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Transformation of solid and liquid wastes into fertiliser to minimize urban catchment pollution.

PhD

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Abstract

Decentralised treatment systems play a critical role in managing faecal sludge (FS) in sub-Saharan Africa (SSA) where safely managed sanitation is lagging with 79% of the population without it. The systems that treat FS and food waste (FW) into faecal derived fertilisers (FDFs) play a more critical role in linking safely managed sanitation to improved food security. To recover nutrients and organic matter to their fullest potential is urgent in the context of SSA. The International Water Management Institute (IWMI) and partners have led research to convert urban FS and FW through co-composting into various FDF for use in agriculture. Though some strides have been made in developing and commercialising FDF, there are still some research and knowledge gaps including limited information on nutrient and pathogen flow in the decentralised FS and FW treatment system; limited information on the shelf life of stored FDF and on residual effect of FDF application on crops and soil. Hence, this study aimed to generate new knowledge and understanding on the recovery of nutrients and *E. coli* inactivation during treatment and use of fertiliser produced from FS and solid waste. The methodology involved different experimental set-ups to collect primary data. This followed an end-to-end monitoring of FS and FW treatment to produce FDF, storage of FDF and the use of the FDF in successive lettuce cultivation. Findings from this study, show that between 50-70% of total N from FS is lost at the dewatering stage of treatment. More than 50% of total N is lost during co-composting. While *E. coli* inactivation efficiency of the dewatering process is minimal (0-14%) in the percolate, dewatered FS on the other hand observed higher *E. coli* inactivation efficiency of 88-98% (1-2 log reductions). Inactivation efficiency of co-composting stage for *E. coli* was 100%. No detectable presence of indigenous *E. coli* was observed in FDF at the end of storage. Storage temperature and duration did not affect re-growth of indigenous *E. coli* in co-composted FDF. Longer storage of enriched FDF co-compost (NECo) under lower temperatures resulted in decreasing NH_4-N concentrations. The field experiment show, residual effect of FDF co-compost (Co) gave lettuce yield of 344% more compared to the control by the second cycle. *E. coli* was absent on lettuce after successive cultivations. Co plots had higher gross margins/profit per cycle of cultivation. The ROI for Co was 385.7 for first cycle and 309.2 for second cycle.

Keywords: faecal sludge, nutrients, faecal derived fertiliser, pathogen, storage, lettuce

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Table of Contents

List of Figures

List of Tables

List of Boxes

Abbreviations

FS	Faecal sludge
FW	Food waste
DFS	Dewatered faecal sludge
FDF	Faecal derived fertiliser
NECo	Enriched FDF co-compost
Co	FDF co-compost
SSA	sub-Saharan Africa
OSS	Onsite sanitation systems
SD	Standard deviation
ROI	Return on investment
CBR	Cost benefit ratio

Chapter 1 Introduction

1.1 Sanitation and Waste Management Status in sub-Saharan Africa

Poor sanitation and inadequate waste management are growing concerns facing the global south. Whilst globally it has been estimated that more than 2 billion people worldwide (> 25% of the world's population) do not have access to basic sanitation (WHO and UNICEF, 2019), a recent estimate by WHO/UNICEF JMP (2021) puts this figure at 1.7 billion of people lacking even basic services. In 2020, nearly five out of ten people used safely managed sanitation services. "Safely managed" sanitation services represent a higher service level that takes into account the final disposal of excreta, in addition to the "basic" service level which requires an improved sanitation facility (such as flush toilets or latrine with a slab) not shared with other households (WHO/UNICEF JMP, 2021). Still, in 2020 3.6 billion people lacked safely managed services, of which approximately half (1.9 billion) had basic services. In terms of regional coverage, an estimated 21% of the population in sub-Saharan Africa (SSA) had access to safely managed sanitation in 2020 (Figure 1-1). This leaves a staggering 79% of the population in SSA without safely managed sanitation resulting in open defaecation, open dumping, and indiscriminate dumping in water ways etc.

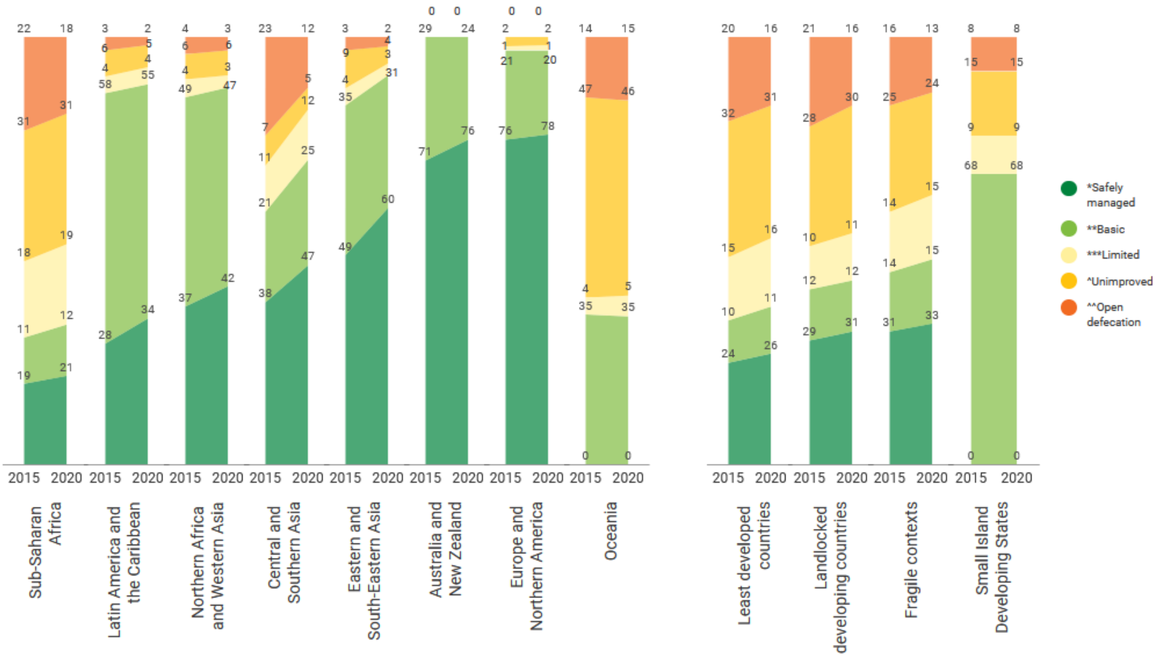

Figure 1-1. Global coverage of safely managed sanitation by region. Source: WHO/UNICEF JMP (2021)

An estimated 2.01 billion tonnes of municipal solid waste is generated annually and it is expected to increase by almost 70% by 2050 globally (Kaza et al., 2018). The SSA region alone generated 174 million tonnes of waste in 2016, at a rate of 0.46 kilogram per capita per day. It is the fastest growing region, with waste expected to nearly triple by 2050 (R. Singh & Singh, 2022). Waste in SSA is mainly organic, with 40% of it being food and green waste (Kaza et al., 2018). Overall waste collection rates are about 44%, although the rate is much higher in urban areas than in rural areas, where waste collection services are minimal (Kaza et al., 2018). Nearly every aspect of human lives produce waste. Rapid urbanization in SSA (42% in 2021 – The World Bank (2022)) is turning urban areas in SSA into waste sinks. As cities rapidly grow, so is the total volume of waste generated, bringing with it the enormous challenges of collecting and safely managing these wastes. About 69% of solid waste is openly dumped in ditches, open burning etc although use of landfills and recycling systems is becoming more prevalent in SSA (Kaza et al., 2018) while 79% of sanitation related waste (liquid waste) is not safely managed. The release of such untreated wastes into the environment in the form of open dumping is continuously leading to pollution of freshwater resources and ultimately marine littering and pollution (UNEP, 2009). Nutrient and pathogen load in water resources cause eutrophication and sanitation related public health risks (Pandey et al., 2014; Yang et al., 2008). Other effects of open dumping are accelerated sedimentation of waterbodies, reducing their effective volumes, and carrying capacities to absorb urban run-off and storm water (Osouli et al., 2017). This phenomenon is quite pronounced in lagoons and wetlands, which are internationally, recognized protected areas.

1.2 Agriculture in sub-Saharan Africa

African economies rely heavily on agriculture. It is estimated that the contribution of agriculture to the gross domestic product (GDP) is as high as 75% in some SSA countries (Pan Africa Chemistry Network, 2012). However, agricultural productivity has not improved or in some countries has suffered decline in Africa making feeding the population in Africa a complex and challenging target. Most of the farming in Africa is carried out by smallholder farmers. A situation that makes food security even more acute due to the combination of factors such as declining soil fertility, climate change variability and low uptake of new technologies (Muzari et al., 2012; Pan Africa Chemistry Network, 2012; Shahzad et al., 2021). In addition, environmental stresses such as limited water availability as well as legacies from the unsustainable management of natural resources and poor agricultural practices (soil erosion and depletion of minerals) further compound the situation. A major revitalisation of agriculture is needed to underpin development ahead of the drivers of population increase, demographic change, and climate change (Pan Africa Chemistry Network, 2012).

New technologies and approaches for sustainably increasing agricultural productivity in Africa are essential. These new technologies must consider the nature of the farming systems in Africa and also consider the socioeconomic conditions of the continent as millions of the population are considered poor. The solutions must be scalable, affordable, and able to reach those who need them the most to have significant impact (Ajayi et al., 2018). For instance, external nutrient application is essential in reducing the soil nutrient deficit. The application of greater amounts of mineral fertiliser will help to address deficiencies in key nutrients, nitrogen, potassium, and phosphorus. However, alongside measures to help smallholder farmers gain access to vital nutrients needed to boost yields, a long-term strategy for monitoring and managing soil health in Africa is needed (Garrity et al., 2010).

1.2.1 Mineral Fertiliser Supply and Use in sub-Saharan Africa.

The global increase in population and greater per capita incomes anticipated through mid-century has necessitated global demand for mineral fertiliser to continuously be on the rise as major fertiliser sources like phosphorus reserves are finite and limited (Godfray et al., 2010; Vanotti et al., 2019). The SSA region currently uses only about 8kg/ha of fertiliser which is less than one tenth of the world average (Wanzala & Groot, 2013). Kenya, Ghana and Madagascar use 26, 15 and 4 kg/ha respectively (NEPAD-CAADP, 2015) indicating a wide range of fertiliser application from various parts of SSA. Critical reasons for this scarce use of fertiliser are its non-availability and the high cost associated with it. Access to finance is a major barrier to increasing fertiliser application and African governments need to improve access to finance for smallholder farmers. Aside finance, access to the fertilisers and alternatives such as the use of nitrogen-rich plants and other organic sources must be guaranteed (Pan Africa Chemistry Network, 2012).

For instance, in Ghana, mineral fertiliser shortages have been reported in various parts of the country in 2021 (Apubeo, 2021; Ashon, 2021). There has been speculations of fertiliser hording and smuggling to other neighbouring countries among others being the causes of fertiliser shortage in the government's program of Planting for food and Jobs (GhanaWeb, 2021). These developments have had direct dire consequences for smallholder farmers productivity and food security (Adzawla et al., 2021). Mineral fertiliser production is energy intensive and energy costs are high and consequently influencing fertiliser prices to go up. According to Baffes and Koh (2022), fertiliser prices have risen nearly 30% since the start of 2022, following the 80% surge in 2021. Soaring prices are driven by a confluence of factors, including surging input costs, supply disruptions caused by sanctions (Belarus and Russia), and export restrictions (China) (Baffes & Koh, 2022). The energy used for fertiliser production is largely due to production of nitrogen fertilisers, the main energy carrier being natural gas (Ahlgren et al., 2011). Rising natural gas prices, especially in Europe has led to widespread production cutbacks in ammonia which contributed to the increase in prices. The prices were also affected by the war in Ukraine, reflecting the impact of economic sanctions and disruptions in Black Sea trading routes (Baffes & Koh, 2022; Hebebrand & Laborde, 2022).

As countries in Africa are also heavily dependent on importation of mineral fertiliser and food such as cereal from Ukraine, the current context significantly affects food security in Africa. Statistics show that, in 2019, Africa's total value of food imports registered roughly 81 billion U.S. dollars. The main contributor to the continent's food imports was cereals and preparations, with an import value of over 28 billion U.S. dollars. Fats and oil imports stood at roughly 9.7 billion U.S. dollars (Kamer, 2023). Due to the global energy crisis and the economic sanctions in geopolitically instable areas, staple food prices in sub-Saharan Africa surged by an average of 23.9% in 2020-2022, the most since the 2008 global financial crisis (Okou et al., 2022). Because the region imports most of its top staple foods—wheat, palm oil, and rice—the pass-through from global to local food prices is significant, nearly one-to-one in some countries. Prices of locally sourced staples have also spiked in some countries on the back of domestic supply disruptions, local currency depreciations, and higher fertiliser and input costs. In Nigeria for example, the prices of both cassava and maize more than doubled even though they're mainly produced locally. In Ghana, prices for cassava escalated by 78 percent in 2020-21, reflecting higher production costs and transport constraints, among other factors (Okou et al., 2022).

