepur, made by Merck

1 millilitre of 50% H_2O_2 - Sigma Aldrich

As can be seen from the Original Results of ICP-MS from Appendix 7.1:
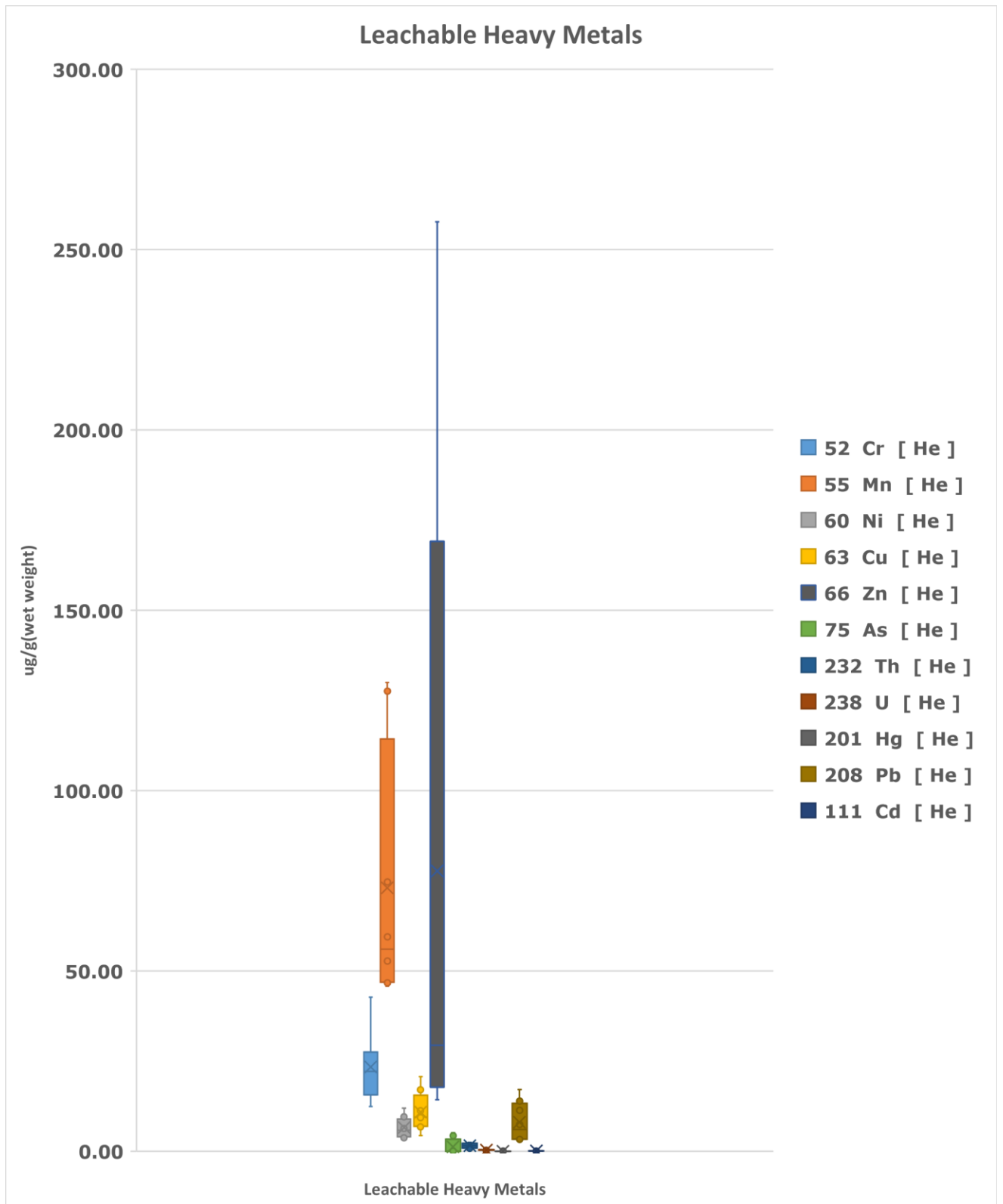
Figure 4-2: Leachable Heavy Metals

Table 4.2: *Soil limits and biosolids classification by contaminant levels (NZWWA, 2003)*

	Soil limit or ceiling concentrations (ug/g dry weight)	Grade a max. concentration (ug/g dry weight)	Grade b max. concentration (ug/g dry weight)
Arsenic	20.00	20.00	30.00
Cadmium	1.00	1.00	10.00
Chromium	600.00	600.00	1500.00
Copper	100.00	100.00	1250.00
Lead	300.00	300.00	300.00
Mercury	1.00	1.00	7.50
Nickel	60.00	60.00	135.00
Zinc	300.00	300.00	1500.00

Table 4.3: *Class 3 Waste Acceptance Criteria for Inorganic and Organic Elements(MfE, 2004)*

Contaminant of concern Unit Maximum allowable total concentration	Maximum allowable total concentration
Arsenic	17.00
Cadmium	0.80
Chromium	290.00
Copper	44.00
Lead	60.00
Mercury	0.70
Nickel	310.00
Zinc	400.00

Table 4.4: Heavy metals in samples and NZ standards for grade 'a' biosolids standards

HEAVY METALS IN SAMPLES VS STANDARDS

Legend:
- Maximum allowable total concentration
- Grade a max. concentration (mg/kg dry weight)
- Meadian(dry weight)
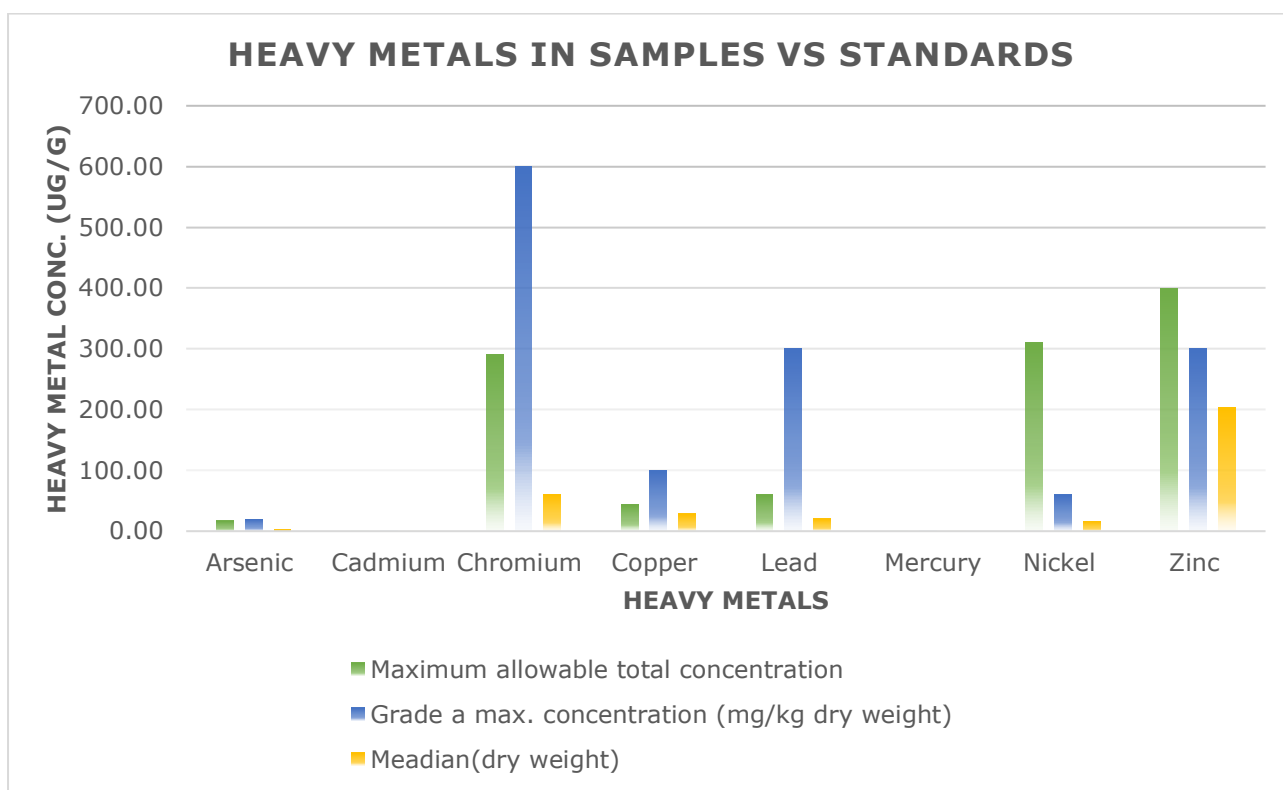
The median heavy metal contents for the 8 samples are shown in **Table 4.5** with the maximum values provided by the NZ standards for grade 'a' biosolids(**Table 4.6**)(blue bars)(NWWA, 2003), which may be safely handled by the public and applied to land, along with the New Zealand screening standards for landfills(**Table 4.7**)(green bars)(MfE, 2004) and the absence of danger of major adverse impacts. As can be seen from this data, zinc is the most problematic heavy metal, followed by copper and lead.

Soil reference values (SRV) have been established for dredged materials by the Minnesota Pollution Control Agency to help with this very purpose. Sediments with SRV values of Level 1 or less may be reused in residential areas, while those with SRV values of Level 2 or less can only be reused on properties classified as suitable for industrial use (Stollenwerk et al. 2009). Copper and lead SRV levels at Level 1 were comparable to those found in New Zealand biosolids (100 and 300 ug/g), whereas the SRV value for zinc was much higher (8700 ug/g). Copper, lead, and zinc maximum concentrations in Level 2 SRV samples were 9000, 700, and 75,000 ug/g, respectively. For this reason, when determining whether

sediment is suitable for reuse in New Zealand, the 100 ug/g restriction set by New Zealand for copper and the 300 ug/g limit set by New Zealand for zinc and lead are more suitable.

Even though some sediment alternatives entail land application, it is still important to take into account NZ sediment quality criteria since there may be concerns about the erosion and transfer of reused sediments to aquatic ecosystems (i.e., incorporation into sediments).

The "low" and "high" interim sediment quality guidelines (ISQG) levels for zinc, copper, and lead are 200 and 410 ug/g, 65 and 270 ug/g, and 50 and 220 ug/g, respectively, according to the ANZECC (2000). However, it must be emphasized that repeated sediments inputs into aquatic habitats are likely to be greatly diluted when included in sediments. Soil and/or biosolid application recommendations are thus seen as being more relevant in the context of properly recycling sediments in New Zealand.

ZINC

The concentration of particulate zinc varied from 37.49 to 675.17 ug/g, with a median of 203.71 ug/g and an interquartile range of 46.64-433.13 ug/g. Using the 2012 maximum zinc concentration for New Zealand biosolids of 300 ug/g as a guideline, a 50/50 mix of sediment with a clean material (e.g., compost or green trash before composting) would lower sediment zinc concentrations to acceptable levels.

COPPER

The concentration of copper varied from 11.31 to 54.04 ug/g, with a median of 28.31 ug/g and an interquartile range of 18.14-40.73 ug/g. Based on these concentrations, copper in the sediment samples did not exceed the recommended value for 'a' grade New Zealand biosolids (which may be applied to land without the danger of substantial deleterious effects) without dilution.

LEAD

The median and interquartile range of particulate lead values in sediment were 20.9 ug/g and 8.76-34.73 ug/g, respectively. As with copper, the maximum concentration of lead (44.84 ug/g) is substantially below the maximum value for grade 'a' biosolids (300 ug/g) and so should not restrict reuse applications of sediment.

4.2.1 Amendment

There is the potential for adverse effects if reusing untreated or undiluted sediments (i.e., fill/compost, etc.), whether the evaluation is made from the standpoint of particulate contaminants (e.g., by comparing with biosolid guidelines) or mobilization of metals in leachates. This is the case whether the evaluation is made from the standpoint of particulate contaminants (e.g., by comparing with biosolid guidelines). It is preferable to have a "treatment train" that either stabilizes or dilutes the contaminants in sediments sufficiently to enable safe and responsible reuse of the material. Even though this risk could most likely be adequately managed by using best management practices to control the conditions of use, it is more desirable to have a "treatment train." The reduction of the potential environmental concerns posed by heavy metal contamination in sediments may be accomplished via the use of the following suggested approaches (Depree, 2008):

1. Physical entrapment: This method includes embedding the sediments into a solid matrix, such as concrete or asphalt, to enclose and secure the metal pollutants.