1.3 Linking Sanitation and Waste Treatment to Agriculture in a Changing Climate

A way of improving sustainable agriculture is by linking it with sustainable waste management through recovery of valuable resources (water, nutrients, and organic matter) from organic fractions of municipal waste streams and sanitation systems such as faecal sludge (FS) thus promoting a circular economy (Keraita et al., 2008; Nikiema et al., 2014). Smallholder farmers in SSA can benefit from more readily available and affordable organic alternative fertilisers such as treated/composted FS if its full potential is realised. Academia, non-governmental organisations, small and medium scale enterprises (SMEs) in SSA can work together to realise the benefits of treated FS as an alternative to mineral fertilisers. Information on product safety is very important to instil confidence in the produce (SDG 2) that will be consumed. A good source of fertiliser in addition to other sustainable farming practices provide a good base to improve soil health (SDG 15) and ensure adequate supply of food (SDG 2). As the state of land in most parts of Africa is degraded (SDG 15), one way of improving soil health is through use of organic sources of fertiliser such as the treated FS (O. Cofie et al., 2009; Nikiema et al., 2013).

Increasingly, sanitation systems incorporating reuse are being positioned within the circular economy discourse (Danso et al., 2017; Mallory et al., 2020; Moya et al., 2019; Schroeder et al., 2019). An era of linking various processes that in the past were not feasible for a more sustainable and holistic approach to solving cross cutting global challenges. The concepts around resource recovery and reuse as well as resource use efficiency are long established and known. However, the notion of wide availability of relatively cheap mineral fertilisers, such as phosphate ($PO4^{3-}$) derived from mined rock phosphates (Cordell et al., 2009) and ammonia (NH_3) produced by the Haber-Bosch process (Maurer et al., 2002) downplayed the necessity to recycle essential plant nutrients contained in waste and manure back to crop production (Kuokkanen et al., 2017). What makes the situation more urgent now is a convergence of several factors including the global energy crisis and its effect on fertiliser prices, geopolitical risks and conflicts around natural resources, climate change urgency, competition in the markets for natural resources, the finite nature of some resources such as phosphate rocks (Cordell et al., 2012), and the growing scientific and technological know-how makes it the right time (Lassaletta et al., 2014; Schröder et al., 2011) to encourage circular economy of resources.

Organic waste such as agricultural wastes (crop residues, animal manure, poultry litter) and municipal waste (food waste, FS and wastewater) (Baudron et al., 2014; Drechsel & Dongus, 2010) supply organic matter to soils. Organic matter is important to highly degraded soils especially in SSA by improving soil structure, which results in increased water holding capacity and enhances root growth in addition to other essential nutrients supply. Food waste is defined as the use of food meant for consumption by humans for non-consumption purposes, the redirection of food to feed animals, or the disposal of edible food (FAO, 2014). It includes the edible as well as inedible parts of food that get removed from the food supply chain and which can be recovered or managed through disposal (Östergren et al., 2014). Globally, around 14% of food produced is lost between harvest and retail, while an estimated 17% of total global food production is wasted (11% in households, 5% in the food service and 2% in retail) (UN, 2022) costing the global economy close to $940 billion each year (FAO, 2014). Food loss and waste undermine the sustainability of our food systems. When food is lost or wasted, all the resources that were used to produce this food - including water, land, energy, labour, and capital - go to waste. In addition, the disposal of food loss and waste in landfills, leads to greenhouse gas emissions, contributing to climate change (UN, 2022). Up to 10% of global greenhouse gases comes from food that is produced, but not eaten (UNEP, 2021).

Food waste burdens waste management systems, increases food insecurity and is a major contributor to the global problems of climate change, biodiversity loss and pollution. In SSA, the estimate is roughly 37% or 120-170 kg/year per capita (FAO, 2011b). In South Africa, 10 million tonnes of food go to waste every year. This accounts for a third of the 31 million tonnes that are produced annually in South Africa. Together, fruits, vegetables and cereals account for 70% of the wastage and loss. This wastage and loss primarily occur early in the food supply chain (WWF, 2017). While some of that waste gets incorporated back into that value chain, in the form of animal feed or fertiliser, a whopping 90% of this food waste (WWF, 2017) gets distributed into landfills. This uncontrolled food degradation in the landfills inevitably leads to the production of environmentally harmful greenhouse gases, like methane (Bessa et al., 2021). A key solution to restoring Africa's land involves rethinking the way we manage food – from what we grow, to what we dispose of and mostly importantly how we dispose of it. A shift from viewing food waste as a problem to one where it can provide a rich foundation for regenerative farming can fundamentally accelerate restoring land soil health, as well as improve food resilience (Bessa et al., 2021). Currently, less than 2% of valuable nutrients in food by-products and waste are recycled, and most of them end up in landfills where they are left to rot and produce greenhouse gases, or else be incinerated (WWF, 2017).

Reusing treated/co-composted FS and food waste for agricultural purposes reduces greenhouse gas emissions, improves sanitation and waste management, saves expenditure for mineral fertilisers, improves soil fertility, reduces poverty and ensures food security (Mariwah & Drangert, 2011). Reusing treated FS and food waste could help tackle some of the Sustainable Development Goals (SDGs) such as SDG 2 and SDG 15 (Moya, et al., 2019). SDG 2 for instance aims to achieve zero hunger with target 2.3 aiming to double the agricultural productivity of small-scale farmers by increasing access to inputs and to markets (UN, 2015). The use of treated FS has become an important policy discussion in the global south as a way to manage urban sanitation to benefit agriculture (Burt et al., 2021).

1.4 Waste Treatment Systems for Reuse (Circular Economy) in sub-Saharan Africa

There have been attempts to implement conventional wastewater treatment technologies (e.g., rotating biological contactors, activated sludge etc.) in SSA to manage liquid waste. But they have not been well adapted to the local context for a number of reasons, among which the high cost of installation, the availability of a reliable energy supply, and local skills and human resources are prominent (Edokpayi et al., 2017; Nowak et al., 2015; Spuhler & Gensch, 2019).

Local authorities in SSA spend huge sums of money to dispose of FS as waste (Mariwah & Drangert, 2011). In that, when these treatment plants were built, they were solely operated and maintained to treat FS for disposal purposes. Where treatment plants were inadequate or non-existent, the sludge is discharged in open drains, open lands and marine environment (Boot & Scott, 2008). Paradigm shifts from ‘treatment for disposal’ to ‘treatment for reuse’ is beginning to gain grounds in SSA because of the huge impact on the local economy and the environment. Innovative decentralised treatment systems that treat and enable productive, safe reuse of water, and facilitate recovery of nutrients and organic matter from waste resources and FS are booming in SSA (Ahmed et al., 2019; Qadir et al., 2020). Examples of the innovative decentralised treatment systems are decentralized separation of waste flows, low-or no-flushing toilets, and converting FS to energy (Andersson et al., 2018; Mrimi et al., 2020; Semiyaga et al., 2015). There are growing examples of successful innovative decentralised treatment systems that treat FS either on-site or off-site from on-site sanitation containment systems (Mrimi et al., 2020; Semiyaga et al., 2015).

These innovations do not only focus on resource-recovery as a way to help offset operational costs, but also to create win-win scenarios between sanitation and waste management and other sustainability goals like clean energy, sustainable consumption and production, and food security (J. R. McConville et al., 2020). One of such decentralised treatment is the FS treatment system comprising of FS dewatering on sand drying beds and coupled with aerobic (open air) co-composting with municipal solid waste (Adamtey et al., 2009; O. Cofie et al., 2016; Nikiema et al., 2013). Faecal sludge treatment systems that encompass co-composting offer the avenue for returning nutrients and organic matter into soils therefore directly linking agriculture. These treatment systems are essential to incentivise safe FSM practices and resource recovery (Mrimi et al., 2020) and to make these resources available to agriculture in SSA allowing for rapid service expansion to underserved populations (Larsen et al., 2013). New processes and technologies that recover and promote the reuse of nutrients from both solid and liquid wastes while simultaneously inactivating pathogens are needed in today's society to close the loop on the nutrient cycle and to address future scarcity of non-renewable nutrients and fossil-based fertilisers (Vanotti et al., 2019).

1.4.1 Co-treatment of Faecal Sludge: End-to-end Nutrient Recovery and Pathogen Inactivation Efficiencies at Treatment Plant Level

Co-composting of manure and selected organic fractions of municipal solid waste is well known and understood as a way to manage and reduce volumes of these wastes. Co-composting of FS is however less well understood (Odey et al., 2017; Oudart et al., 2015) even though traditionally, FS have been used for crop fertilisation in many countries including Japan, China and Sweden (Mariwah & Drangert, 2011) over many centuries. The development of appropriate and sustainable technologies for efficient nutrient recovery has been identified as one of the main challenges in waste management within a circular economy (Bernal, 2017). The International Water Management Institute (IWMI) piloted a treatment system which comprises of different individual units and levels of FS treatment into one holistic treatment unit (one stop shop) to allow for nutrient and organic matter recovery from liquid and solid waste into a fertiliser. This holistic co-treatment consist of a primary treatment (FS dewatering on sand drying beds) coupled with secondary treatment (aerobic/open air co-composting with municipal solid waste) (Nikiema et al., 2013). IWMI had piloted and commercialised the co-treatment system by facilitating the setting up of a Public-Private Partnership (PPP) business between private SMEs and the Local Authority as a proof of concept in Ghana. What is novel about this treatment system is that it treats FS and FW from various sites into co-composts, there is storage of the co-composts and a local site for agricultural trials and farming using the co-compost as fertiliser. What is also significant about this process is because it is one stop shop, it allows for closer monitoring for reasonably adequate operation compared to the other FS treatment plants (FSTPs). Although FSTPs are prevalent in SSA, this novel system is coming up gradually into the mainstream. These co-treatment systems are currently being replicated in SSA in by IWMI and partners because it is highly applicable to the context of SSA by improving FSM and improving soil health. It is also expected that, this will be probably the leading treatment system for FS in SSA because it is low-cost.

While there is abundance of information on the efficiency of the individual primary (dewatering) and secondary treatment (composting) units in literature, there is however a dearth of information on the nutrient recovery efficiency and pathogen inactivation in the end-to-end FS treatment system (primary and secondary treatment combined). Nutrient flows in a FS treatment involve complex mechanisms comprising many biological and physico-chemical transformational processes such as mineralisation into leachate, volatilisation into the atmosphere, assimilation into living tissues of microorganisms, remineralisation in substrates, adsorption to surfaces, sedimentation etc. (Baby & Ahammed, 2022; Forbis-Stokes et al., 2020). Meanwhile, recent studies have shown that some sanitation interventions are not sufficient to mitigate possible routes of pathogen transmission (Freeman et al., 2017). Understanding the flow and changes of pathogen levels in a FS co-treatment system from process start to end can help provide new knowledge on the treatment process to safeguard public health and the environment and promote confidence in recycling.

To be able to assess the public health risks posed by faecal pathogens and to minimize the risk of environmental release of these pathogens, FS treatments plants need to be assessed for their pathogen reduction profile and efficiency. It is generally accepted that conventional wastewater treatment plants reduce the numbers of enteric pathogen (Anastasi et al., 2012). However, the extent to which this occurs can vary extensively depending on the treatment process (Anastasi et al., 2012). There is abundance of information available on behavior of pathogens in conventional wastewater treatment and the performance of wastewater treatment plants (Al-Gheethi et al., 2018; Anastasi et al., 2012; Pärnänen et al., 2019) but very little information exist on the behavior of pathogen in terms of their reduction profile in decentralised FS co-treatment systems in SSA.

1.5 Storage of Faecal Derived Fertilisers

Stockpiling or storage of compost is a common practice if field conditions are not suitable for immediate land application. In addition, they could also be kept in warehouses awaiting transportation and sale following production. Co-composted FS is referred to as faecal derived fertiliser (FDF) in the rest of this thesis and just like any other compost material, it has mouldy grain, fine, brown, and light characteristic material which could change with time especially during storage awaiting field application. During storage, compost continues to decompose and CH_4 and N_2O gases can be released (Hao, 2007). The major concern for compost material especially FDF is the possibility for re-contamination or recolonization of pathogens from either outside sources or subsequent regrowth of these inherent organisms to hazardous levels (Fane et al., 2021; Fane, 2016; Zaleski et al., 2005). As well as losses in nutrient characteristics (Sundermeiser, 2016). The waiting period (storage duration) is when FDF is produced, and it is yet to be applied especially where conditions do not allow for immediate field application. To contribute meaningfully to the ongoing debate regarding safety of using FDF, it is important to investigate how the fertiliser characteristic is affected by storage duration and conditions. It is obvious that, some composts can provide niches for pathogen survival and even regrowth, but the fate of pathogens during FS derived fertiliser storage is still not clear after going through the treatment processes.