2. Dilution: entails mixing the sediments with material that has not been contaminated with the particulate matter to lower the concentrations of particle contaminants to a certain level (that is, below a specified regulatory guideline value).

3. Stabilisation: entails adding chemicals/materials to reduce the mobility of metals in sediments (through pH regulation, chemical modification, and/or sorption). In contrast to 'dilution,' the objective is to lower the concentration of leachate pollutants, even if the particle concentrations may be comparable to unstabilized sediments.

This section's primary objective was to investigate the efficacy of various stabilization methods in lowering the mobile concentrations of zinc and copper. Lead concentrations were also analyzed, although there was less of an emphasis placed on lead stabilization since it did not surpass the ANZECC 95% protection trigger value even when a cautious dilution factor of 10 was used. The final regulatory criteria that are applied to sediment reuse will determine the relative relevance of stabilization and dilution, even though there is considerable overlap between the two processes. For instance, if the standards specify maximum particle concentrations of pollutants, then the need for dilution will increase. If, on the other hand, water quality standards are implemented, then stabilization procedures may handle pollutant solubilization regardless of the actual particle contamination amounts that are present. Coal

fly ash, which acts as an alkalinity modifier, compost, which acts as humic metal sorption, and phosphate are all examples of stabilization treatments (formation of insoluble metal phosphates).

4.2.2   PHOSPHATE AMENDMENT

Cotter-Howells and Caporn (1996) and Kumpiene et al. (2008) found that phosphate amendment may effectively stabilize heavy metals such as lead and zinc in soil. As can be seen from *Figure 4.4*, there is a strong linear relationship between leachable heavy metals

and phosphate dosage. Increased phosphate dosage lowers the concentration of leachable heavy metals There is a 56%, 37% and 31% reduction in concentrations of leachable Zn

(from 321 µg/L to 142 µg/L), Cu (from 27 µg/L to 17 µg/L), and Pb(from 16 µg/L to 11 µg/L) respectively, with increased phosphate loading ranging from 0.05%-0.25%.

PHOSPHATE AMENDMENT(Residential Area)

y = -848x + 351.6
R² = 0.9753

y = -52x + 29.6
R² = 0.9826

y = -24x + 17.6
R² = 0.9
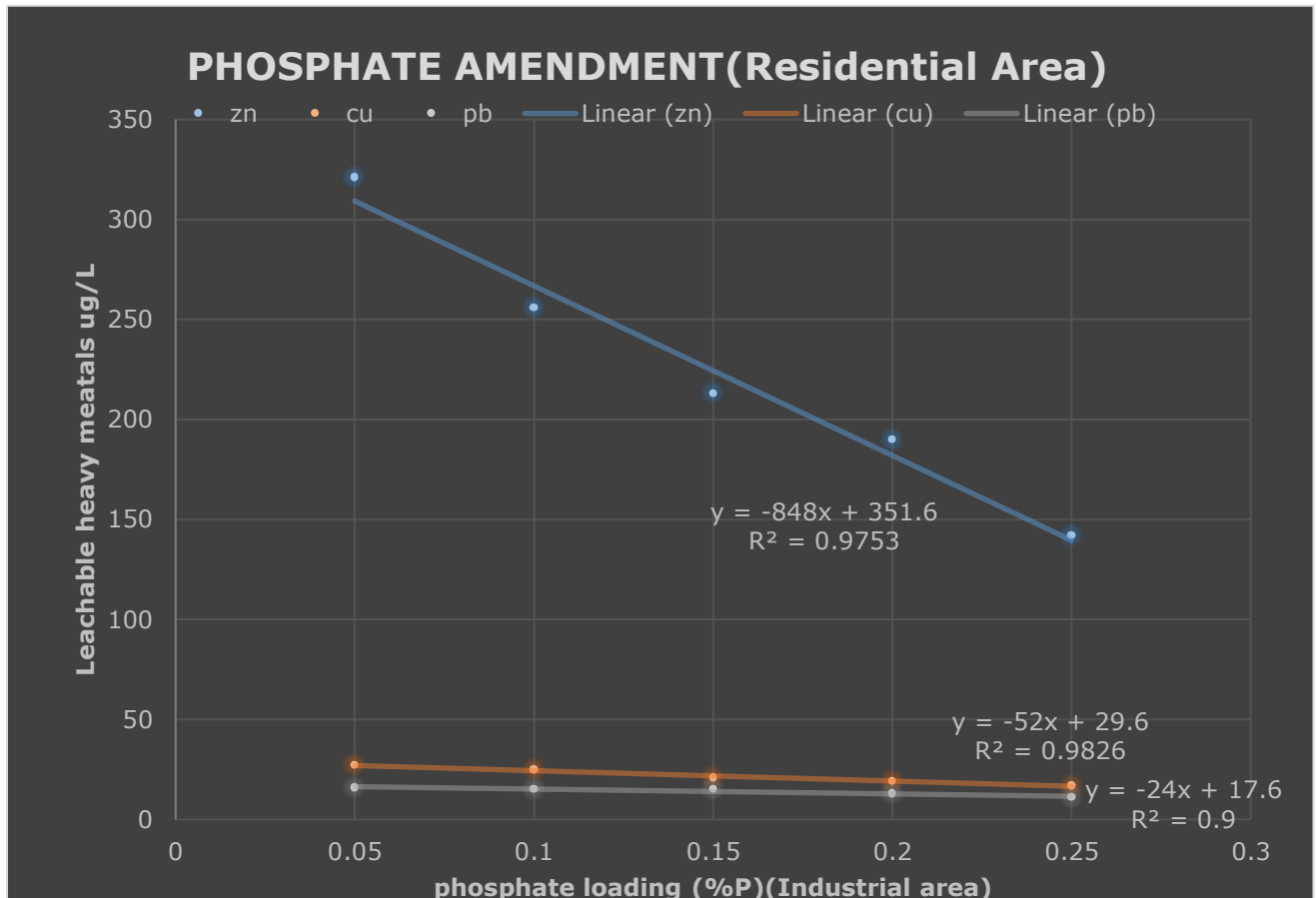
Figure 4-3: Phosphate Amendment (Residential Area)

In the sediment samples from industrial areas, however, the Copper, Zinc, and even Lead leachate concentrations rose linearly with phosphate input. Phosphate's inability to immobilize Copper, Zinc, or Lead was unexpected in the samples from industrial areas. The concentration of leachable Zinc dramatically increased by 403% when phosphate loading

changed from 0.05% to 0.25%. The rise in metal concentrations was probably related to phosphate-induced DOC production.

PHOSPHATE AMENDMENT(Industrial Area)

Legend: zn, cu, pb, Linear (zn), Linear (cu), Linear (pb)

Y-axis: Leachable heavy meatals ug/L
X-axis: phosphate loading (%P)(Industrial area)

$y = 5948x + 244$
$R^2 = 0.9166$

$y = 604x + 9.2$
$R^2 = 0.9739$

$y = 370x + 12.9$
$R^2 = 0.9173$
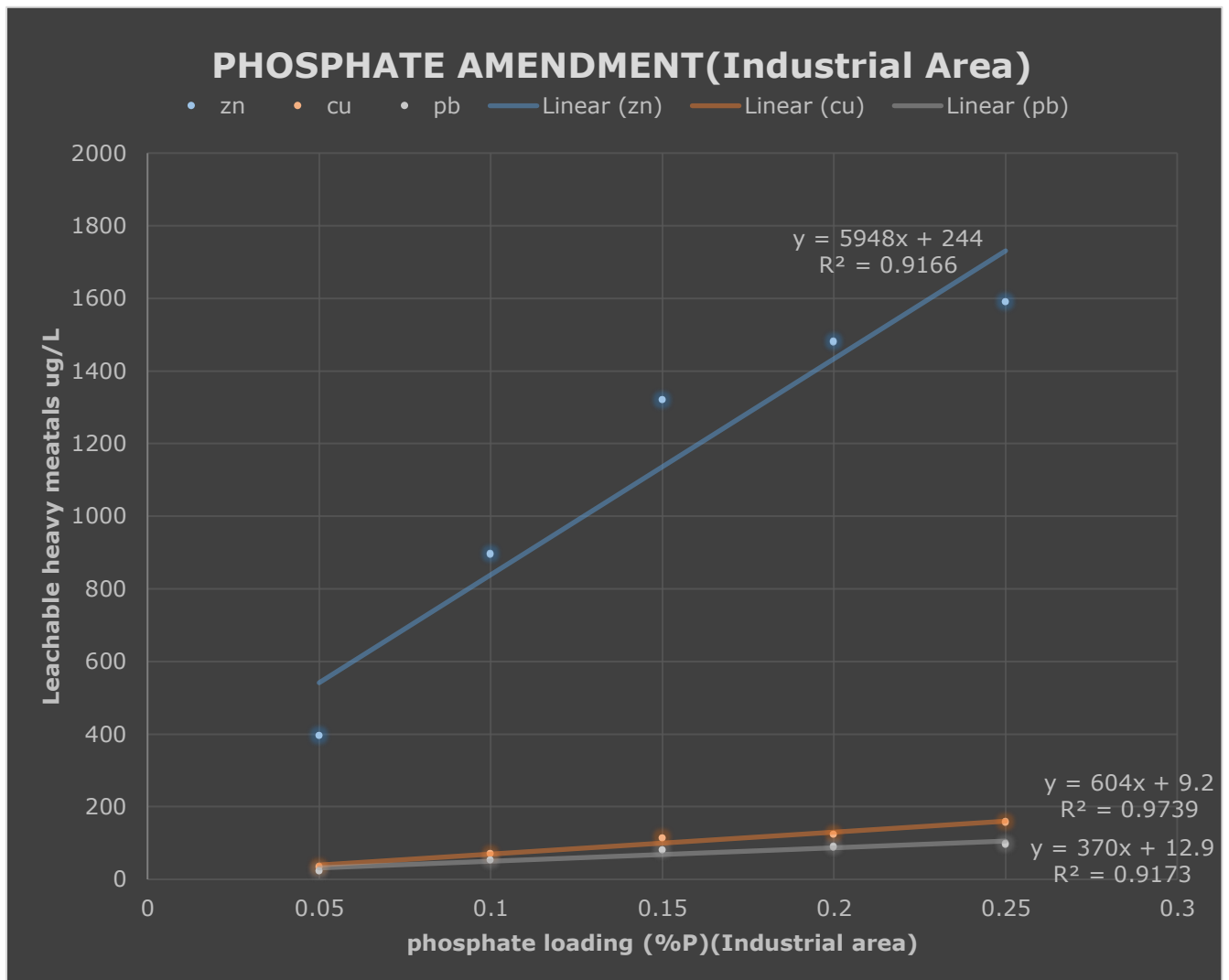
Figure 4-4: Phosphate Amendment (Industrial Area)
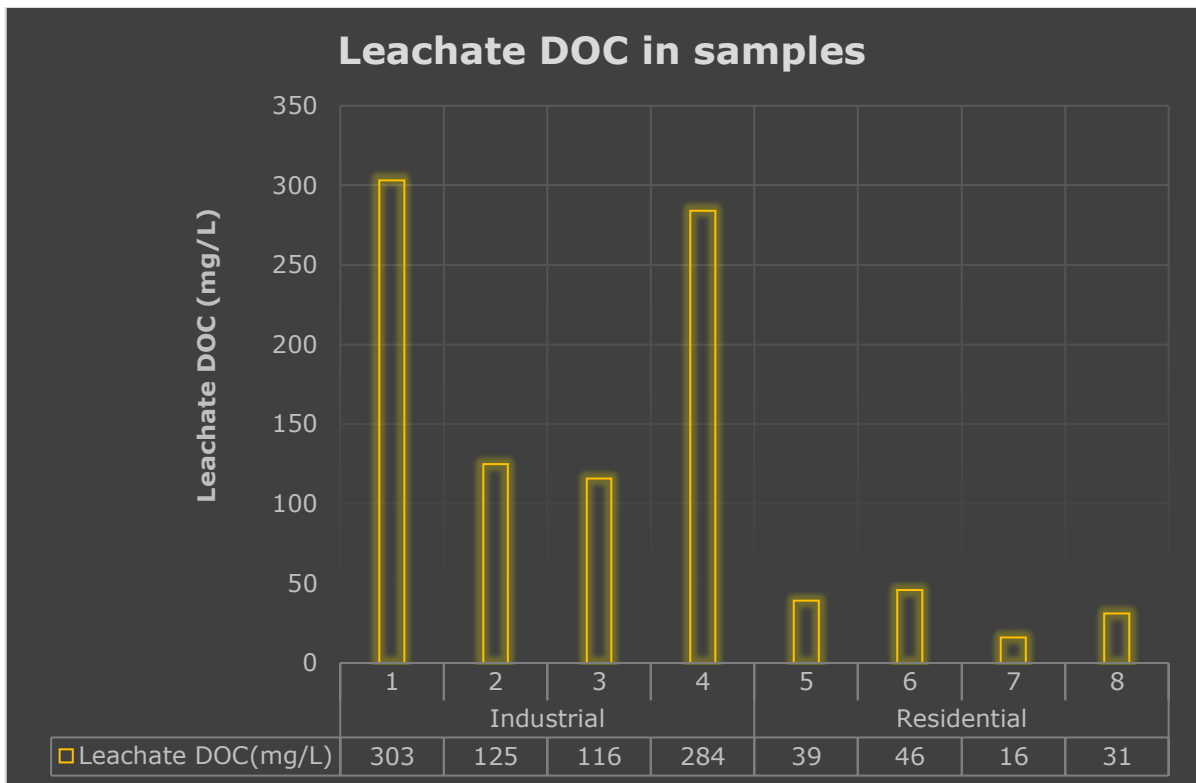
Figure 4-5*: Leachate DOC(µg/L)*