Foodborne outbreaks linked to mostly animal waste-based fertilisers such as manure or FS have also revealed gaps in knowledge of the microbiological safety of such compost products and highlighted the need for implementing proper risk-reduction storage strategies for them (Chen et al., 2018). Therefore, if composting ensures complete deactivation of pathogens or reduction to levels below detection limits and yet, there are reported cases of contamination, then it means the storage duration and conditions could play a critical role to the quality of the stored compost. The duration of storage exposes matured compost to some changes in physico-chemical and pathogen characteristics. The survival of pathogens and the availability of nutrients and heavy metals during storage, depend on abiotic (temperature, pH, humidity) (Pawłowska et al., 2019) and biotic (composition and diversity of the microbial community) factors. In SSA, where FDF is being promoted, as an innovative solution to poor sanitation management and soil fertility, there remains a knowledge gap on its quality characteristics following storage.

1.6 Residual Effect of Faecal Derived Fertiliser Use on Vegetables

Over the years, FDF have been used to cultivate different types of crops with positive effects on the physico-chemical properties of soil, growth and yield of crops (Moya et al., 2019; Sommer et al., 2013). Earlier studies have looked at the use of FDF for vegetable production (Magwaza, et al., 2020; Nartey et al., 2017; Pradhan et al., 2019; Torgbo et al., 2018), cereal production (Adamtey et al., 2010; Pradhan et al., 2016) and oil palm (Ofosu Ansong, 2014). In many of these studies, the direct effect of one time application of FDF were assessed on growth and yield components of crops and soils properties after only one cultivation cycle (Magwaza, et al., 2020; Nartey et al., 2021; Torgbo et al., 2018). So far, only one publication had attempted to study the residual effect of FDF but limited it to after two cultivation cycles (Pradhan et al., 2019) . There is very little information and knowledge on the residual effect of applications of FDF on crops and mineralisation in soils over successive cultivations. Acquiring knowledge on the residual effect of FDF will give more depth and understanding of the effect of FDF on soils and crops especially leafy vegetables like lettuce.

Lettuce is one of the vegetables that is widely cultivated and consumed mostly raw. It is grown at global level on over a million of hectares every year and a production of more than 22 million tons (FAOSTAT, 2022). Lettuce is often studied for microbial safety following use of manure or excreta-based fertiliser for its cultivation. Intensive lettuce cultivation practices are major consumers of fertilisers and pesticides, which can lead to widespread contamination. Furthermore, these practices involve significant costs in terms of energy and the environment. It is therefore necessary to provide alternatives to improve the sustainability of such agricultural ecosystems without reducing productivity (Hernández et al., 2016). One way to ensure food safety and to provide evidence of safety of FDF use in crop production is to assess the yield and microbiological safety of vegetables cultivated on residual fertilization of FDF under field conditions.

1.7 Chapter conclusion, Thesis Aim and Objectives

1.7.1 Chapter 1 conclusion

Food security is an urgent global issue, but it is even more acute in Africa as millions of people across the continent still live in poverty with declined soil fertility resulting in reduced agricultural productivity. Relying on only mineral fertiliser sources cannot build soil health in Africa to accelerate agricultural productivity also due to the unrenewable nature of the mineral fertiliser sources. A way of improving agriculture sustainably is by linking it with sustainable waste management through recovery of valuable resources (water, nutrients, and organic matter) from organic fractions of municipal waste streams and sanitation systems such as faecal sludge (FS) thus promoting a circular economy (Keraita et al., 2008; Nikiema et al., 2014). The concepts around resource recovery and reuse as well as resource use efficiency are long established and known. However, the situation is more urgent now because of the convergence of several factors such as geopolitical risks and conflicts around natural resources, climate change urgency, competition in the markets for natural resources, the finite nature of some resources such as phosphate rocks (Cordell et al., 2012), and the growing scientific and technological know-how, makes now the right time (Lassaletta et al., 2014; Schröder et al., 2011) to encourage circular economy of resources.

Decentralised ways of co-treating FS and food waste (FW) are being piloted as a sustainable waste management tool to recycle useful nutrients and organic matter in a safe manner for smallholder farmers in SSA. IWMI and partners have commercialised a one stop shop system for FS treatment. What is novel about this treatment system is that it treats FS and FW from various sites into co-composts, there is storage of the co-composts and a local site for agricultural trials and farming using the co-compost as fertiliser. What is also significant about this process is because it is one stop shop, it allows for closer monitoring for reasonably adequate operation compared to the other FS treatment plants (FSTPs). However, there is a dearth of information on the end-to-end assessment of nutrient recovery efficiency and pathogen inactivation efficiency in the FS co-treatment system and the mechanisms of nutrient losses and pathogen inactivation. The co-treatment process leads to the production of faecal derived fertiliser (FDF) which will undergo storage. The survival of pathogens and the availability of nutrients and heavy metals during storage, depend on abiotic (temperature, pH, humidity) (Pawłowska et al., 2019) and biotic (composition and diversity of the microbial community) factors. In SSA, where FDF is being promoted, as an innovative solution to poor sanitation management and soil fertility, there remains a knowledge gap on its quality characteristics following storage. There are knowledge gaps in the residual effect of applications of FDF on crops and mineralisation in soils over successive cultivations. Acquiring knowledge on the residual effect of FDF will give more depth and understanding of the effect of FDF on soils and crops especially leafy vegetables like lettuce. The chapter concludes by highlighting knowledge gaps along the co-treatment of FS and FW value chain from production to use in Africa.

1.7.2 Thesis Aim

Therefore, the aim of this study is to generate new knowledge and understanding on the recovery of nutrients and *E. coli* inactivation during treatment and use of fertiliser produced from faecal sludge and solid waste.

As such it was hypothesised that through optimisation of key variables, the co-treatment of faecal sludge and solid waste has the potential to recover nutrients and organic matter at International Standard safety levels while remaining economically viable for smallholder farmers. As such faecal derived fertilisers could be considered an alternative to mineral fertilisers and a potential solution in the advancement of circular economy and food security. To test the hypothesis and deliver against the overall aim, the following specific objectives were set:

1.7.3 Specific objectives

1. To determine NPK fluxes, losses and recovery efficiencies in a faecal sludge and solid waste treatment system.
2. To assess the effect of faecal sludge and solid treatment system on *E. coli* inactivation.
3. To determine the effect of storage conditions on inherent *E. coli* loads and nutrient levels of faecal derived fertiliser in storage.
4. To determine the direct and residual effects of one-off application of faecal derived fertiliser on soil nutrient, nutrient uptake, and yield of lettuce.
5. To assess the costs and benefits of the direct and residual effect of faecal derived fertiliser on lettuce growth and yield at farm level
6. To recommend an approach to the assurance of the quality of product arising from faecal sludge and solid waste treatment processes.

A conceptual diagram of the transformation of solid and liquid waste into fertiliser research flow with the objectives is highlighted in Figure 1-2.

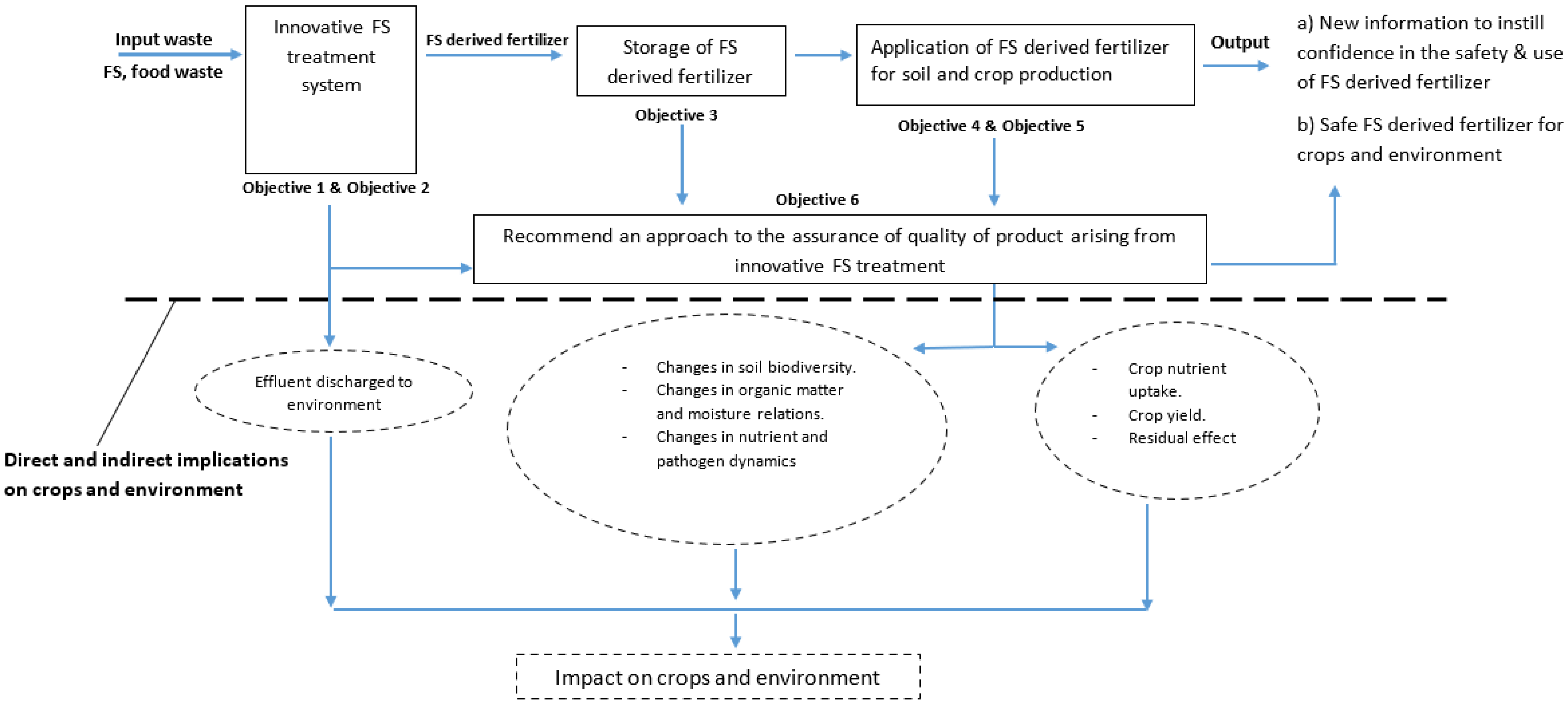

Figure 1-2. A conceptual diagram of research flow with the objectives

1.8 Thesis Plan

This thesis is presented as a series of chapters formatted as journal papers. All papers were written by the primary author, Eric Gbenatey Nartey and edited by Prof. Ruben Sakrabani, Prof. Sean Tyrrel and Dr Olufunke Cofie. All experimental work was designed, co-ordinated and completed by Eric Gbenatey Nartey in Ghana at the International Water Management Institute (IWMI) with contributions from MSc students and interns, University and National laboratories in Ghana. The initial field set up and treatment efficiency assessment described in Chapters 3, 4 & 6 were carried out at the JVL-YKMA Recycling Plant and was assisted by IWMI's research interns and staff of the JVL-YKMA Recycling Plant. In addition, assistance in sample collection and analysis during all the experimental regimes described in the chapters were aided by Christopher Ameho, Edna Dapaah, Jestyl Arku and Selorm Ayeduvor of IWMI, Dr Ivy Mensah and Christian Bonsu of Noguchi Memorial Institute for Medical Research.

A literature review was completed in Chapter 2 on the quantitative nutrient recovery of decentralised FS treatment systems in SSA, the dynamics of pathogen survival in the decentralised FS treatment system as well as the effect of storage conditions on recovered nutrient and organic matter in the form of compost/faecal derived fertiliser (FDF) in storage. The outcome of the review was addressed in the succeeding Chapters. For example, one of the outcomes of the review indicated that there is very little information on the end-to-end assessment of decentralised FS and FW treatment system in Africa for nutrient recovery and pathogen inactivation efficiencies. Chapter 3 therefore espoused on the end-to-end assessment of the FS and FW treatment system for NPK fluxes, determining losses and recovery efficiencies in treatment system (Chapter 3, Paper 1 – in preparation: Nartey, E.G., Sakrabani, R., Tyrrel, S., Dapaah, E. and Cofie, O. Nutrient dynamics and recovery efficiencies in a faecal sludge and food waste treatment system, *Water Science and Technology*). Findings from the chapter revealed that, the treatment efficiency of the various components of the system is variable and one of the major factors is the variable characteristics of the FS. Significant losses (50-70%) of total N was observed around the dewatering stages.

While Chapter 4 espoused on the effect of the end-to-end treatment system on *E. coli* inactivation (Chapter 4, Paper 2 – in preparation: Nartey, E.G., Sakrabani, R., Tyrrel, S., Arku, J., Mensah, I. and Cofie, O. Assessing the effect of the decentralised faecal sludge treatment process on *E. coli*, *Water Science and Technology*). Findings from this chapter revealed that the *E. coli* levels observed in Cycles 2 and 3 were below the maximum limits set by the Ghana EPA and EU. Mass balance estimations around the FS dewatering reveal that between 0.1 – 0.3% of all incoming *E. coli* is retained in the DFS per cycle. A range of 25 – 55% of the *E. coli* is retained in the percolate moving on to the effluent polishing in the facultative and maturation ponds.

Chapter 5 is divided into two parts (a and b), following the assessments carried out in Chapters 3 and 4. In Chapter 5a, faecal derived fertilisers (FDF) were prepared in batches (Chapter 5a, Paper 3 – abstract presented to 21st AFWA & FSM7 2023 conference, Abidjan: Nartey, E.G., Sakrabani, R., Tyrrel, S. and Cofie, O. Assessing consistency in the co-composting of faecal sludge and food waste in Ghana). The study assessed the extent of consistency in compost characteristics between and within batches. Between batch consistency was assessed in three successive batches (1, 2 and 3) of co-composted FS and food waste (FW). Within batch consistency was assessed in each of the three batches by dividing the batch into four separate replicate piles. While variations between batches were only observed for EC and nutrient parameters, variations were evident for several measured parameters within batches.

It is recommended that a threshold value be created for determining what is an acceptable level of variation in FS co-composting.

In Chapter 5b, the FDF from one of the batches was utilised for the effect of storage conditions on inherent *E. coli* loads and nutrient levels of faecal derived fertiliser in storage (Chapter 5b, Paper 4 – *manuscript prepared*: Nartey, E.G., Sakrabani, R., Tyrrel, S. and Cofie, O. The effect of storage duration and temperature on pathogen load, heavy metals, and nutrient levels of faecal derived fertiliser, *Nutrient cycling in agroecosystems*). Findings show no *E. coli* presence in FDF at the end of the storage period indicating no indigenous regrowth. However, total coliform counts were present. There were significantly higher counts of total coliform in non-enriched FDF than in enriched FDF occurring at lower storage temperatures (25°C and 5°C) though there was a general decline in counts with increasing storage duration. Nitrogen was significantly higher in the enriched FDF with 2.4% than in the non-enriched FDF with 1.1% but storage temperature and duration did not have any effect on nitrogen. The available forms of nitrogen (NH_4-N and NO_3-N) were significant for the main effects of fertiliser, temperature, duration, and their interaction effects. An inverse relationship was observed between NH_4-N and NO_3-N concentrations in enriched FDF under lower temperatures and increasing duration of storage.

Chapter 6 assessed the direct and residual effect of faecal derived fertiliser on soil nutrient, nutrient uptake, and yield of lettuce for four planting cycles (Chapter 6, Paper 5 – *manuscript prepared*: Nartey, E.G., Sakrabani, R., Tyrrel, S., Ameho, C. and Cofie, O. The direct and residual effect of one-off application of faecal derived fertiliser on soil nutrient, nutrient uptake, and yield of lettuce for four planting cycles, *Agronomy for Sustainable Development*). Findings show, for the direct effect (first cycle), the highest lettuce yield of 27.9 t/ha was recorded in the co-compost fertilized plots followed by 11.1 t/ha for mineral fertilized plots. There was residual effect of faecal derived fertiliser which could improve productivity and income by as much as 344.4% by the second cycle even though benefits could be gained up to the fourth cycle. *E. coli* was absent on lettuce leaves after successive cultivations. Therefore, farmers who could not afford to buy and apply mineral fertiliser every season could rely on FDFs.

Based on Chapter 6, the economic assessment of the direct and residual effect of faecal derived fertiliser on lettuce growth and yield was carried out in Chapter 7 (Chapter 7, Paper 6 – in preparation: Nartey, E.G., Ayeduvor, S, Sakrabani, R., Tyrrel, S., Gebrezgabher, S. and Cofie, O. Economic Assessment of Faecal Derived Fertilisers on Lettuce Yield Following Successive Cultivations, *XX*). The findings show that, profit from lettuce cultivation increased from USD3,233.00/ha for unamended control to USD8,342.17/ha for mineral, USD5,186.34/ha for FDF enriched co-compost, and USD22,272.24/ha for FDF Co-compost in the direct effect (Cycle 1). Profit levels decreased for all fertiliser treatments for the residual effects except the FDF enriched co-compost plots which increased from USD5,186.34/ha to USD8,670.75/ha during the cycle 2. The FDF Co-compost and enriched had a benefit cost ratio (BCR) of 14.5 and 8.4 and return of investment (ROI) of 385.7 and 281.1, respectively. For control and mineral fertiliser, a BCR of 3.2 and 4.1 and ROI of 109.5% and 130.2%, respectively was obtained. An ROI of 385.7 indicates that for every dollar that is invested in the cultivation of lettuce using a co-compost, about USD 385.7 is generated.

Finally, Chapter 8 presents the Final discussion. The main points of the overall conclusion and the implications of the findings are discussed in the sub-sections. The chapter ends with a conclusion that highlight the contribution of this research to knowledge.

Table 1-1 summarizes the thesis plan and the status of paper submissions.

Table 1-1. Thesis plan and status of paper submissions

Chapter	Paper	Objective	Title	Journal/Conference	Status
3	1	1	Nutrient dynamics and recovery efficiencies in a faecal sludge and food waste treatment system	Water Science and Technology	In preparation
4	2	2	Assessing the effect of the decentralised faecal sludge treatment process on *E. coli*	Water Science and Technology	In preparation
5	3	3	Assessing consistency in the co-composting of faecal sludge and food waste in Ghana	21st AFWA & FSM7 2023 conference	Abstract submitted
5	4	3	The effect of storage duration and temperature on pathogen load, heavy metals, and nutrient levels of faecal derived fertiliser	Nutrient cycling in agroecosystems	Manuscript prepared
6	5	4	The direct and residual effect of one-off application of faecal derived fertiliser on soil nutrient, nutrient uptake, and yield of lettuce for four planting cycles	Agronomy for Sustainable Development	Manuscript prepared
7	6	5	Economic Assessment of Faecal Derived Fertilisers on Lettuce Yield Following Successive Cultivations	XX	In preparation
8	-	1,2,3,4,5	Implications of the work, conclusions, and future work	-	-

Table 1-2 Statement of authors contribution

Chapter	Author Contributions
3	<u>*E. G. Nartey*</u>: *Developed the concept of study and was the principal investigator and writer. Designed and implemented the experimental design, sampling framework for assessing the NPK flux in the faecal sludge (FS) and food waste (FW) treatment system. Undertook sample collection in the field, data and statistical analysis and the write up of chapter 3.* <u>*Professor R. Sakrabani*</u>*: Was the principal supervisor supporting development of structure and evaluation of final chapter write up. Supporting research concept development, guidance on experimental design, data analysis and evaluation of final chapter 3 write up.* <u>*Professor S. Tyrrel:*</u> *Was co-supervisor supporting with development of structure and evaluation of final chapter. Supporting research concept development, guidance on experimental design, data analysis and evaluation final chapter write up.*

	<u>E. Dapaah</u>: was a 6-month student intern placed with the International Water management Institute (IWMI) supporting with field work (sampling and data collection) and statistical analysis. *<u>Dr. O. Cofie</u>: Was the Industrial supervisor at IWMI. Provided supervisory support during the experimental design phase and research implementation.*
4	*<u>E. G. Nartey</u>: Developed the concept of study and was the principal investigator and writer. Designed and implemented the experimental design, sampling framework for assessing the E. coli inactivation in FS and FW treatment system. Undertook sample collection in the field as well as laboratory work in two different microbiological labs in Accra, Ghana. He organized data collection and statistical analysis and wrote chapter 4.* *<u>Professor R. Sakrabani</u>: Was the principal supervisor supporting development of structure and evaluation of final chapter write up. Supporting research concept development, guidance on experimental design, data analysis and evaluation of final chapter 4 write up.* *<u>Professor S. Tyrrel:</u> Was co-supervisor supporting with development of structure and evaluation of final chapter. Supporting research concept development, guidance on experimental design, data analysis and evaluation of final chapter 4 write up.* *<u>E. Dapaah</u>: was a 6-month student intern placed with the International Water management Institute (IWMI) supporting with field work (sampling and data collection) and statistical analysis.* *<u>Dr. O. Cofie</u>: Was the Industrial supervisor at IWMI. Provided supervisory support during the experimental design phase and research implementation.*
5a	*<u>E. G. Nartey</u>: Was the principal investigator and writer of chapter 5a. Designed and implemented the experimental design on consistency assessment of FS and FW composting system, designing sampling framework and sample collection, laboratory work, data collection and statistical analysis as well as chapter write up. Presented an oral presentation of this work at 21st AfWA & FSM7 Conference in Abidjan in 2023.* *<u>Professor R. Sakrabani</u>: Was the principal supervisor supporting development of structure and evaluation of final chapter write up. Supporting research concept development, guidance on experimental design, data analysis and evaluation of final chapter 5a write up.* *<u>Professor S. Tyrrel:</u> Was co-supervisor supporting with development of structure and evaluation of final chapter. Supporting research concept development, guidance on experimental design, data analysis and evaluation of final chapter 5a write up.* *<u>Dr. O. Cofie</u>: Was the Industrial supervisor at IWMI. Provided supervisory support during the experimental design phase and research implementation.*
5b	*<u>E. G. Nartey</u>: Developed the concept of study and was the principal investigator and writer. Designed and implemented the experimental design and sampling framework for assessing storage environment on characteristics of stored faecal derived fertilisers. Undertook sample collection in the field as well as laboratory work in two different microbiological labs in Accra, Ghana. Organized data collection and statistical analysis and wrote chapter 5b.* *<u>Professor R. Sakrabani</u>: Was the principal supervisor supporting development of structure and evaluation of final chapter write up. Supporting research concept development, guidance on experimental design, data analysis and evaluation of final chapter 5b write up.* *<u>Professor S. Tyrrel:</u> Was co-supervisor supporting with development of structure and evaluation of final chapter. Supporting research concept development, guidance on experimental design, data analysis and evaluation of final chapter 5b write up.* *<u>Dr. O. Cofie</u>: Was the Industrial supervisor at IWMI. Provided supervisory support during the experimental design phase and research implementation.*

6	*<u>E. G. Nartey</u>: Developed the concept of study and was the principal investigator and writer. Designed and implemented the field experimental design and sampling framework for assessing the residual effect of faecal derived fertilisers on lettuce. Undertook field sampling as well as laboratory work on pathogens. Organized data collection and statistical analysis and wrote chapter 6.* *<u>Professor R. Sakrabani</u>: Was the principal supervisor supporting development of structure and evaluation of final chapter write up. Supporting research concept development, guidance on experimental design, data analysis and evaluation of final chapter 6 write up.* *<u>Professor S. Tyrrel:</u> Was co-supervisor supporting with development of structure and evaluation of final chapter. Supporting research concept development, guidance on experimental design, data analysis and evaluation of final chapter 6 write up.* *<u>C. Ameho</u>: was a 6-month placement intern with IWMI supporting with the crop and fertiliser field trials as well as supporting with field sampling and data collection and statistical analysis.* *<u>Dr. O. Cofie</u>: Was the Industrial supervisor at IWMI. Provided supervisory support during the experimental design phase and research implementation.*
7	*<u>E. G. Nartey</u>: Developed the concept of the study. Completed the field experiment in chapter 6 and provided data for the analysis in this chapter 7. Wrote chapter 7.* *<u>Professor R. Sakrabani</u>: Was the principal supervisor supporting development of structure and evaluation of final chapter write up. Supporting research concept development, guidance on experimental design, data analysis and evaluation of final chapter 7 write up.* *<u>Professor S. Tyrrel:</u> Was co-supervisor supporting with development of structure and evaluation of final chapter. Supporting research concept development, guidance on experimental design, data analysis and evaluation of final chapter 7 write up.* *<u>S. Ayeduvor</u>: was a 5-month placement economics intern with IWMI supporting with economic analysis of applied faecal derived fertiliser on lettuce yield over successive cultivations.* *<u>Dr. S. Gebrezgabher</u>: Researcher in Economics at IWMI. Provided supervisory support to economic intern for economic analysis of faecal derived fertiliser application over successive cultivation.* *<u>Dr. O. Cofie</u>: Was the Industrial supervisor at IWMI. Provided supervisory support during the experimental design phase and research implementation.*

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Chapter 2 Literature Review

2.1 Introduction

Chapter 2 reviews the relationships and linkages between resource recovery potential of wastes generated in sub-Saharan Africa (SSA) especially faecal sludge (FS) and food waste (FW) to rising food security issues, fertiliser demand and supply issues, and climate change issues at the global scale and in the global south. The chapter comprises of review of literature on quantitative nutrient recovery of decentralised FS treatment systems in SSA. The nutrient and organic matter recovered from decentralised FS treatment systems must be safe and sanitized, hence a review of literature on the dynamics of pathogen survival of the decentralised FS treatment system was also carried out. The chapter also reviewed literature on the effect of storage conditions on recovered nutrient and organic matter in the form of compost/faecal derived fertiliser (FDF) in storage over a period of time before finally reviewing literature on the effect of FDF application on crops and soil environment.