As can be seen from **Figure 4-5**, the leachate DOC in samples from the industrial areas is higher than that of samples in residential areas. The highest concentration of DOC in samples from industrial areas is 303 µg/L, while that from reside dential area is 46 µg/L, which explains why phosphate amendment does not work in samples from industrial areas.

Overall, the findings indicate that the effective use of phosphate amendment in the stabilization of heavy metals is dependent on the amount and quality (fresh or humified) of organic matter (OM) that is present in sediments as well as the heavy metals that need to be stabilized. Because the discharge of large quantities of DOC will accelerate the mobilization of heavy metals, phosphate amendment is generally not going to be effective in sediments that have significant concentrations of fresh organic matter (OM). Metals may, depending on their affinity for DOC, be stabilized by a phosphate amendment; however, this is only possible when the amounts of leached DOC are very low, as was the case with samples 5-8.

4.2.3   COMPOST AMENDMENT
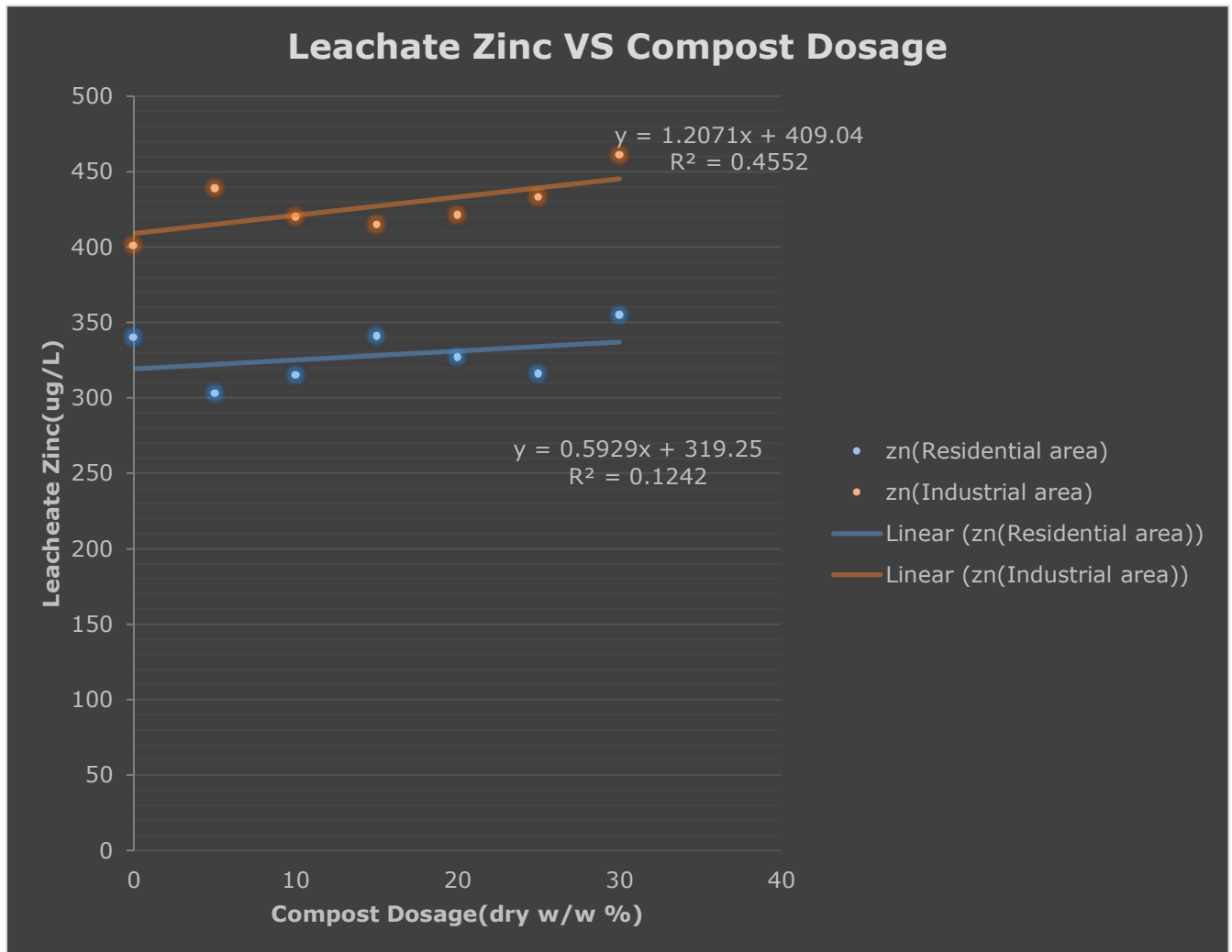
Figure 4-6 *Leachate Zinc VS Compost Dosage*

As can be seen from **Figure 4-6**, the leached zinc content was not affected by the addition of up to 30% compost. In the samples from the industrial area, the zinc content was 401 µg/L in the control sample (which included no compost), 420 µg/L in the 10% amended sample, and 461 µg/L in the 30% amended sample. Similarly, the relative leachate zinc concentrations for samples from the residential area were 340, 315, and 355 µg/L when 0, 10, or 30% compost additions were respectively used.

There is no linear relationship between leachate zinc and compost dosage, neither in samples from residential nor industrial areas. Leachate zinc slightly fluctuated with compost dosage within 15% of its original value when compost dosage ranges from 0%‑30%.

Therefore, compost is not an effective amendment option for stabilizing zinc in sediments. This is consistent with zinc being a moderately mobile metal that is readily outcompeted for adsorption sites by other cations (such as lead or copper) (Cao et al. 2004).
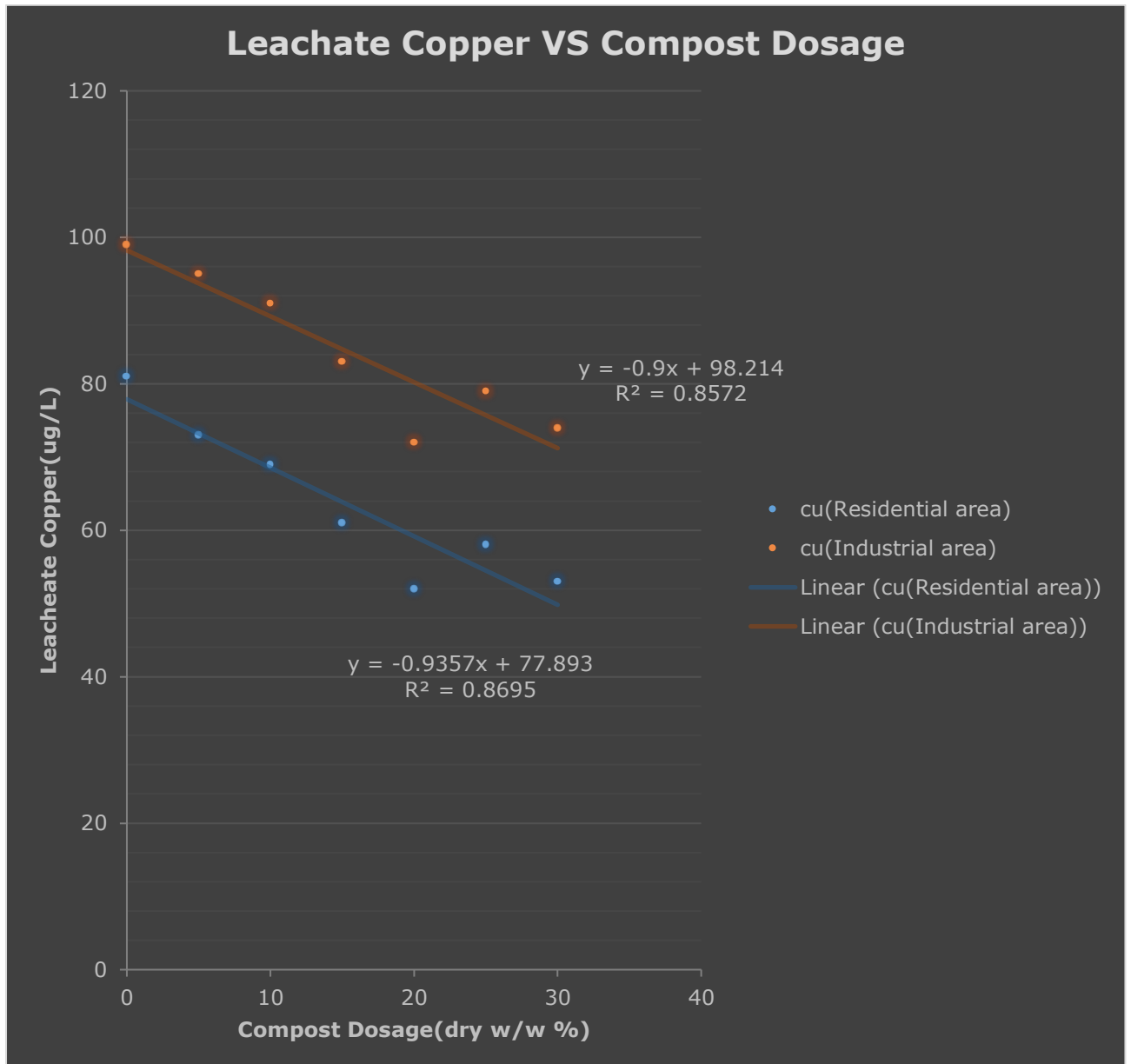
Figure 4-7 Leachate Copper VS Compost Dosage