2.2 Food Demand in sub-Saharan Africa

Food insecurity is a threat to all regions of the world though the concentration and distribution differs in different regions (Figure 2-1). A person is food insecure when they lack regular access to enough safe and nutritious food for normal growth and development and an active and healthy life. This may be due to unavailability of food and/or lack of resources to obtain food (FAO, 2021). In sub-Saharan Africa (SSA), almost one in four people were estimated to be undernourished in 2017, representing about one third of the 821 million people suffering from chronic hunger globally (FAO et al., 2018). In addition to a high prevalence of chronic hunger in SSA, many more people suffer from micronutrient deficiencies (Harika et al., 2017; Joy et al., 2014; Kumssa et al., 2015).

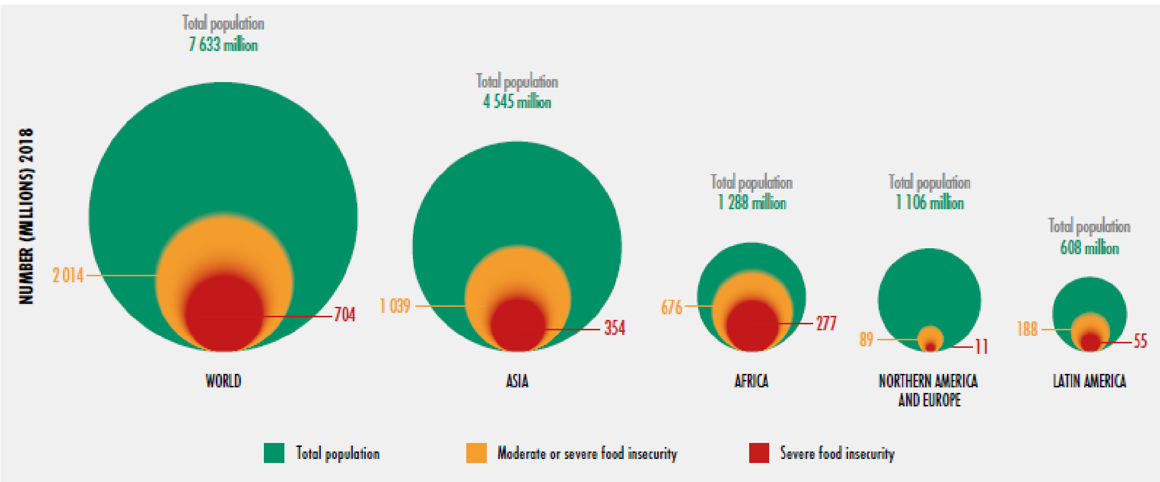

Figure 2-1. The Concentration and Distribution of Food Insecurity by Severity across the Regions of the World. *Source: FAO*

The population in SSA is expected to double from the current 1.2 to about 2.4 billion by 2050 even though at current levels, the region still experiences food shortages. The continent is still not able to feed itself, and there is an urgent need to improve agricultural productivity (AfDB, 2020). Crop yields are low as a result of low use of agricultural inputs and a dependency on rainfed agriculture (Harrison, 2020). Although fertilizer is one of the most needed inputs in agriculture, its use remains below the 2006 Abuja Declaration objective of at least 50 kg of nutrients of fertiliser use per hectare of arable land (AfDB, 2020). Meanwhile the majority of the food in SSA is produced by smallholder farmers

(Herrero et al., 2017) and they are the most vulnerable to food insecurity and poverty (Fanzo, 2018; Sibhatu & Qaim, 2017). Hence, smallholder farmers are a crucial entry point for agricultural orientated interventions to improve food and nutrition security.

2.2.1 Fertiliser Demand and Supplies in sub-Saharan Africa

Global demand for mineral fertilisers is continuously on the rise (Vanotti et al., 2019) as global agriculture is highly dependent on inputs of mineral fertilisers (Cordell *et al.*, 2009). However only 6% of Africa's cultivated land is irrigated and average fertiliser consumption in SSA is estimated at 17kg of nutrients per hectare of cropland (Agra 2018), which is nearly double the baseline of 8–9 kg/ha of 2006, but less compared to a world average fertiliser consumption of 135kg/Ha (Harrison, 2020). Fertiliser consumption among smallholder farmers, who make up most farmers in the region and farm most of the land, has grown in the past decade but it is still far below what is needed as farmers face numerous challenges that limit their effective fertiliser demand. One of the challenges faced by farmers include limited access to finance, which affects their demand for mineral fertiliser and other inputs. Markets in the region are fragmented and intra-regional trade is weak. Agro-dealers and other rural entrepreneurs need to trigger farmers' demand for new fertiliser products, especially blends. Willingness to invest in fertilisers will be driven by improved knowledge and information, better availability of fertilisers, and better market opportunities for farm output (Harrison, 2020).

Most soils in Africa are highly weathered and nutrient poor. Soil fertility across much of Africa is low. Over 40% of African soils face nutrient depletion, partly because of a failure to apply sufficient levels of fertilisers. While nitrogen (N) is the most limiting nutrient for crop production, many agricultural soils in SSA are deficient in phosphorus (P), potassium (K), sulphur (S), micronutrients, and organic matter as well which are critical for a balanced nutrient application. Therefore, Africa will be a key driver of future fertiliser growth, with SSA consumption expected to increase 5% annually to reach 5.9 million tonnes nutrients in 2022, boosted by Nigeria and Ethiopia, and rising to represent 3% of the world global consumption. Current growth is mainly driven by higher application rates rather than increased cropland. The average fertiliser application rate in SSA is expected to increase to 22kg/ha from the current low level of around 16kg/ha by 2022, still far below the continent-wide recommendation of about 100kg/ha (Harrison, 2019). Excessive applications of mineral fertilisers also intensify losses of N and P to soils, water bodies and the environment (Desmidt et al., 2015).

The future availability of mineral fertilisers, especially of P, in the face of high mineral fertiliser demand in agriculture is questionable as rock phosphates are a finite resource (Cordell et al., 2012). Furthermore, rock phosphates mined for the production of mineral P fertilisers are located in a small number of countries, with Morocco and China controlling more than half of the reserves currently known, constituting a geopolitical risk to global P availability (Cordell & White, 2015). The likely future resource scarcity as well as environmental and health concerns associated with excess nutrients in the environment have triggered research and policy developments aimed at increasing circularity for N and P, especially in the past decade (Cordell *et al.,* 2009; Lassaletta *et al.*, 2014). In addition to N and P, however, also potassium (K) is an essential plant macronutrient, required in comparable quantities as N (Bellarby et al., 2018). Owing to the rising global demand, the supply from diminishing K reserves (K - rich rocks) is insecure (Santos et al., 2017). Unlike N and P, K has not been linked to deleterious environmental effects. However, some areas of the world are highly dependent on the input of K fertilisers (Santos et al., 2017) and for those areas, K recycling offers clear advantages as it diminishes the dependence on external resources.

2.3 Waste Generation and Reuse Potential in sub-Saharan Africa

The countries in SSA have the highest population growth rates in the world yet agricultural soils are deficient in plant nutrients impairing their ability to feed its growing population. There is potential to recover nutrients and organic matter from organic waste and sanitation management related interventions that are being developed and implemented in cities in SSA. For decades, common agricultural wastes have been reused to maintain crop yield and soil health. Over time, it was realized that cultivation of staple crops using agricultural wastes (crop residues, animal manure and composted product derived from such inputs) is getting beyond the reach of African smallholder farmers, due to competing demands on available crop residues, and lack of immediate benefits (Nigussie et al., 2015; Vanlauwe *et al.*, 2015). Hence, it would be more prospective to find locally untapped waste resources and integrate them into smallholder farming. The application of organic amendments have been recommended to maintain one or more of the following benefits: improve crop yield through the supply of plant nutrients, enhance the efficiency of mineral fertiliser and improve soil health (Baudron et al., 2014; Lal, 2015).

The source of organic amendments used in agriculture include raw and processed agricultural wastes (crop residues, animal manure, poultry litter and composted products) and municipal waste (faecal/ sewage sludge and wastewater) (Baudron et al., 2014; Drechsel & Dongus, 2010). Human excreta/ faecal sludge (FS) and food waste (FW) was usually considered as waste to be disposed of; but the ultimate disposal of FS and other municipal organic waste into water bodies and open dumps only adds to the already high environmental burden (Spångberg *et al.,* 2014; Simha and Ganesapillai, 2017). It is therefore an important strategy to recycle FS from sanitation interventions and organic wastes such as FW to recover the nutrients inherent in them for use in agriculture as fertiliser or soil amendment.

2.3.1 Faecal sludge

Faecal sludge refers to the raw, slurry, or partially digested semisolids resulting from the collection of a combination of black water and human faeces with or without the combination of grey water (Strande et al., 2014; Zhou et al., 2018). It is either contained in an onsite sanitation technology or it is connected to or transported in a sewer. The current state of sanitation in urban areas of low- and middle-income countries is 2.8 billion people served by onsite sanitation, with the majority of excreta not safely managed. For example, only 37% is safely managed in 12 reported cities (Peal et al., 2014). In SSA, 65–100% of sanitation access in the urban centres is provided by onsite technology (Odey et al., 2017). Moreover, the recognition that vital resources from FS, such as organic matter and nutrients, can be used for soil amendment in agriculture has led to the development of resource-oriented sanitation approaches (Jayawardhana et al., 2016; Kah et al., 2016; Lau et al., 2017). Faecal sludge can be managed more efficiently through a resource-oriented sanitation approach that can recover organic matter and nutrients available in FS to be utilized in agriculture and close the loop in the nutrient cycle (Schönning et al., 2002). Such sanitation systems could provide efficient resources, sustainable sanitation, and an economically sound alternative that protects both humans and the environment.

Human faeces consist of about 75% water by weight and 25% solid material, mainly organic matter (Rose et al., 2015). Carbon (C) is a major constituent of the dried solids as approximately half of organic matter generally is C (Vassilev et al., 2010) and this is also true for faeces (Rose et al., 2015). N, P, and K make up 5–7%, 3–5.4%, and 1–2.5% of the dried solids respectively (Rose et al., 2015). Both urine and faeces also contain a range of micronutrients such as magnesium (Mg) and selenium (Se). The amount of excreted nutrients depends on dietary intake, while the digestibility of the diet determines the partitioning of nutrients between urine (digested) and faeces (undigested) (Jonsson et al., 2004). In addition, by the time the sludge gets to a treatment plant, the nutrient contents will depend on the storage time (age of sludge) and dilution with other materials. Generally, urine contains the majority of N and about half of P and K contained in human excreta, while faeces are rich in P and K and contain the majority of C (Heinonen-Tanski & van Wijk-Sijbesma, 2005). Aside nutrients and organic matter, human faeces also contain pathogens such as bacteria, helminths, virus, and protozoa. *E. coli*, *Shigella, Salmonella* and *Vibrio spp*. have been the predominant pathogens linked to waterborne outbreaks in Africa (Bessong et al., 2009; Olaniran et al., 2011b), whereas certain clonal strains of these pathogens have been reported to survive conventional wastewater treatment processes (Adefisoye & Okoh, 2016; Anastasi et al., 2012; Cañigral et al., 2010). Aerobic composting can stabilize the organic matter in the human faeces as well as minimize the adverse effects of indigenous pathogens (Duan, Awasthi, Liu, Verma, et al., 2019).

2.3.2 Food waste

Food waste are wastes from food market, homes, institutions and the hospitality industry principally constituted by vegetable waste, fruit waste and foodstuffs waste usually containing high levels of organic matter (OM), moisture and nutrients (Sudharsan Varma & Kalamdhad, 2015). In a previous study, Jara-Samaniego et al. (2015) reported average values of 77.3% OM, and 2.5 – 0.7 – 3% of N – P_2O_5 – K_2O, respectively, nutrient values in food market wastes from the Chimborazo Region in Ecuador. The increased food waste generation is a global problem. The food waste generation has several facets, which is understood by its quantity and nature of food waste generated across all phases of the food production and consumption cycle. Majority of the stakeholders are concerned about the impact of food waste, which has on generation of methane and carbon dioxide, which are greenhouse gases (Jayathilakan et al., 2012). The major cost of municipal waste management ranges from 75% to 80% of a municipal waste budget and additional 30% cost for landfilling. During land filling the greenhouse gases are generated and energy value along with soil nutrient value is lost during landfilling. Segregation of food waste at source has the potential to decrease greenhouse gas emission and there will be no adverse impact on public health (Arvanitoyannis, 2008).