When it comes to the stabilization of copper in the sediment samples, there is a negative linear relationship between leachate copper and compost dosage. The addition of compost led to modest reductions in the leachate copper concentrations, and the ideal loading rate was 20% (dry weight). The concentration of copper in the leachate was reduced by 36% in

samples from the residential areas when 20% compost was added, going from 81 µg/L in the control (0% compost) to 50 µg/L with 20% compost added. The results for C11 were fairly comparable, with the leachate copper level dropping by 27% from 99 µg/L in the control to 72 µg/L when 20% compost was applied. Increasing the amount of compost to 30% did not result in an improvement in the immobilization of copper.
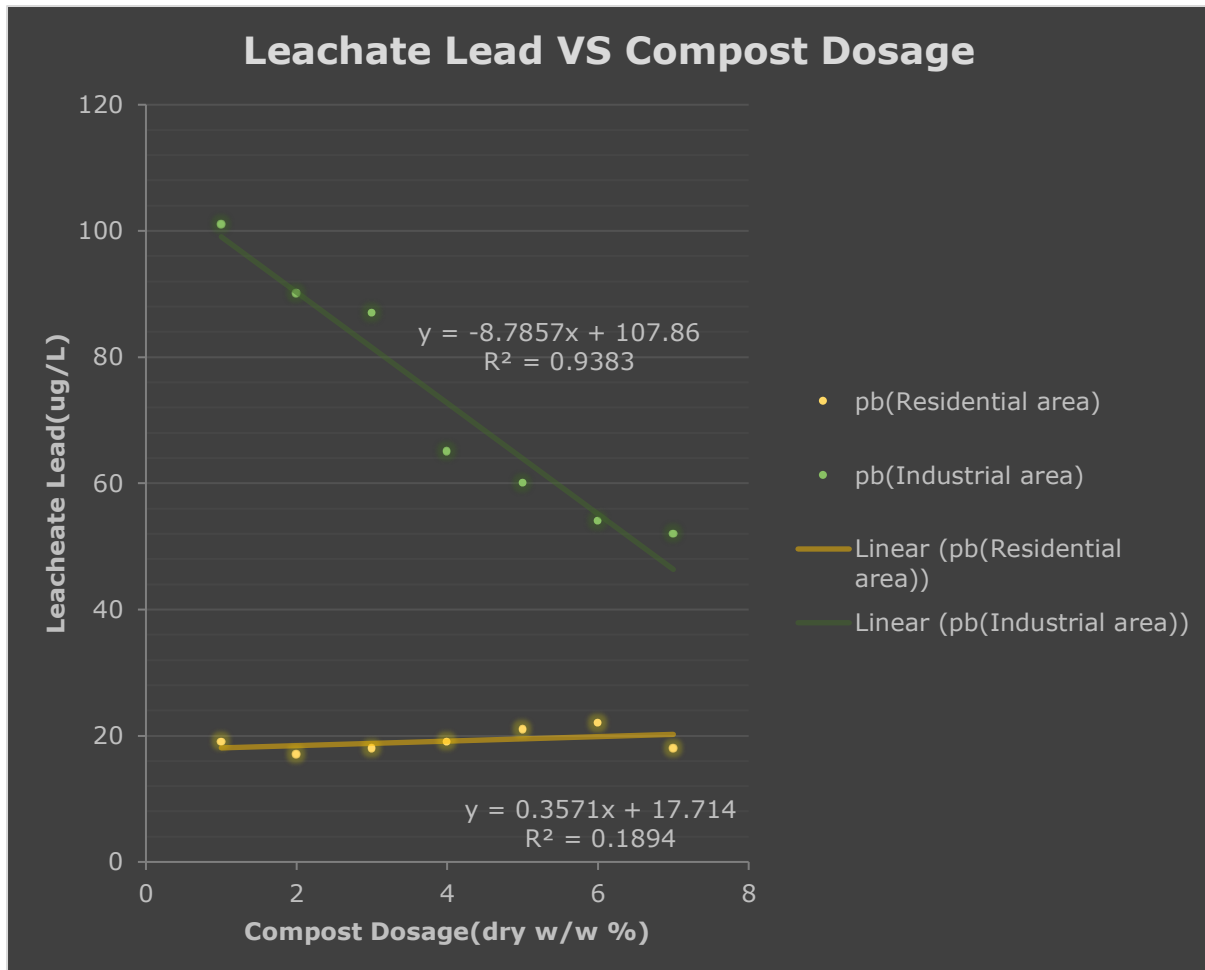
Figure 4-8 Leachate Lead VS Compost Dosage

Compost amendment was successful in lowering the amount of lead that leached from sediment samples from industrial areas, similar to what was found for copper leaching. There is a strong negative linear relationship between leachate lead and compost dosage. Raising the compost loading to 30% brought the leachate lead concentration down by 41%, from 101 µg/L down to 52 µg/L. However, the leachate concentration was quite low in the non-amended sediment (19 µg/even though the addition of compost did not have any influence on the lead leachate concentrations of sediment samples from the residential areas.

In conclusion, when 20 per cent of compost was added to sediment, the quantities of copper and lead in the leachate were lowered by 36%-41%. compost was ineffective at stabilizing zinc; however, the combination of compost addition and pH control may be successful.

4.2.4   Amendment by pH control

When it comes to the mobilization of copper, lead, and zinc from soils, the two most critical elements are dissolved organic carbon (DOC) and pH. (Jordan et al. 1997; Sauve et al. 2000; Impellitteri et al. 2002; Linde et al. 2007).

Leacheate Copper VS pH

- cu(Residential area) • cu(Industrial area)
— Poly. (cu(Residential area)) — Poly. (cu(Industrial area))

$y = 23.464x^2 - 368.16x + 1552.7$
$R^2 = 0.9614$

$y = 19.554x^2 - 304.06x + 1279.8$
$R^2 = 0.9409$
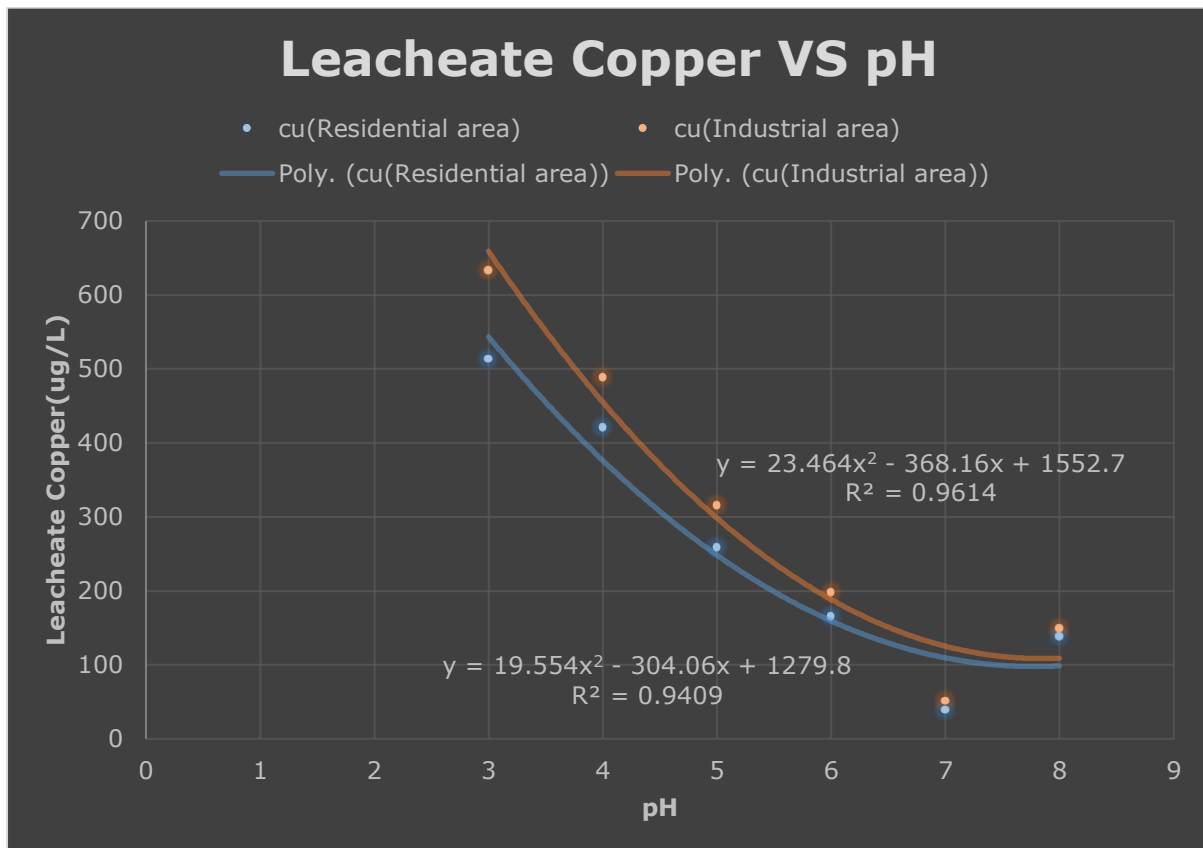
Figure 4-9 Effect of leachate pH on leachate Copper from sediment samples

As can be seen from **Figure 4-9**, there is a very strong polynomial relationship between leachate copper and pH. Even while copper and lead are more resistant to shifts in pH than zinc, even very minor pH shifts between 6 and 8 may cause the concentrations of copper and lead to shifting dramatically. (**Figure 4-9 & Figure 4.10**). It is confident to estimate that when leachate pH is 7.84, the leachate copper will be at the lowest point (around 40 µg/L).

Leacheate Lead VS pH

- pb(Residential area)
- pb(Industrial area)
- Poly. (pb(Residential area))
- Poly. (pb(Industrial area))

$y = 13.589x^2 - 176.4x + 599.14$
$R^2 = 0.9762$

$y = 5.8929x^2 - 79.25x + 297.93$
$R^2 = 0.9213$
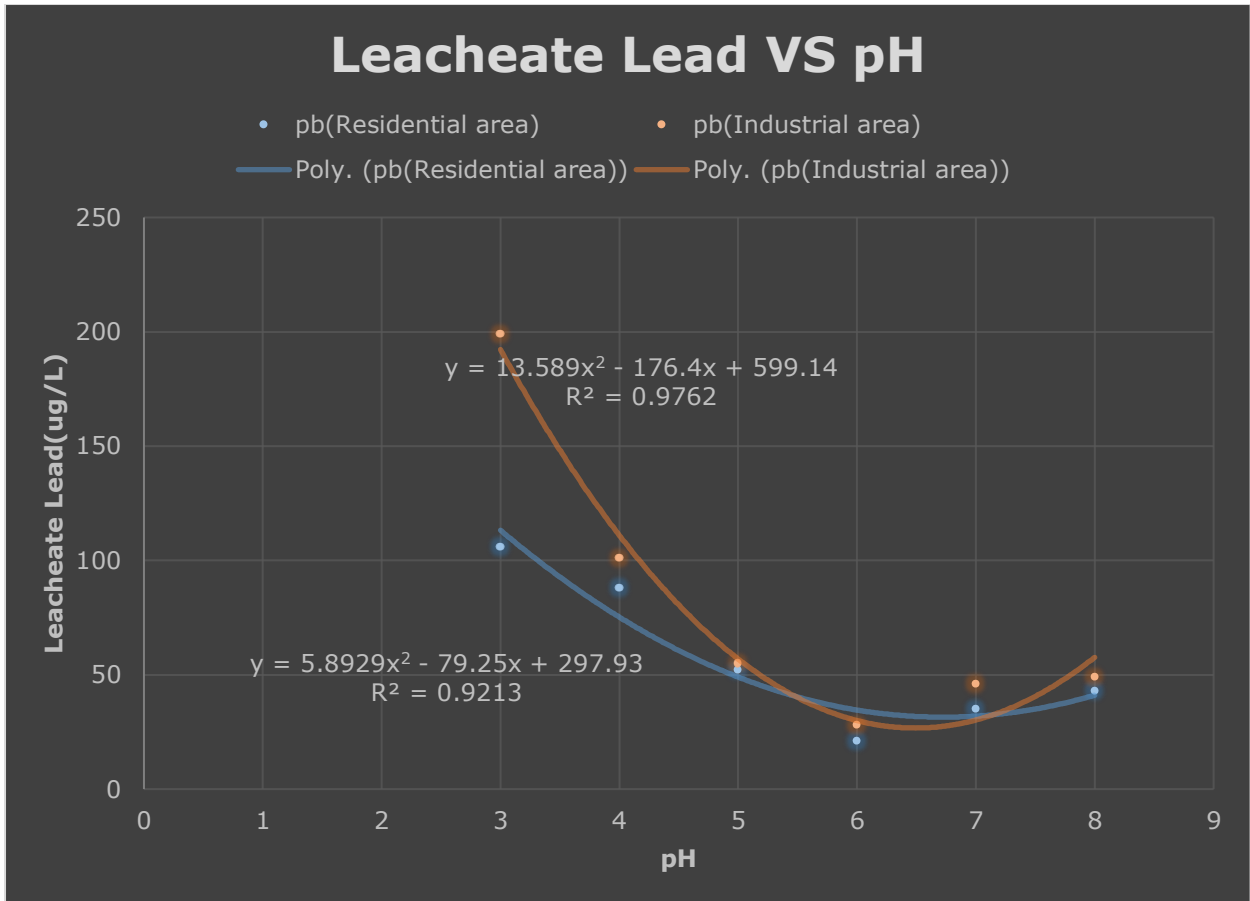
Figure 4-10 Effect of leachate pH on leachate Lead from sediment samples

Like copper, there is a very strong polynomial relationship between leachate lead and pH. raising the pH to 6.49 brought the leachate lead concentration down by 92%, from 633 µg/L down to 51 µg/L. When determining the effectiveness of certain materials in terms of their ability to be stabilized, it is important to ensure that the pH is kept consistent. When the pH of the leachate was less than 5, there was a significant increase in the solubilization of heavy metals from the sediment.
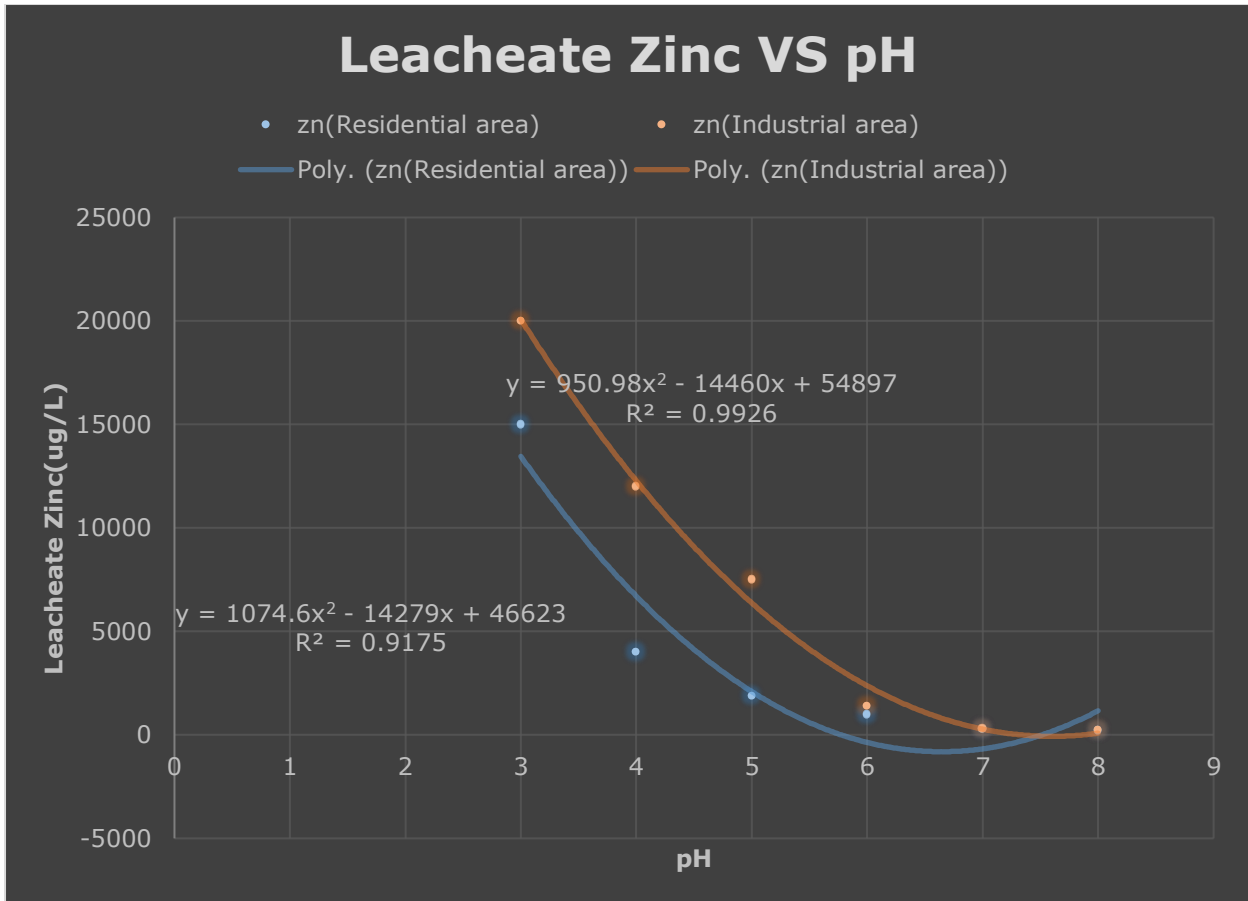
Figure 4-11 Effect of leachate pH on leachate Zinc from sediment samples

The leachate pH had a more significant impact on the solubilization of zinc from sediment samples (**Figure 4.12**). For instance, the concentration of leachate zinc produced by sediment samples from the industrial areas was as follows: 20000, 12000, 7500, 1400, 320 and 235 µg/L when the pH was 3, 4, 5, 6, 7 and 8, respectively. What is more, it is important to keep the pH of leachates between 7-8. Nevertheless, it is very improbable that leachates would have a pH lower than 5. Changing the pH of the leachate from about 6 to 7 brought a reduction in the zinc content in the leachate by a factor of 4.375, from 1400 to 320 µg/L. The reduction of leachate zinc in sediment samples from residential areas was noticeable but not dramatic (from 985 to 288 µg/L), which is most likely due to the conflicting impact of high DOC concentrations.

Zinc is arguably the most problematic heavy metal limiting the reuse of sediment (concerning both particulate and leachate concentrations), and as a result, any stabilization strategy should be effective against zinc. Zinc is a relatively abundant and mobile metal (Murakami et al.

2009), and because of this, any stabilization strategy should be effective against zinc. Phosphate and compost have both been shown in prior research to be beneficial in lowering zinc leaching (Depree, 2008), however, both amendments were found to be ineffective in the current investigation. The regulation of the pH of the leachate proved to be the most successful strategy for lowering the zinc solubility of sediment.

In conclusion, a pH range of 7.5 to 8 is desirable due to the quick increase in solubility that occurs at pH values lower than 7, the optimal pH range for zinc was somewhere between 7 and 8. In relation to the other metals, it demonstrated a copper leachate minimum at pH 7.84, with copper concentrations rising at pH 8, while the lead leachate minimum matched a pH of 6.49. Although lead concentration is not a key concern, copper concentration is. As a result, a pH of around 7.2 seems to be the optimal number for managing the concentrations of leachate copper and zinc.

4.2.5   Stabilisation Using Alkaline Modifier

Limestone ($CaCO_3$) is the most often utilised alkaline material for improving soil alkalinity, In the absence of fly ash, then, limestone, calcium oxide, or even wood ash (about 70% of $CaCO_3$) may be used as an alkaline modifier.

Effect of Limestone (alkaline modifier) on leachate Zinc

$y = 2.4814x^2 - 63.559x + 598.93$
$R^2 = 0.978$

$y = 1.6279x^2 - 35.592x + 332.96$
$R^2 = 0.9101$

Legend:
- zn(Industrial area)
- zn(Residential area)
- Poly. (zn(Industrial area))
- Poly. (zn(Residential area))

Y-axis: Leacheate Zinc (ug/L)
X-axis: Limestone (% w/w)
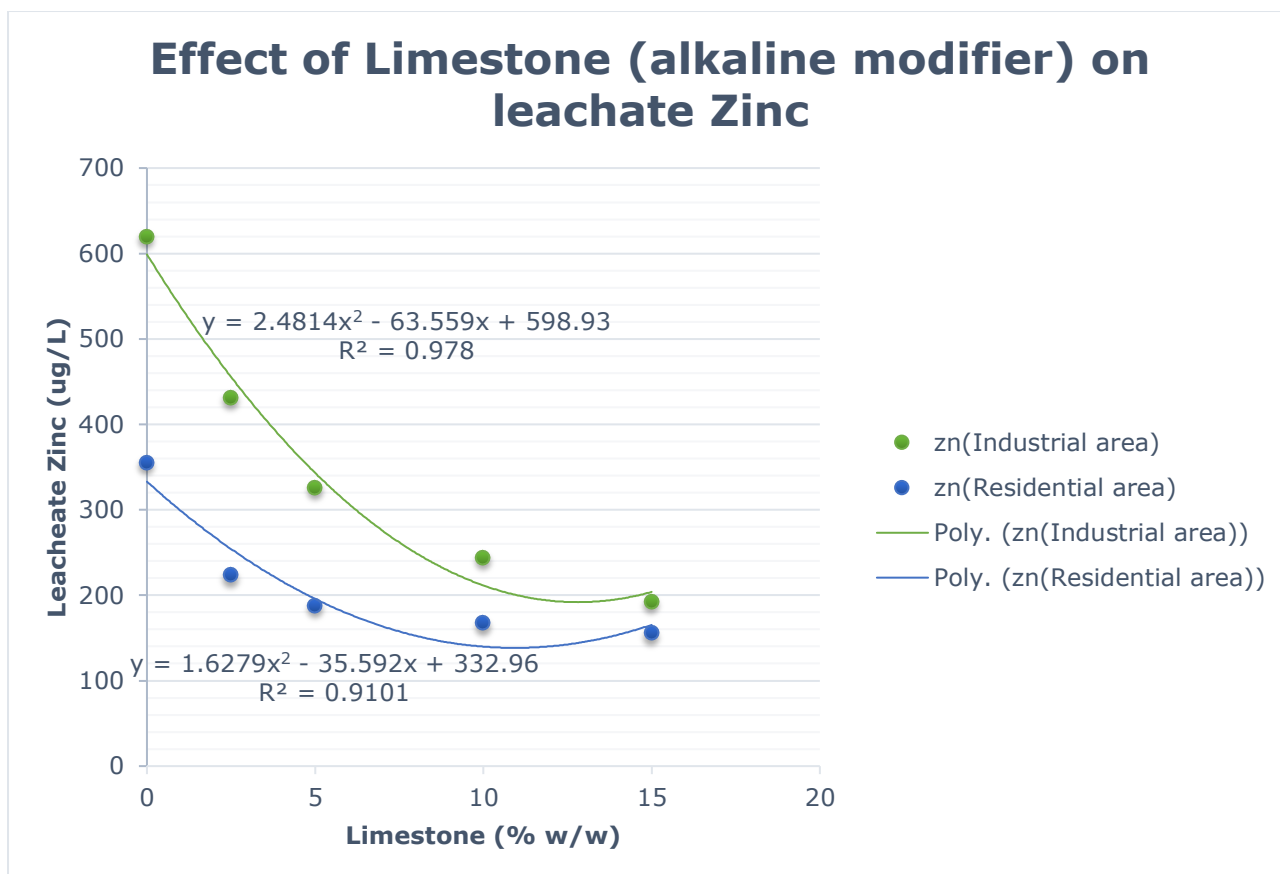
Figure 4-12 Effect of Limestone (alkaline modifier) on leachate Zinc