Presently FW is mostly disposed of at landfills along with other municipal solid wastes. Earlier data reveals that in 2012, nearly 9,278 tonnes of municipal solid waste were disposed at landfills each day, where about 3,337 tonnes is FW only (FAO, 2011a). Nearly 809 tonnes of FW were generated from restaurants, hotels, wet markets, food production and processing industries on the daily basis. However, FW and their definition are greatly varied from cities and countries to other cities and countries. Food wastes in the European Union are defined as ‘‘raw or cooked of any food substances that are discarded or intended or required to be discarded”. On other hand, the U.S. Environmental Protection Agency (EPA) defines FW as ‘‘Un-eaten foods and food preparation wastes from residences and commercial establishments including restaurants, grocery stores, and produce stands, institutional cafeterias, and kitchens, as well as industrial sources such as employee lunchrooms.” Furthermore, ‘‘Food loss” and ‘‘Food waste”, in the United Nations, are recognized differently. The term ‘‘Food losses” refers to the decrease in food quality and/or quantity. On the other hand, the term ‘‘food waste” refers to the food losses due to retailers’ and /or consumers’ behaviour (Russ & Meyer-Pittroff, 2004). In this review we refer to FW to include the uncooked raw materials, wasted foodstuffs, and the edible materials from groceries or the wet market.

2.4 Quantitative Nutrient Recovery Potential in Decentralised Faecal Sludge Treatment System

Nutrient from treatment plants pose significant challenges to the environment and public health. One major objective of waste treatment is to reduce the concentration of nutrients, especially nitrogen (N), phosphorus (P) and trace elements in effluents going into the receiving environment. Excessive levels of nutrients and trace elements discharged causes pollution of surface and groundwater bodies and could lead to bioaccumulation in living tissues. While cities are sinks for nutrient – rich food products, they also produce waste such as FS and food waste (FW) which makes them centres of resources and also potential locations for resource recovery (Keraita et al., 2015). However, most of these nutrients are lost, mostly in Global South due to poor sanitation and solid waste mismanagement resulting in widespread pollution of waterways, intensified by unplanned urban expansion (Currie et al., 2015). Water pollution by N and P has been a major driver of efforts towards more sustainable nutrient recovery (Cordell et al., 2011). Options that allow for the recovery of nutrients, water, and organic matter from FS such as decentralised FS treatment system are desirable, especially where such an option is low-cost, efficient, and appropriate to the local socio-economic and environmental context.

2.4.1 Nutrients of Importance in Faecal Sludge Treatment

Growing concern about future fertiliser availability has re-emphasized the need for better nutrient management, including comprehensive recycling of nutrients contained in FS to agriculture (Dawson & Hilton, 2011; Elser & Bennett, 2011; J. McConville et al., 2015). The trend towards nutrient extraction from FS coincides with a focus on the macronutrients (N, P, and K). There is no single recovery pathway that captures all nutrients and carbon in FS in a single product free of contamination. Harder *et al.* (2019) in a review on recycling nutrients contained in FS to agriculture suggested that, rather than focusing on a specific recovery pathway or product and on a limited set of nutrients, there is scope for exploring how to maximize nutrient recovery by combining individual pathways and products and including a broader range of nutrients. Both urine and faeces contain a range of micronutrients such as magnesium (Mg) and selenium (Se). Not many studies have focused on these micronutrient (calcium (Ca), magnesium (Mg), sodium (Na), iron (Fe), copper (Cu), manganese (Mn), zinc (Zn), boron (B), and molybdenum (Mo) recoveries from FS. These micronutrients are important in complimenting aspects of plant nutrition and soil health. For example, the presence of Ca is essential to counteract the effects of alkali salts and organic acids (Montejo et al., 2015) in soils.

While major nutrients such as N, P, K are often studied due to their environmental and economic importance, there is also to the need to characterise and quantify the micronutrients in resource recovery schemes to extend the beneficial uses of those nutrients from waste especially from faeces and urine (Harder et al., 2019). There is a clear divide between recovery pathways that target the recovery of some of the macronutrients and those that more broadly target a wider selection of nutrients, and possibly also organic matter (Harder et al., 2019). The call for further development of technologies that recover N and K in addition to P (Mehta et al., 2015) is a step in the right direction. But considering soil nutrient stripping (Jones et al., 2013) and soil carbon losses (Amundson et al., 2015), It is believed the scope of nutrient recovery should be even broader and also include micronutrients and organic matter. The value and importance of organic matter recovery is also quite crucial (Mnthambala, 2021). This will ultimately require a shift away from thinking in terms of individual recovery pathways, towards thinking in terms of sensible combinations of recovery pathways that maximize recovery of nutrients and organic matter while minimizing risks associated with contaminants (Harder et al., 2019).

There is a relatively little research that had been conducted on an end-to-end assessment of decentralised FS treatment system as a whole system unit with regards to nutrient recovery, as non-sewered sanitation has only recently been acknowledged as a long-term sustainable solution (Strande et al., 2014, 2018; USEPA, 2005). On the contrary, centralized treatment processes such as activated sludge treatment have been researched for over one hundred years (Stensel & Makinia, 2014). The demand for decentralised infrastructure to manage FS is increasing, and reliable methods are being developed and validated to estimate total accumulated quantities and qualities of FS to support management and treatment technology solutions (Englund et al., 2020; Strande et al., 2018). However, no information exists on estimation of total nutrient recoveries and the purported recovery efficiencies required to make nutrients from FS available for crop production and to minimize pollution in urban catchment areas. A few of the studies that considered nutrients, considered it from the perspective of meeting minimum discharge guidelines for NH_4, NO_3 and PO_4 in effluents for different countries (Abagale et al., 2020; Levira et al., 2023; Mofokeng et al., 2022). Studies on end-to-end assessment of nutrient recovery in the decentralised FS treatment system from start (when waste enters) to finish (treated products leave) is limited in the global south.

FS treatment technologies usually comprise primary treatment for solid - liquid separation and secondary treatment, for final polishing. Treatment at every primary level result in reduction of sludge volume which in turn minimises the storage requirement as well as transportation costs. After primary treatment, three types of end products is produced, i.e., screenings, treated sludge and liquid effluents. The liquid effluent generated from the primary treatment must be treated further to meet the requirements for water reuse or discharge into the environment. Low-cost technologies such as waste stabilisation ponds or wetlands could be used for the liquid treatment (S. Singh et al., 2017). Earlier studies on decentralised FS treatment systems only focused on some components of holistic FS treatment system such as primary treatment alone (e.g., FS dewatering on unplanted beds) or secondary treatment alone (e.g., co-composting, or anaerobic digestion). The studies on primary treatment of FS were focussed on dewatering efficiency of different FS types using different drying/dewatering methods (Getahun et al., 2020; Gold et al., 2018; Stefanakis & Tsihrintzis, 2011; Ward et al., 2019). Other studies focussed on pollutant removal efficiencies of FS dewatering systems such as constructed wetlands/vertical – flow constructed wetlands (Kengne et al., 2009, 2011; Koottatep et al., 2006).

While on secondary treatment, several studies on FS composting have been conducted and published to date (Al-Muyeed et al., 2017; Berendes et al., 2015; O. Cofie et al., 2009; Hashemi et al., 2019; Koné et al., 2007; Mengistu et al., 2018; Mulec et al., 2016; Nakasaki et al., 2011; Nartey et al., 2017; Oarga-Mulec et al., 2019; Thomas et al., 2018). Many of these studies have been on the optimization of FS co-composting with various organic wastes (Al-Muyeed et al., 2017; Berendes et al., 2015; O. Cofie et al., 2009; Hashemi et al., 2019; Koné et al., 2007; Mengistu et al., 2018; Nakasaki et al., 2011; Nartey et al., 2017; Thomas et al., 2018) and the effectiveness on FS sanitation during open air composting (Al-Muyeed et al., 2017; Fidjeland et al., 2013; Manga et al., 2021; Mengistu et al., 2018; Nartey et al., 2017). The end-to-end assessment of the decentralised FS treatment system regarding nutrient recovery has not been thoroughly assessed especially in sub-Saharan Africa. Very limited information is available on the nutrient fluxes and nutrient recovery efficiency after FS dewatering using sand drying beds as a primary treatment step to aerobic (open air) composting treatment.

2.4.2 Nutrient Recovery Efficiency in Decentralised FS Treatment

The nutrient loop involves agricultural production, processing, consumption, and collection/treatment of waste (including wastewater, FS etc.). Nutrients, when recovered from wastewater or FS can return as valuable organic and mineral compounds to agriculture, closing the loop (McConville et al., 2017). It is estimated that, globally, the total phosphorus content excreted by humans (just considering available phosphorus from faeces and urine) could meet 22% of the P demand (Mihelcic et al., 2011). In conventional domestic wastewater, over 80% of N and 50% of both K and P come from urine (Beler-Baykal et al., 2011). While traditionally, the objective of wastewater and FS treatment systems have generally been the removal of carbonaceous organic matter, total suspended solids, chemical pollutants, excess nutrients and pathogens before disposing the effluents to the environment (Tchobanoglous et al., 2015; Verstraete et al., 2009), the recycling of resources such as nutrients through recovery processes is an emerging objective due to environmental and economic motivations such as reducing pollution and generating revenue for the wastewater treatment works (Mehta et al., 2015; Rao et al., 2017).

Estimates by Qadir et al., (2020) suggest that, currently 380 billion m^3 of wastewater are produced annually across the world. This is expected to increase by 24% by 2030 and 51% by 2050 over the current level. Among major nutrients, 16.6 Tg (Tg = million metric ton) of N are embedded in wastewater produced worldwide annually; P stands at 3.0 Tg and K at 6.3 Tg. The full nutrient recovery from wastewater would offset 13.4% of the global demand for these nutrients in agriculture. Several previous studies have conducted reviews to provide information about different well established and new/emerging technologies which could be used to recover nutrients from wastes and bring zero waste concepts in practical life (Ahmed et al., 2019; Chrispim et al., 2020). A better understanding of the recovery efficiency and actual quantification of nutrients recovered for the various new/emerging treatment technologies is necessary. Yet the dearth of information on recovery efficiencies of decentralised FS treatment in general, and quantifiable nutrients recovered from FS treatment in particular, limit opportunities to fully understand and practicalise the nutrient loop.

2.5 Dynamics of Faecal Pathogen Survival in Decentralised Faecal Sludge Treatment System

2.6.1 Faecal Pathogen Survival In Conventional Wastewater Treatment Systems

Large investments in FS and wastewater treatment by Governments or international agencies is often done without an understanding of where faecal pathogens are released within a city, how people are exposed, or the potential effectiveness of investments to reduce disease. It is generally accepted that conventional wastewater treatment reduces the numbers of enteric bacteria. However, the extent to which this occurs can vary extensively depending on the treatment process (Anastasi et al., 2012). For instance, sewage treated by the activated sludge method or other biological processes often still contains faecal bacteria or pathogens (Guardabassi et al., 2002; Kay et al., 2008). To minimize the risk of environmental release of faecal bacteria, effluents are disinfected using oxidative processes to destroy or deactivate these organisms (Anthony et al., 2007; Hamilton et al., 2010). Chlorine is the most used disinfectant (Spellman, 1999). Alternative processes such as ozonation and UV irradiation are currently used for disinfection in many countries (Darby et al., 1993; Gómez-López et al., 2009; Hallmich & Gehr, 2010; Hijnen et al., 2006) or are under extensive review. These alternative processes may prove capital intensive for developing countries.

Traditionally, wastewater treatment plants are designed to eliminate organic matter and pathogens from aquatic environment. Engineers have traditionally focused on environmental objectives, with treatment primarily designed to protect downstream waterways, rather than to address faecal pathogen pathways in the urban environment (Mills *et al.,* 2018). Recent studies have shown that some sanitation interventions are not sufficient to mitigate possible routes of transmission (Freeman et al., 2017). Many human settlements throughout the world source their drinking water from rivers or water bodies that receive upstream inputs of human waste and effluents. Poorly treated wastewater from on-site and decentralised septic tanks and small treatment plants often infiltrates into groundwater, either intentionally or inadvertently (Headley et al., 2013). While many people in SSA lack access to improved sanitation, there is a widespread need for appropriate technologies that can reduce faecal pathogen risks while being simple and low-cost to build, maintain and operate (Headley et al., 2013). Natural systems and ecotechnologies, such as treatment wetlands, sand filters and ponds are often hailed as being an appropriate solution for such situations due to their robust operation and low maintenance requirements (Ansola et al., 2003; Mara, 2003).