Figure 4-12 depicts the impact on heavy metal leachate concentrations of adding 2.5-15% limestone to sediments. The leachate zinc declined with increasing limestone loadings up to a maximum of 15%, and limestone amendments reduced zinc leachate by 56 and 69% from 355 to 155 ug/L and 620 to 192 ug/L, in sediments from residential and industrial areas, respectively. Presumably, the increased effectiveness is attributable to the much lower DOC concentrations, which promote metal solubilization otherwise. Practical applications to stabilise zinc in sediments by altering alkalinity must be quite durable. It has been reported that acidic soils modified with around 5% peat and 5% limestone maintained a neutral pH for at least two years (the period of the research), during which the average decrease in leached copper and lead was 98-99%. (Kumpiene et al. 2007). Therefore, there is tremendous promise for the effective long-term stabilisation of zinc in sediments based on ph modification.
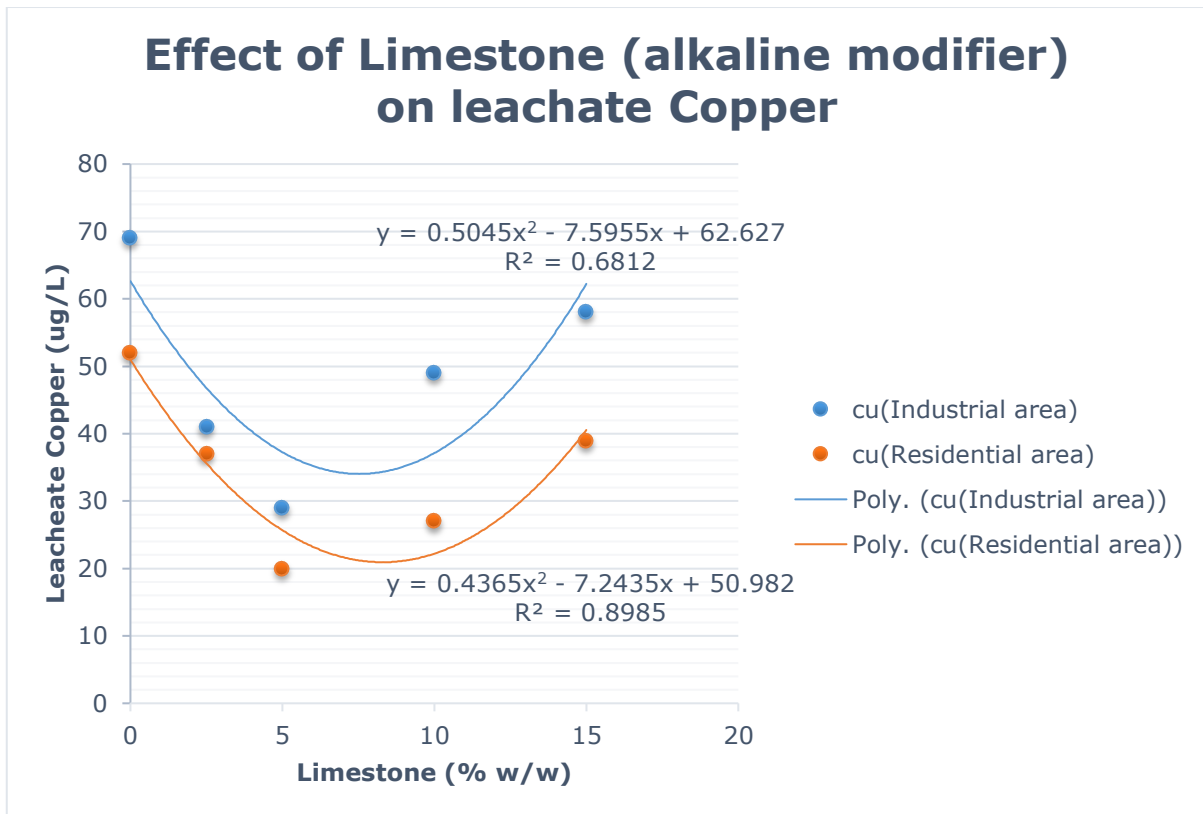
Figure 4-13 *Effect of Limestone (alkaline modifier) on leachate Copper*
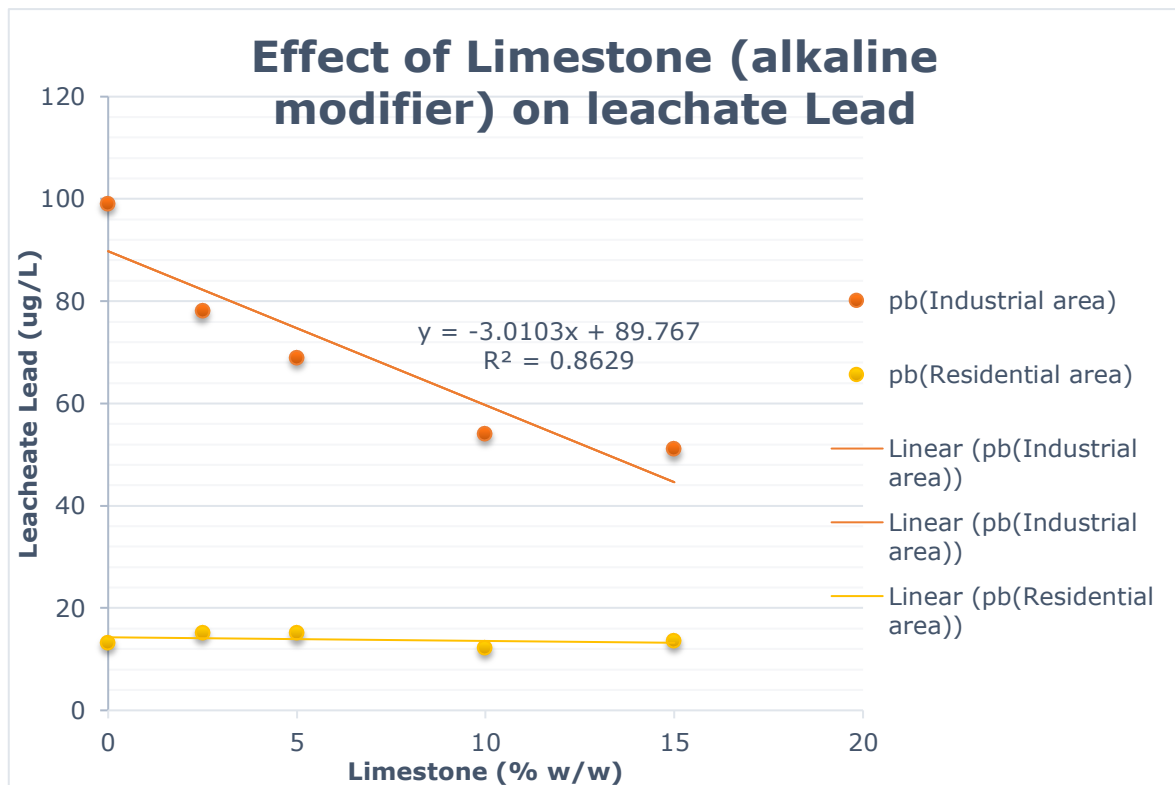
Figure 4-14 *Effect of Limestone (alkaline modifier) on leachate Lead*

Figure 4-13 & Figure 4-14 summarises the effect of limestone in stabilising copper and lead in sediments compared to sediments having no limestone. Similar to the pH amendment, when 7.6% w/w of alkaline modifier was added into the sediment, the leachate copper will drop dramatically by 55%. It is also effective in reducing the concentration of leachate lead by adding 15% of alkaline modifier, resulting in a drop from 99 to 51 ug/L.

In conclusion, similar to pH amendment, alkaline modifier could reduce the concentration of leachate heavy metals effectively. The leachate zinc declined with increasing limestone loadings up to a maximum of 15%, and limestone amendments reduced zinc leachate by 56 and 69% from 355 to 155 ug/L and 620 to 192 ug/L, in sediments from residential and industrial areas, respectively. Furthermore, when adding 7.6% w/w of alkaline modifier into the sediment, the concentration of leachate copper will drop by 55%. It is also effective in reducing the concentration of leachate lead by adding 15% of alkaline modifier, resulting in a drop from 99 to 51 ug/L.

To conclude, after several stabilisation trials in Section 4.2.2, 4.2.3 and 4.2.4, several stabilisation attempts using phosphate, compost, and alkaline modifiers were detailed. According to the results of these studies, leachate concentrations were reduced by roughly 60-70% for lead and copper and 55% for zinc. Although zinc removal efficiency was greater in sediment from the residential area, this is likely due to lower leachate DOC concentrations (the average is about 33 mg/L). Comparatively, the average concentration of DOC in the samples from the industrial area was 207 mg/L. Higher DOC concentrations promote the solubilisation of heavy metals, hence diminishing the efficacy of stabilisation procedures.

4.3   GC-MS analysis - Tentatively Identified Compounds (TIC) Analysis

There are several TICs were identified in the sediment samples:

Saturated hydrocarbons (those with just one link between carbon atoms, such as octane) are called alkanes, while unsaturated hydrocarbons are called alkenes. Alkyl benzenes, such as ethylbenzene, are benzene-based compounds with an alkane group added. Compounds based on benzene that have an alkyl acid group added to them are known as alkyl phenyl acids. One example of this is phenyl propionic acid (benzene propanoic acid. Fatty acid esters are organic acids in which the hydrogen ion has been replaced by an organic group, such as ethyl stearate, which is one of the several soap esters.

Animals often create indole-based chemicals as breakdown products of proteins.

Terpenes are naturally occurring cyclic hydrocarbons found in plants; their usual formula is $(C_5H_8)n$, for example, pinene $(C_{10}H_{16})$, which is the main component of turpentine. Despite being a terpene, limonene was reported independently. Methanethiol or methyl mercaptan is a result of the biological breakdown and a natural substance. Natural gas is odoured with ethyl mercaptan, which has a powerful, unique, and disagreeable odour.

The simplest polyaromatic hydrocarbon is naphthalene (PAH). Two occur in the PAH tables.

The nitro-, chloro-, and methyl-phenols are examples of substituted phenols. Neither tannins nor lignins are included.

Organic sulphides consist mostly of amino acids containing sulphur, as well as other natural substances, such as some compounds found in the onion family. Sulphur is included, but hydrogen sulphide and other inorganic sulphides are excluded.

Table 4.5: *Tentatively Identified Compounds (TIC) in sediment samples*

mg/kg dry wt	Residential		Industrial		Median	Average
	Raw sediment	Digested Sediment	Raw sediment	Digested Sediment		
%DS	1.91	0.92	4.285	36.584		
alkanes and alkenes	32.2	123.8	122.2	42.2	82.2	80.1
alkyl benzenes	2.82	8.84	46.6	6.452	7.646	16.178
alkylphenyl acids	480.7	1307	142.15	/	480.7	643.2833
fatty acid esters	106.3	599	151.6	/	151.6	285.6333
indole based compounds	7.37	95	142.15	/	95	81.50667
Li3540monene	17.25	83	/	/	50.125	50.125
methanethiol	/	7.64	/	/	7.64	7.64
Squalen18e	/	/	/	40.04	40.04	40.04
steroids	675.7	4559	3717.4	11069	4138.2	5005.275
substituted naphthalenes	1.26	/	/	/	1.26	1.26
substituted phenols	/	/	/	1165.4	1165.4	1165.4

sulphides (organic)	5.03	13.4	19.3	/	13.4	12.57667

Table 4.5 reveals that a few chemicals were sometimes discovered at amounts sufficient to justify discussion.

Eucalyptol was detected In three sediments, ranging from 8 to 15 mg/kg. Eucalyptol is the primary component of eucalyptus oil, which has pharmacological use. Camphor, which has both pharmacological and commercial use, was also discovered in the sediments from the industrial areas.

Isocyanates were discovered in three samples between 75 to 130 mg/kg. They were discovered in the sediments from industrial area digesters, as well as the activated sediment from the residential areas. Isocyanates have several industrial applications, including the manufacture of insulating materials.

Benz enamines detected in the sediments from industrial areas, ranging from 2.82 to 46.6 mg/kg. Benz enamines are frequent components of colouring agents.

Alkyl thiols, the sulphur counterparts of alcohol, which consist of mercaptans. Alkyl thiols were detected in activated sediment from the industrial area and dewatered raw sediment from the residential area, with concentrations, ranging from 160 to 300 mg/kg.

Sediment from industrial areas contained between 10 and 60 mg/kg of polyaromatic hydrocarbons. The aerobic sediments from the residential areas had detectable quantities of vitamin E (30 to 60 mg/kg).

In conclusion, there are multiple useful compounds in noticeable concentrations were found in the pond sediment both from the industrial and residential areas. They have pharmacological, commercial use and industrial applications, including the manufacture of insulating materials or medicinally and as an antioxidant in vegetable oils.

4.4   Development of construction material - Strength of Bio-concrete

A variety of laboratory tests were conducted to evaluate the short- and long-term performance of the Bio-Concrete. These test findings are crucial for the design and

construction. There were three elements to the laboratory experiments. First, the optimal mixing ratio was determined; second, wetting and drying experiments were conducted to determine the long-term performance of the Bio-Concrete; and third, the influence of biodegradation on the long-term qualities of the Bio-Concrete was explored.

Table 4.6: *Compressive strength of Bio-Concrete*

Sample	The ratio of binder to sludge (%) by weight				Specimen designation	Compressive Strength (MPa)
	Sediment (%)	Lime (%)	Gypsum (%)	General Purpose Concrete Blend (%)		
1	15	5	0	80	S15-L5-G0-PC80	19.1
2	15	10	0	75	S15-L10-G0-PC75	17.8
3	15	15	0	70	S15-L515-G0-PC70	13.7
4	15	20	0	65	S15-L20-G0-PC65	8.6
5	15	0	5	80	S15-L0-G5-PC80	25.1
6	15	0	10	75	S15-L0-G10-PC75	26.3
7	15	0	15	70	S15-L0-G15-PC70	24
8	15	0	20	65	S15-L0-G20-PC65	17.1
9	10	0	5	85	S10-L0-G5-PC85	29.2
10	10	0	10	80	S10-L0-G10-PC80	30.7
11	10	0	15	75	S10-L0-G15-PC75	29.7
12	10	0	20	70	S10-L0-G20-PC70	19.2
13	5	0	5	90	S5-L0-G5-PC90	34.9
14	5	0	10	85	S5-L0-G10-PC85	35.6
15	5	0	15	80	S5-L0-G15-PC80	31.3
16	5	0	20	75	S5-L0-G20-PC75	21.9
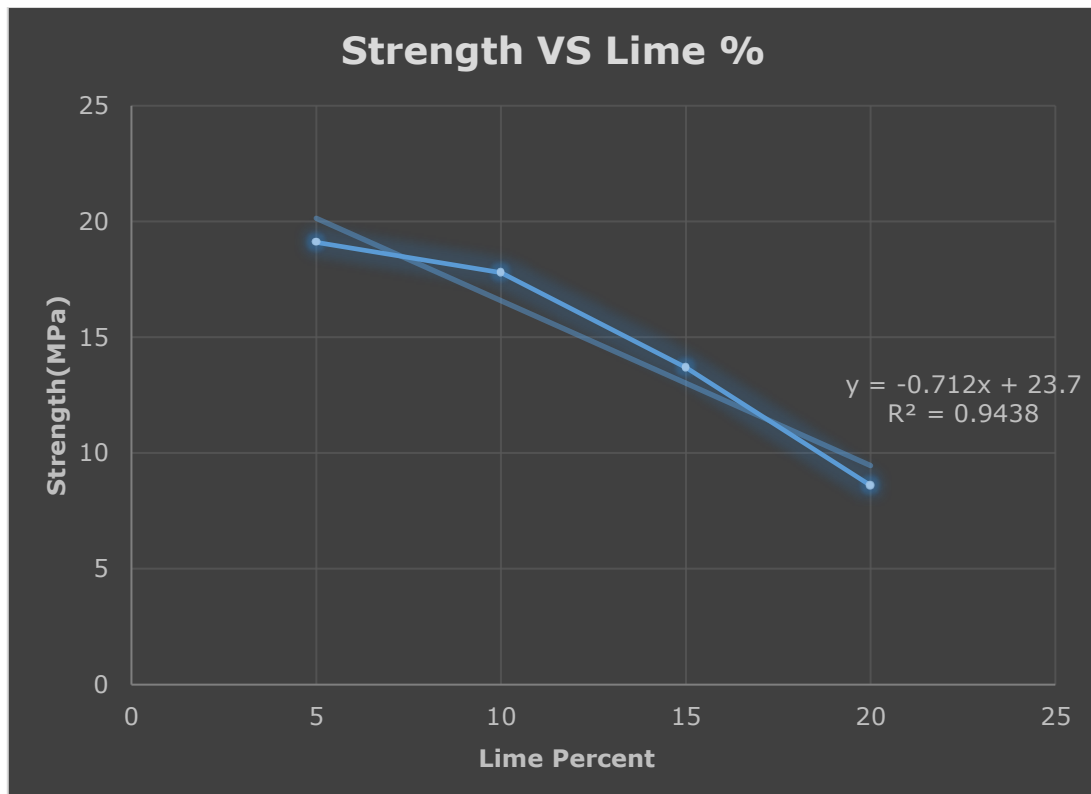
Figure 4-15: Unconfined compression strength versus lime percentage by weight

As can be seen from **Figure 4-15**: Unconfined compression strength versus lime percentage by weight, there is a strong and negative linear regression between the unconfined compression strength and lime percentage. An increase in lime dose led to a significant decrease in unconfined compression strength. The precise cause of this finding is unknown, but it is probably due to the short-term response (mostly drying sludge and inducing sludge particle flocculation and aggregation) and the long-term pozzolanic reaction, the mechanism of lime solidification is recognised (contributing to strength development). It seemed that a larger lime dose impeded the pozzolanic response. For the pozzolanic process to continue, sufficient soluble silica and alumina must be present in the concrete blend, which contains Portland cement, aggregates, and sand, to react with the lime. In this instance, it is quite probable that the quantity of lime supplied surpassed the pozzolanic reaction's need, given the amount of lime added was more than that of the concrete blend. The excessive use of lime compared to concrete blend decreased the Bio-Concrete UCS (Bell, 1996).

Therefore, lime is not an appropriate additive to Bio-concrete, as it decreases the UCS significantly. Sample 5 – Sample 16 will not contain any lime.
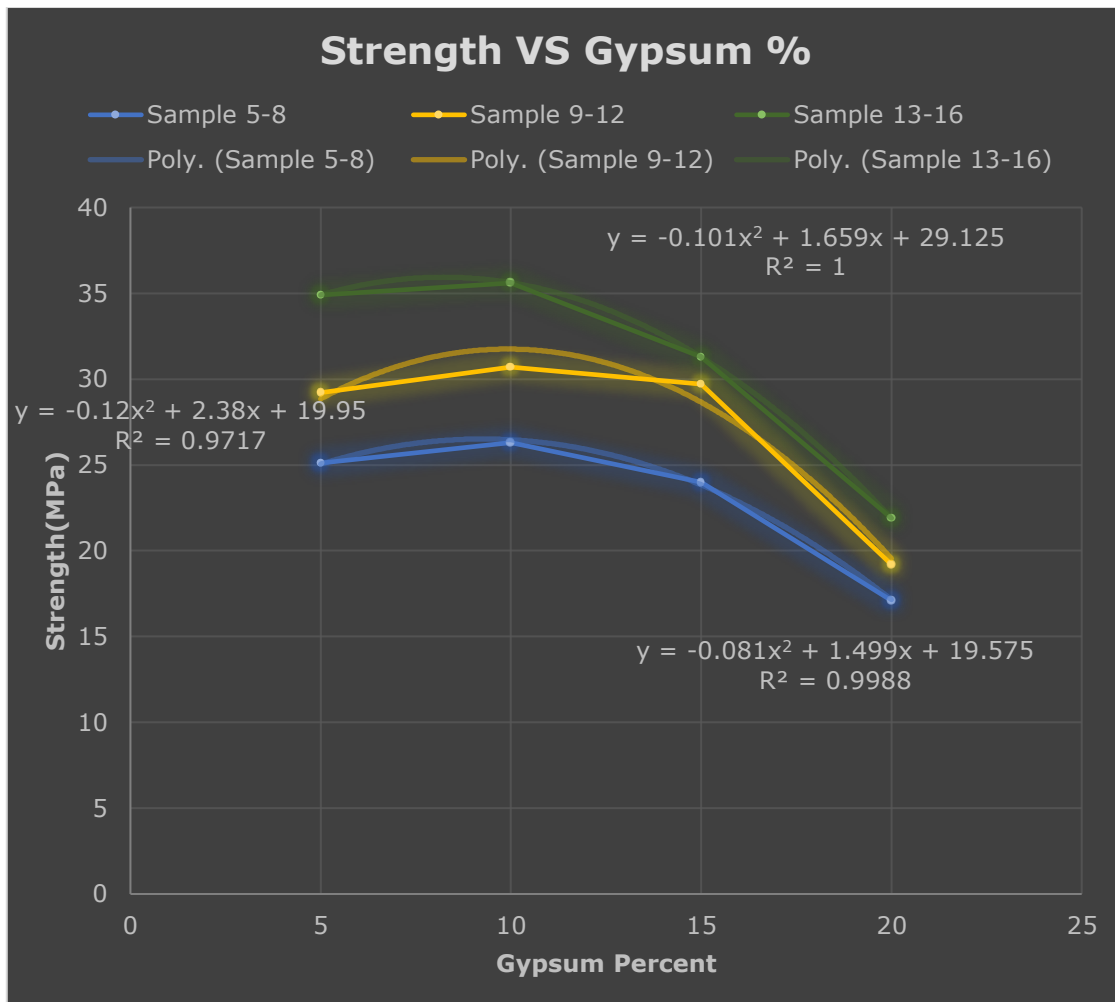
Figure 4-16 Unconfined compression strength versus gypsum percentage by weight

As can be seen from **Figure 4-16** Unconfined compression strength versus gypsum percentage by weight, there is a significantly strong polynomial regression between the unconfined compression strength and gypsum percentage. It is a concave down shape so an increase in gypsum dose led to a slight increase in unconfined compression strength from 34,9MPa to 35.6 MPa then gypsum percentage will dramatically decrease the unconfined compression strength of Bio-concrete from 35.6 MPa to 21.9MPa.

To control the drying period of cement, gypsum is added as a retarding agent. Condensation may occur instantly when water is added to cement clinker without the use of gypsum. An insufficient gypsum concentration will have no discernible slowing effect. Because gypsum may produce its coagulating agent, adding too much of it can hasten the setting of cement. The fineness of cement and the SO3 content of clinker influence the ideal quantity of gypsum to use, as do the C3A and SO3 contents of gypsum. Gypsum should make up between 3 and 5

per cent of the total weight of the cement. If the gypsum concentration is too high, the cement's strength will suffer, and the paste's dimensional stability will suffer, leading to a more rapid breakdown. This means that the maximum allowable level of SO3 is 3.5%, as stipulated by the national standard. (Asakura, 2013)

Furthermore, it can be concluded from Fig.2 that a higher dosage of sediment from a stormwater pond will gradually decrease the unconfined compression strength from 35.6 MPa to 8.6 Mpa.

In this case, gypsum is an appropriate binder and the optimal percentage of gypsum in the Bio-concrete is 10%. The highest unconfined compression strength is 35.6 MPa and the best mixing ratio is 5% of sediments, no lime, 10% of gypsum, and 85% of the general-purpose concrete blend(S5-L0-G10-PC85). It is confident to predict that the best gypsum percentage is 8.3% at the turning point of the concave shape and the strongest Bio-concrete has an unconfined compressive strength of 35.93MPa.

4.5   Potential Commercial Reuses of Sediment

- Useful Tentatively Identified Compounds (TIC)

Eucalyptol was detected in three sediments, ranging from 8 to 15 mg/kg. Eucalyptol is the primary component of eucalyptus oil, which has pharmacological use. Camphor, which has both pharmacological and commercial use, was also discovered in the sediments from the industrial areas.

Isocyanates were discovered in three samples between 75 to 130 mg/kg. They were discovered in the sediments from industrial area digesters, as well as the activated sediment from the residential areas. Isocyanates have several industrial applications, including the manufacture of insulating materials.

Benz enamines were detected in the sediments from industrial areas, ranging from 2.82 to 46.6 mg/kg. Benz enamines are frequent components of colouring agents.

In conclusion, there are multiple useful compounds in noticeable concentrations were found in the pond sediment both from the industrial and residential areas. They have pharmacological, commercial use and industrial applications, including the manufacture of insulating materials or medicinally and as an antioxidant in vegetable oils.

- Biopesticide biotech products

Biopesticide biotech products have also been developed and prepared with tremendous potential for use in the field. Enzymes and the synthesis of bioplastics are both undergoing similar studies. However, some products, such as biosurfactants and bio flocculants, are still in their infancy, and their productivity may be improved in the future.

- Land Recovery - Leachable heavy metals amendments

There are multiple leachable heavy metals in sediments from the stormwater ponds and a few of them request attention and amendment before applying to land recovery(e.g. wastelands from the mining industry). After several stabilisation trials in Sections 4.2.2, 4.2.3 and 4.2.4, several stabilisation attempts using phosphate, compost, and alkaline modifiers were detailed. According to the results of these studies, leachate concentrations were reduced by roughly 60-70% for lead and copper and 55% for zinc. Furthermore, DOC is a significant factor when applying amendments to leachate zinc. Although zinc removal efficiency was greater in sediment from the residential area, this is likely due to lower leachate DOC concentrations (the average is about 33 mg/L). Comparatively, the average concentration of DOC in the samples from the industrial area was 207 mg/L. Higher DOC concentrations promote the solubilisation of heavy metals, hence diminishing the efficacy of stabilisation procedures.