Untreated wastewater contains a diverse range of pathogens that are potentially associated with water borne diseases, including various species of enteric bacteria, protozoa, cyanobacteria, helminthes, and viruses (Asano et al., 2007). Because of the costs and analytical difficulties associated with identification and enumeration of these organisms in water samples, it is common practice to test for surrogate microorganisms which indicate the presence of faecal contamination. One of the most used indicator organisms is *E. coli*, which is a member of the faecal coliform group of bacteria. *E. coli* is commonly found in the intestinal tracts of humans and other warm-blooded animals and, although most strains are harmless, its presence in water indicates faecal contamination (Mara, 2003). The concentration of *E. coli* in untreated municipal wastewater typically ranges from 10^5–10^8 MPN/100 mL, while the median infectious dose (e.g., the typical dose needed to cause disease in humans) is in the range of 10^6–10^{10} (Asano et al., 2007). However, recent studies revealed that detectable amounts of pathogens, as well as heavy metals and antibiotics, remain in effluents, even after sewage treatment (Spongberg & Witter, 2008).

2.5.2 Faecal Pathogen Treatment and Survival in Decentralised Faecal Sludge Treatment System

There are growing examples of successful innovative decentralised treatment systems that treat FS either on-site or off-site from on-site sanitation systems (OSS) (Semiyaga et al., 2015) in SSA. One of such decentralised systems is the innovative FS treatment system comprising of FS dewatering on sand drying beds and coupled with aerobic (open air) co-composting with municipal solid waste (Nikiema et al., 2013). To be able to access the public health risks posed by faecal pathogens and to minimize the risk of environmental release of these pathogens, decentralised faecal treatments plants need to be evaluated for their pathogen reduction profile and efficiency especially within the context of developing countries where these decentralized systems are promising. It is generally accepted that conventional wastewater treatment plants reduce the numbers of enteric pathogen, though the review in section 2.6.1 reveal that, the extent to which this occurs can vary extensively depending on the treatment process (Anastasi et al., 2012). There is a lot of information existing on behavior of pathogens in conventional wastewater treatment and the performance of wastewater treatment plants regarding pathogen treatment or removal (Al-Gheethi et al., 2018; Anastasi et al., 2012; Pärnänen et al., 2019) but very little information exist on the behavior of pathogen in terms of their reduction profile in decentralised FS treatment systems in SSA.

In a recent study in India, the treatment efficiency of FS and septage in a total of 11 FS treatment plants and one sewage co-treatment system were subjected to performance evaluations. The treatment systems reduced faecal coliform count in the range of 0–5 log values while the leachate passed through the series of treatment modules. But most of the systems did not meet the discharge standard limits (< 1000 MPN/100 mL) recommended by the Ministry of Environment, Forest and Climate Change and/or NGT, which has a more stringent norm. It is evident that the FSTPs with tertiary treatment modules like a sand filter, activated carbon filter followed by UV, ozone or chlorination is a good choice to remove the pathogens (coliform) from the discharge water (Vijayan et al., 2020).

However, in SSA there is very little information on the end-to-end assessment of the effect of treatment on pathogen reduction and reduction profile in the treatment chain. Mrimi et al., (2020) reports on the design and evaluation of a novel FS treatment system in Tanzania where *E. coli* and total coliforms were treated to reach safe guideline values for unrestricted agricultural reuse. Levira et al., (2023) also studied the operational performance of two novel FS treatment plants in Tanzania and found that digested and dried FS from one of the systems had *E. coli* concentrations of 2.5×10^6 CFU g^{-1} and was not recommended for unrestricted agricultural use based on the WHO guidelines. Hence very little research exists in SSA on the end-to-end performance assessment of the treatment for pathogen reduction efficiency and the risks posed thereof. The type of pathogens most found in FS depends on the state of health of the population, as well as on the OSS containment practices in developing countries.

The most commonly used indicator organisms are the *E. coli* O157:H7, *Salmonella sp*. (Guan & Holley, 2003) and Helminths (Berendes et al., 2015; Koné et al., 2007). And with the ongoing COVID-19 pandemic, traces of the SARS-CoV-2 genetic material have been found in wastewater and faecal material of infected populations (Balboa et al., 2021; Chavarria-Miró et al., 2021). Regrowth of pathogen is a problem for certain bacterial pathogens such as *Salmonella spp*. and *E. coli*, which, unlike some other bacterial species, viruses, protozoa and helminths, do not require a host organism for reproduction (Haug, 1993; Wichuk & McCartney, 2007). Poorly treated FS from on-site and decentralised treatment plants often infiltrates into groundwater or surface water, either intentionally or inadvertently (Headley et al., 2013) or end up on land when used as soil amendment. The microbial safety of organic amendments and fertilisers used in agriculture is required to prevent colonization by food-borne illness pathogens, like *E. coli O157:H7* and *Salmonella spp*. (Lemunier et al., 2005). The study of these pathogen behavior and reduction profiles in a FS treatment chain in SSA context is thus critical to minimize or eliminate public health risks to humans and the rest of the environment.

2.6 Storage Effect on Compost/Faecal Derived Fertiliser Quality

2.6.1 Compost storage

When organic fertiliser or compost is produced and not utilized immediately, it is kept or stockpiled until the next use. The period between when compost is produced, and it is utilized is referred to as storage. Compost storage simply means keeping compost in a condition where it is still moist and nutrient rich for the next season (Grant, 2021) or for the next use. Moreover, for a commercial or large-scale composting plants, storage of finished compost is equally an important step in the production cycle that requires critical attention. Based on the end use, compost can be stored for a considerable period, spanning from few days or weeks to years. This period of storage versus the conditions of storage will affect the quality or characteristics of the compost at the point of use. Good storage of compost will depend on the storage/environmental conditions, period of storage, mode of storage etc. Pathogen behaviour and nutrient dynamics of compost in storage is of utmost interest to both producers and consumers alike. FS used as composts are identified as a route of produce contamination, because they may contain pathogens of public health concern, such as *E. coli* O157:H7 and *Salmonella sp.* (Guan & Holley, 2003). This is especially of concern as FS is being treated for reuse in the global south. Apart from pathogens, characteristics such as trace elements (heavy metals) in soil amendment acts as a potential source of bioaccumulation in soils which can be transferred to crops and then to humans and animals. It is therefore critical to assess the level of pathogen growth or regrowth in FDF and to investigate how storage conditions affect the fertiliser characteristics.

2.6.2 Previous Studies on Compost Storage

Pathogen behaviour, nutrient dynamics, and other characteristics of composts in storage is of utmost interest to both compost producers, regulators, and consumers alike. A review of previous studies on compost storage reveal very limited studies on different aspects of compost storage ranging from pathogen prevalence, greenhouse gas emissions and compost's ability to degrade plastics for instance (Hao, 2007; Saadi et al., 2010; H. Yang et al., 2004; Zaleski et al., 2005). In relation to pathogen prevalence and potential for re-growth, Sidhu et al., (2001) performed storage experiments on anaerobically digested dewatered biosolids from wastewater treatment systems that were subsequently composted using windrows. The main objectives were to determine the role of indigenous microflora and maturity of composted biosolids in the inhibition of *Salmonella* re-growth in storage for two weeks to two years. They found a significant decline in the growth rate of seeded *Salmonella* in sterilized compost when the compost was stored, suggesting that bio-available nutrients declined with storage. However, in non-sterilized compost this was not the case. This suggested that the indigenous microflora played a significant role in suppression of *Salmonella* regrowth in composted biosolids. The conclusions from this study were that all composts seem to have the potential for regrowth, and the long-term storage of the compost is not recommended. The inactivation rate of *Salmonella* decreased significantly with longer storage times, most likely due to the decline in indigenous microflora over time.

Similarly, a review by Zaleski et al. (2005) on the survival, growth, and regrowth of enteric indicator and pathogenic bacteria in biosolids compost stored at various periods and revealed that: (1) growth of inoculated *Salmonella* and *E. coli* is possible in sterile biosolids compost. (2) growth or regrowth of indicators and *Salmonella* is inhibited in non-sterile compost, most likely due to competition or predation from indigenous microorganisms. (3) moisture is an important factor in the survival and potential regrowth of *Salmonella* and *E. coli* in compost. Temperature and desiccation become more important in decreasing survival at lower moisture contents. Zaleski et al. (2005) further pointed out that, some studies showed storage of biosolids did not seem to be an effective means for decreasing *Salmonella* in biosolids, because mature biosolids seemed to support growth of pathogens. Other studies suggest that storage alone can create safe biosolids products if maintained under specific moisture conditions, as moisture and temperature are critical in the survival or inactivation of enteric bacteria in biosolids and compost. Moist environments allow for survival of microorganisms, as moisture is necessary for cells to function properly. By maintaining particular temperatures and moisture, and by monitoring maturity, an indigenous bacterial population could be maintained in the stored biosolids, which should help prevent pathogen regrowth or recolonization during storage.

In another study by Wang et al. (2017), dairy compost with 20, 30, or 40% moisture content was inoculated with a mixture of six non-O157 Shiga toxin–producing *E. coli* (STEC) serovars at a final concentration of 5.1 log CFU/g. They were then stored at 22 and 4°C for 125 days. After the 125-day storage, the reductions of non-O157 STEC for 4°C and 20% MC, 4°C and 30% MC, 4°C and 40% MC, 22°C and 20% MC, 22°C and 30% MC, and 22°C and 40% moisture content storage conditions were >4.52, >4.55, 3.89, >4.61, 3.60, and 3.17 log CFU/g, respectively. All the survival curves showed an extensive tail, indicating non-O157 STEC can survive at least for 125 days in the dairy compost (H. Wang et al., 2017). Chen et al. (Chen et al., 2018) investigated the survival of *E. coli* O157:H7 and *Salmonella enterica* in animal waste – based composts (dairy manure – based and poultry litter – based composts) with different compost types, storage conditions and inoculum levels and concluded that *E. coli* O157:H7 and *S. enterica* could potentially survive for long periods of time in dairy and poultry composts. Some factors influencing the pathogen survival included compost type, storage condition and inoculum level. Some studies looked at the suppressiveness of stored composts on disease causing organisms of plants (Saadi et al., 2010). For example, Saadi et al. (2010) investigated the effect of storage conditions on cow manure compost's suppressiveness against fusarium wilt of melon, caused by *Fusarium oxysporum* f. *sp. melonis* (FOM) in relation to the dynamics of compost microbial activity and biodegradability. The mature suppressive compost was prepared from tomato plants, separated cow manure, and stored for one year under cool/warm or dry/wet conditions, in four different combinations: cool-dry, warm-dry, cool-wet, and warm-wet. All composts retained and even enhanced their suppressive capacity during storage, with no significant differences among them by the end of the storage period. However, significant differences were found in the dynamics of some of the measured chemical and microbial properties. The microbial activity of composts stored under wet conditions was higher than that of those stored under dry condition, which resulted in a substantial decrease in dissolved organic matter content.

Some studies also focused on both microbiological and chemical changes during storage. For example, Kleawklaharn and Iwai (2014) in their study utilized vermicompost made from cassava waste, soil and manure mix, which had been maintained for different periods (0, 1 and 3 months) to study changes in its biological and chemical qualities. Their study found that the amount of living bacteria and fungi remained stable until the third month. The pH and EC values decreased during storage in the first month from 8.5 to 7.5 and from 1.25 dS/cm to 0.84 dS/cm respectively, and increased again during storage in the third month, which were 8.33 and 2.18 dS/cm respectively. Changes in amount of total N and calcium (Ca) were not observed throughout the storage period. On the other hand, the amounts of total K and magnesium (Mg) decreased during storage in the first month and increased in the third month whereas the amount of total P increased during the first month of storage period and stabilized in the third month. Aside pathogens and nutrients, the biodegradation of plastics was also investigated in animal fodder compost stored at -20°C, 4°C and 20°C for different periods by Yang et al. (2004). A study by Hao (2007), investigated greenhouse gases (CO_2,CH_4 and N_2O) emission and nitrogen (N) levels on composted livestock manure that had been stored up 233days. The compost materials were produced by composting livestock manure for 133 d with 0, 10, 20 and 30% phosphogypsum (PG) or 10, 20 and 30% sand amendment. Results from this study indicated that TN content did not change but mineral N content increased significantly during the 233d storage for all treatments. There were only trace amounts of CH_4 and N_2O emissions. The C loss during storage was mainly as CO_2 and accounted for about 2.9 to 10% of total C initially in the compost.

Some early researchers also studied the persistence of antibiotics in stored compost. Their study investigated the dissipation and persistence of three groups of residual antibiotics (sulfonamides, quinolones, and tetracyclines) in anaerobically digested (AD) biosolids and compost during 28 days of storage under environmental conditions. Results showed that the total dissipation of sulfonamides was above 70%, which was higher than that of quinolones and tetracyclines. Quinolones were more persistent in compost than in AD biosolids. Nutrient (i.e. total phosphorus and total nitrogen) and dissolved organic matter contents were relatively stable, while total organic matter increased slightly during the storage of AD biosolids and compost. There was no significant variation in the concentration of soluble heavy metals in both AD biosolids and compost. The Pb content in AD biosolids significantly increased while an opposite profile was observed in compost during storage (Liu et al., 2019). Composts may go through some period of storage before being applied to land. In the next sub sections, we discuss the effect or influence of the various storage factors and conditions that could potentially affect compost characteristics and quality during storage.