- Construction materials

The development of items such as construction materials is exhibiting a positive trend, thanks to new technical developments to minimize metal leaching and hazardous gas emissions.

In the development of construction materials, gypsum is an appropriate binder and the optimal percentage of gypsum in Bio-concrete is 10%. The highest unconfined compression strength is 35.6 MPa and the best mixing ratio is 5% of sediments, no lime, 10% of gypsum, and 85% of the general-purpose concrete blend(S5-L0-G10-PC85). It is confident to predict that the best gypsum percentage is 8.3% at the turning point of the concave shape and the strongest Bio-concrete should have an unconfined compressive strength of 35.93MPa

5 CONCLUSIONS

To accomplish the aim of zero landfills by 2040, sludge and sediments from stormwater retention ponds should be adequately treated and reused. This can be accomplished by anaerobic digestion, aerobic fermentation, or other novel approaches such as combining sludge or sediment handled in manufacturing bricks and cement. Stricter environmental restrictions, as well as the negative health and environmental implications of disposal procedures (such as incineration and landfilling), have prompted the development of value-added sludge management strategies. Although the presence of smells, infections, and heavy metals has historically hampered the rate of sludge reuse, these variables have been decreased and/or controlled to acceptable levels by improving/or optimizing the manufacturing process. Engineered soils, decorative horticultural fertilizers, turfs, synthetic carbon, and activated carbon are some of the goods that have been developed as a result of the added value of sediments.

In conclusion, the recovery of sediments is an ecologically and socially desirable effort. There are quantities of tentatively identified compounds identified in sediments and they have pharmacological, commercial use, and industrial applications, including the manufacture of insulating materials or medicinally and as an antioxidant in vegetable oils. Furthermore, they can be blended into concrete with certain ratios, without lowering too much strength of it. Last but not least, sediments can be used for land recovery in mining industries, due to their high organic matter content after applying amendments on leachable heavy metals. This research could provide a wider scope for future sediments' utilisation and commercialisation.

6 APPENDIX

6.1 Original Results of ICP-MS

Original Results of ICP-MS							
Data File	Acq. Date-Time	Type	Level	Sample Name	Vial Number	Comment	Total Dil.
004CALB.d	4/10/2022 18:05	CalBlk	1	Blank	1101		1
005CALS.d	4/10/2022 18:10	CalStd	2	STD 1	1102		1
006CALS.d	4/10/2022 18:15	CalStd	3	STD 2	1103		1
007CALS.d	4/10/2022 18:21	CalStd	4	STD 3	1104		1
008CALS.d	4/10/2022 18:26	CalStd	5	STD 4	1105		1
009SMPL.d	4/10/2022 18:31	Sample		rinse	4		1
010SMPL.d	4/10/2022 18:36	Sample		rinse	4		1
011SMPL.d	4/10/2022 18:42	Sample		QC Check	1106		1
012SMPL.d	4/10/2022 18:47	Sample		rinse	4		1
014QBLK.d	4/10/2022 18:58	FQBlk		- Blank	1201		1
015SMPL.d	4/10/2022 19:03	Sample		B-1	1202	+ 20x dilute	2986
016SMPL.d	4/10/2022 19:08	Sample		B-2	1203	+ 20x dilute	2231

017SMPL.d	4/10/2022 19:14	Sample		B-3	1204	+ 20x dilute	2059
018SMPL.d	4/10/2022 19:19	Sample		B-4	1205	+ 20x dilute	1951
019SMPL.d	4/10/2022 19:25	Sample		B-5	1206	+ 20x dilute	1399
020SMPL.d	4/10/2022 19:30	Sample		B-6	1207	+ 20x dilute	1979
021SMPL.d	4/10/2022 19:35	Sample		B-7	1208	+ 20x dilute	1836
022SMPL.d	4/10/2022 19:41	Sample		B-8	1209	+ 20x dilute	1880
023SMPL.d	4/10/2022 19:46	Sample		rinse	4		1
024SMPL.d	4/10/2022 19:51	Sample		STD 3 - check	1104		1
025SMPL.d	4/10/2022 19:57	Sample		rinse	4		1

66 Zn [He]			75 As [He]		
Conc. [ug/g]	Conc. % RSD	CPS	Conc. [ug/g]	Conc. % RSD	CPS
0.00	N/A	28289	0.00	N/A	1223
0.01	1.85	135476	0.00	2.19	5559
0.10	0.44	1095790	0.01	2.85	47654
0.49	0.68	5212270	0.05	2.75	236146
1.00	0.46	10415709	0.10	1.88	477088

Conc. [ug/g]	Conc. % RSD	CPS	Conc. [ug/g]	Conc. % RSD	CPS
<0.000000	N/A	19463	0.00	11.41	14106
<0.000000	N/A	18080	0.00	9.26	5889
1.01	1.45	10149600	1.09	1.43	5002023
<0.000000	N/A	16205	0.02	10.18	83580
<0.000000	N/A	15695	0.00	11.93	15143
257.70	0.71	949402	4.32	14.83	22935
212.59	1.06	1058221	5.06	6.33	27296
38.76	0.35	229269	0.49	55.45	17680
31.51	0.94	195337	<0.000000	N/A	13831
15.67	0.96	138161	<0.000000	N/A	11358
24.20	0.47	142821	<0.000000	N/A	12464
14.31	1.12	103344	<0.000000	N/A	11788
27.25	1.70	172830	<0.000000	N/A	7731
<0.000000	N/A	4121	<0.000000	N/A	3232
0.52	0.21	5467747	0.05	2.04	246561
<0.000000	N/A	5953	<0.000000	N/A	8568

66 Zn [He]			75 As [He]		
Conc. [ug/g]	Conc. % RSD	CPS	Conc. [ug/g]	Conc. % RSD	CPS
0.00	N/A	28289	0.00	N/A	1223
0.01	1.85	135476	0.00	2.19	5559
0.10	0.44	1095790	0.01	2.85	47654
0.49	0.68	5212270	0.05	2.75	236146
1.00	0.46	10415709	0.10	1.88	477088

<0.000000	N/A	19463	0.00	11.41	14106
<0.000000	N/A	18080	0.00	9.26	5889
1.01	1.45	10149600	1.09	1.43	5002023
<0.000000	N/A	16205	0.02	10.18	83580
<0.000000	N/A	15695	0.00	11.93	15143
257.70	0.71	949402	4.32	14.83	22935
212.59	1.06	1058221	5.06	6.33	27296
38.76	0.35	229269	0.49	55.45	17680
31.51	0.94	195337	<0.000000	N/A	13831
15.67	0.96	138161	<0.000000	N/A	11358
24.20	0.47	142821	<0.000000	N/A	12464
14.31	1.12	103344	<0.000000	N/A	11788
27.25	1.70	172830	<0.000000	N/A	7731
<0.000000	N/A	4121	<0.000000	N/A	3232
0.52	0.21	5467747	0.05	2.04	246561
<0.000000	N/A	5953	<0.000000	N/A	8568

111 Cd [He]			201 Hg [He]		
Conc. [ug/g]	Conc. % RSD	CPS	Conc. [ug/g]	Conc. % RSD	CPS
0.00	N/A	40	0.00	N/A	589
0.00	1.15	17227	0.00	1.29	7299
0.01	0.84	169865	0.01	0.98	93048
0.05	0.58	843000	0.05	0.61	496080

0.10	0.68	1673025	0.10	1.00	1043343
0.00	42.83	119	0.00	2.17	12803
0.00	41.78	89	0.00	2.82	6210
0.94	0.49	15288557	0.00	3.21	6798
0.00	13.23	514	0.00	4.60	2716
0.00	38.08	95	0.00	10.73	2040
0.12	10.96	791	<0.000000	N/A	1744
0.36	1.84	2950	0.10	33.43	2647
0.05	10.53	543	<0.000000	N/A	1417
0.06	9.00	617	<0.000000	N/A	1335
0.10	6.52	1415	<0.000000	N/A	1165
0.07	10.03	657	<0.000000	N/A	875
0.01	30.72	157	<0.000000	N/A	929
0.03	14.78	381	<0.000000	N/A	723
0.00	332.61	103	<0.000000	N/A	389
0.05	0.31	857489	0.04	0.91	452666
0.00	83.17	259	0.00	9.56	4080

208 Pb [He]			232 Th [He]		
Conc. [ug/g]	Conc. % RSD	CPS	Conc. [ug/g]	Conc. % RSD	CPS
0.00	N/A	8689	0.00	N/A	483
0.01	0.60	1310741	0.00	18.56	32944
0.10	0.34	13749911	0.01	11.31	477332
0.50	0.65	68604169	0.04	6.49	3922867

1.00	0.64	137254689	0.10	3.77	9810065
0.00	4.74	31424	0.00	5.20	198055
0.00	4.01	20560	0.00	4.95	50745
0.98	0.93	129578180	0.01	4.29	490734
0.00	4.59	32358	0.00	4.94	9781
0.00	8.57	11845	0.00	7.13	7436
11.32	0.36	554135	0.61	4.84	27746
13.90	0.80	912695	0.85	4.80	45444
7.45	1.29	551687	1.94	1.43	104225
17.11	0.47	1294620	2.47	2.13	134202
3.26	0.82	347415	1.57	2.79	117736
2.62	0.57	194002	0.98	3.70	54121
4.56	1.31	377053	2.35	4.58	136291
3.58	1.27	284005	1.42	4.39	81262
0.00	8.58	16682	<0.000000	N/A	6025
0.49	0.52	68120253	0.05	4.44	4713394
0.00	47.74	33042	0.00	3.32	109064

238 U [He]			159 Tb (ISTD) [He]		
Conc. [ug/g]	Conc. % RSD	CPS	ISTD Recovery %	CPS	CPS RSD
0.00	N/A	21	100.00	374589	1.01
0.00	0.89	266562	100.67	377094	0.79
0.01	0.83	2887184	101.51	380256	1.08
0.05	0.60	14266947	101.77	381215	0.83

0.10	0.56	28659975	100.67	377116	0.72
0.00	13.68	7732	99.12	371282	1.16
0.00	14.56	2276	98.10	367476	2.02
0.00	14.20	1253	97.52	365303	2.49
0.00	12.36	656	98.10	367486	0.58
0.00	20.26	371	100.70	377206	1.20
0.23	2.24	23278	105.05	393514	1.23
0.46	0.90	62275	106.20	397822	0.84
0.36	1.78	54697	109.49	410118	0.48
0.39	2.08	61625	107.44	402441	0.95
0.36	0.96	78294	105.56	395434	1.14
0.27	1.51	38890	100.99	378314	0.57
0.43	0.79	71342	107.85	403975	0.83
0.27	0.56	42899	104.93	393073	1.45
<0.000000	N/A	151	100.79	377546	1.03
0.05	1.22	13960197	101.45	380002	1.77
0.00	34.26	5371	99.41	372369	1.13

6.2 Unconfined compressive strength calculation

Unconfined compressive strength is calculated by:

Breaking Load(kN)/ Area(mm^2) = Unconfined compressive strength

Sample	D1(mm)	D2(mm)	MEAN D(mm)	Length(mm)	Weight(kg)	Area(mm^2)	Density(kg/m^3)	Breaking Load(kN)	Com Strength (MPa)
7	100	100	100	196.4	3.54	7853.981	2294.943007	274	34.8867641
4	101.5	101.08	101.29	201.6	3.73	8057.921	2296.123735	236	29.2879508
6	100.1	100	100.05	198.56	3.62	7861.837	2318.957271	186	23.6585914
5	100.08	100.1	100.09	195.92	3.62	7868.1	2348.328	150	19.0642623
1	100.1	100	100.05	196.5	3.42	7861.837	2213.80565	140	17.8075419
3	101.18	101	101.09	197.9	3.51	8026.131	2209.810614	110	13.7052328
8	99.5	100.1	99.8	198.64	3.41	7822.596	2194.505714	104	13.2948175
2	100.2	100	100.1	197.6	3.34	7869.697	2147.837881	68	8.64073893

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