2.6.2.1 Influence of Storage Duration on Compost Characteristics

The conclusions from a study by Sidhu et al. (2001) revealed that all composts seem to have the potential for pathogen re-growth, and this was affected by storage duration. In their study, seeded sterile biosolid compost and non-sterile biosolid compost were stored from 2 weeks to 2 years. It was found that long-term storage of the compost is not recommended as the inactivation rate of *Salmonella* decreased significantly with longer storage times, most likely due to the decline in indigenous microflora over time. In terms of changes in physico – chemical properties, Kleawklaharn and Iwai (2014) found that the chemical properties of the vermicompost were changing differently, and yet there was no clear trend in the changes of pH, EC, total potash and magnesium during the different duration (0, 1 and 3 months) of storage. Magnesium decreased in the first period of storage and increased again in later periods while the total nitrogen and total calcium were not significantly different throughout the study. There was very limited information however, on the effect of storage duration on characteristics such as mineralisation of nitrogen.

2.6.2.2 Influence of Storage Temperature on Compost Characteristics

Temperature conditions in storage is an important parameter to consider because differences in climatic conditions around the globe generate different temperature variations. In addition, different temperature regimes affect microbial activities. Yang et al. (2004) observed ice crystals in the compost stored at – 20 °C. Formation of ice crystals in the compost increased the concentration of salts in the liquid phase and thus caused osmotic stress to microbes. Intracellular ice crystals may also damage sensitive microbial cells (Stenberg et al., 1998). Therefore, it was expected that viable cells in the compost stored at 20 °C should be more numerous than those in the compost stored at – 20 °C. Similarly, it is expected that pathogens will be destroyed at such lower freezing temperatures by the same or similar mechanisms. Some pathogens can survive for long periods of time under stressful conditions (Wesche et al., 2009). In the study by Chen et al. (Chen et al., 2018), they observed that dairy and poultry composts can provide the long-term survival conditions for *E. coli* O157:H7 and *S. enterica*. For example, *E. coli* O157:H7 and *S. enterica* at the high inoculum level survived for longer periods (>168 days) in composts at 5°C than at 22°C and greenhouse conditions. Wang et al. (2017) reported similar findings for a non-O157 Shiga toxin–producing *Escherichia coli* (STEC) which survived better at 4°C than 22°C for 125 days in dairy compost.

It was becoming obvious that, one of the factors potentially influencing the abilities of *E. coli O157:H7* and *S. enterica* to survive in composts was temperature, with lower temperatures being more protective (Chen et al., 2018). In addition, the metabolic processes of bacteria are believed to slow down, as the temperature is decreased from the optimum (37°C for *E. coli O157:H7 and S. enterica*) (Farrell & Rose, 1967). Comparatively little is known about the underlying mechanisms of extended pathogen survival in animal wastes. Chen and Jiang (2017) highlighted the rapid induction of several genes in *S. Typhimurium*, including the rpoS (regulator of the general stress response), proV (osmoprotectant transporters), dnaK (chaperone protein) and grpE (heat shock protein) genes that could contribute to its desiccation survival in aged broiler litter. Moreover, entry into the viable but non – culturable (VBNC) state could also be a survival strategy for pathogens in response to harsh environmental conditions, which is believed to constitute an important reservoir of pathogens in the environment (Oliver, 2010). Given the ability of *E. coli O157:H7* and *S. enterica* to enter the VBNC state, the possible presence of these VBNC bacteria in animal waste-based composts thus poses a health risk to humans, since the cells may retain their infectivity.

2.6.2.3 Influence of Indigenous Microflora on Stored Compost Characteristics

Sidhu et al. (2001) in their study found that *Salmonella* growth increased significantly when indigenous microorganisms were not present, and the presence of indigenous microflora seemed to be the most important factor suppressing *Salmonella* growth. The authors speculated that the presence of somatic *Salmonella* phage, the ability of some indigenous microorganisms to outcompete *Salmonella* for limited nutrients, or the higher activity of indigenous microorganisms during favourable conditions may be the reason for suppression of growth. Hussong et al. (1985) drew similar conclusions that competition from other organisms suppressed the growth of *Salmonella* in compost. The role of microorganisms in suppressing bacterial pathogen re-growth in stored compost as a function of temperature and moisture was assessed by Pietronave et al. (2002). Both sterile and non-sterile composts, adjusted to moisture contents of 10, 40, and 80%, were tested. Samples were inoculated with *S. arizonae* or an enteropathogenic strain of *E. coli* and were incubated at either room temperature or 37°C.

In non-sterile compost, with moisture contents of 40 and 80%, *Salmonella* concentrations increased briefly and then declined below the original seeded levels within two weeks. The concentrations of organisms were lower in the 40% moisture sample than in the 80% moisture sample. Temperature did not have an effect at higher moisture contents, but it did influence the survival of *Salmonella* at 10% moisture. After 48 hours, concentrations were much lower in non-sterilized compost incubated at 27°C than in samples incubated at 37°C. In sterilized samples, *Salmonella* concentrations increased within 24 hours for all moisture levels, and then decreased gradually over 30 days. For *E. coli*, low moisture had a detrimental effect on the survival in both sterile and non-sterilized composts. At 40 and 80% moisture, *E. coli* levels initially increased and then decreased below the seeded dose at 40% moisture, while remaining above the seeded dose at 80% moisture.

In conclusion, the high suppressive effect against pathogens observed in non-sterilised compost indicated that the indigenous microflora played an important role in control of pathogen regrowth. The antagonistic effect on pathogens does not seem to depend on a single group of microorganisms. Although the compost indigenous microflora was key to the pathogens suppression, storage conditions also seem to influence this phenomenon. High moisture content and increased temperature positively affected the regrowth of the pathogens though the influence of temperature seemed weaker (Pietronave et al., 2002). It has been suggested that compost be sterilized as a final treatment, but this could allow for potential regrowth of pathogens due to the lack of competitive indigenous microorganisms in the compost (Zaleski et al., 2005).

2.6.2.4 Influence of moisture content and relative humidity on compost characteristics

High moisture content was found to positively affect the re-growth of the pathogens in stored composts (Pietronave et al., 2002). This similar trend was also observed by Wang et al. (2017) when dairy compost with 20, 30, or 40% moisture content was inoculated with a mixture of six non-O157 Shiga toxin–producing *Escherichia coli* (STEC) serovars and then stored at 22 and 4°C for 125 days. After inoculation, the non-O157 STEC population increased to 0.69 and 0.79 log CFU/g in the dairy compost with 30 and 40% moisture content at 22°C within the first day, respectively. For all other storage conditions, the pathogen population decreased rapidly. After 125 – day storage, the reductions of non-O157 STEC for 4°C and 20% moisture content, 4°C and 30% moisture content, 4°C and 40% moisture content, 22°C and 20% moisture content, 22°C and 30% moisture content, and 22°C and 40% moisture content storage conditions were > 4.52, >4.55, 3.89, >4.61, 3.60, and 3.17 log CFU/g, respectively. The mechanism for pathogen reduction under lower moisture content conditions could be that the desiccation of stored compost induced water stress in pathogens.

Little is known of the effect of moisture on nutrient loss or mineralisation of stored composts. While the moisture content is inherent, environmental conditions can also induce the availability of moisture in the atmosphere (relative humidity). Humidity factor affecting storage and handling of compost has not been researched into yet. One of the conditions that would influence the composts susceptibility to humidity is the type of bagging or packaging material. The driving force for the passage of water vapour is proportional to the difference between the atmospheric humidity and the critical relative humidity (CRH) of the fertiliser so that products of low CRH are more likely to give problems in prolonged storage unless packaged in more resistant (and higher cost) materials. Fertilisers of low CRH, such as ammonium nitrate, require a heavier gauge liner, and particular attention must be paid to proper sealing or closure of the liner. Fertilisers of high CRH, such as TSP, are sometimes packed in bags without a polyethylene liner or other vapour barrier (Clayton, 1998).

2.6.2.5 Influence of Bagging/Packaging Material on Compost Characteristics

In the chemical fertiliser industry, fertiliser bags generally have a strong outer bag to contain the product during handling and an inner liner of fairly light-gauge polyethylene to act as a vapour barrier and to prevent sifting or loss of product. The polyethylene layer is not completely waterproof, as is often assumed; it has some permeability to water vapour (Clayton, 1998). The ability for water vapour to move across will depend on the atmospheric humidity and the CRH of the compost. Little or no research had been carried on the effect of packing bags on quality of compost in storage. Yang et al. (2004) stored the compost in 500 ml glass bottle which were sealed with cotton plugs for their storage studies. Vermicompost were stored in polypropylene woven bags which were bound tight by Kleawklaharn and Iwai (2014). The bags were kept in a room at a temperature range of 24◦C to 35◦C with no humidity control, for storage periods of 0, 1, and 3 months. Adamtey et al. (2009) stored their compost in 50 – kg perforated bags for up to 2 years. There are variations in the packaging bags used for storage. It is worth research interest to investigate the impact of commonly used packaging on the characteristics of the composts stored therein.

2.6.2.6 Influence of Heavy Metals and Nutrients on Characteristics of Stored Compost

Heavy metals in soil amendment and soils have been known to cause toxicity issues. Heavy metals added to soil at a sufficiently high rate have been reported to result in both a decrease in the amount of microbial biomass and a change in the structure of microbial community (Giller et al., 1998) . Noticeably, microorganisms in soil subjected to a long-term heavy metal stress, even at modest levels of exposure, could not maintain the same overall microbial biomass as in the unpolluted soil. The detrimental effects of heavy metals, such as Cu and Zn, on micro-organisms in biosolids have also been previously documented (Chaudri et al., 1999; Kao et al., 2006). Therefore, the consideration of heavy metals in compost is clearly of a general interest, as it draws attention to the interaction that may occur in compost between pathogens and heavy metals. Further research is thus necessary to study the impacts of various heavy metals on the pathogen survival in compost.

Adamtey et al. (2009), investigated the type and form of inorganic N fertiliser that is capable of improving the nitrogen content of municipal waste co-compost and monitored the changes in the properties of this N-enriched product under storage for two years under ambient conditions. They found that in the first four months of storage, total N content of 50 kg Co-compost + 3.26 kg urea (CoUD) increased from 31,333 to 54,000 mg/kg, and 50 kg Co-compost + 7.14 kg $(NH_4)_2SO_4$ (CoASD) from 35,333 to 52,000 mg/kg. At the end of two years of storage, the initial N content of CoUD and CoASD decreased by 47% and 24%, respectively. Based on their results, it was recommended that dry $(NH_4)_2SO_4$ should be used in N enrichment of Co, and that the enriched Co should be stored in sealed bags but not more than four months. However, the sanitization effect of this enrichment process on indigenous pathogens has not been assessed. Enriching compost with inorganic fertilisers (e.g. urea) is reported to further sanitise the product (Vinnerås, 2007), enhance fertiliser use efficiency and return organic matter into soil, restoring soil health and improving crop yield on sustainable basis (R. Ahmad et al., 2008; Han et al., 2004), as well as reducing application rates. The synergistic effect between the organic matter from compost and inorganic fertiliser could have so many positive effects on nutrient conservation in soil, decrease soil erosion and improve soil structure (Ahmad et al., 2008).

Dalias and Christou (2020) studied the relationship between N mineralization and sheep/goat, cattle, swine and poultry manure storage duration prior to soil application. They found out that the percentage of organic N mineralized after soil incorporation was clearly greater for poultry, ranging between 41 and 85%, in relation to the other three manure types, for which maximum mineralization ranged between 4.5 and 66%. For sheep/goat, cattle, and swine, the interaction between mineralization and immobilization processes showed a distinct pattern with two phases of net N release during the twelve months of storage. They recommended that farmers should preferably use well-digested manures that have been aerobically stored more than six months to avoid materials that provoke intense immobilization, unless problems associated with the use of fresh manure are managed.

2.7 Compost/Faecal Derived Fertiliser Application on Crops and The Soil Environment

2.7.1 Effect of Application of Faecal Derived Fertilisers on Different Crops and Soil Properties

Faecal Derived Fertilisers (FDFs) are gradually revealing their effectiveness in the agricultural industry as suitable organic fertiliser. It has the potential to enhance crop growth comparable to mineral fertilisers (Sommer et al., 2013). Using FDF in agriculture has a long and geographically diverse history (Ferguson, 2014), occurring both formally in developed countries, and informally in the Global South (Christodoulou & Stamatelatou, 2016; Thebo et al., 2017), and can produce crop yields comparable to those grown with mineral fertiliser (Moya et al., 2017; Sommer et al., 2013). FDF application is reported to have positive effect on the physicochemical and biological properties of the soil which often leads to higher crop growth and yield and thus improving productivity(Abedi et al., 2010; Nartey et al., 2021). The crop productivity mainly relies on the available nutrients' proportions, application method and water application frequency. For instance, composted sewage sludge showed significant improvements in crop yield (Jindo et al., 2016; Ranjbar et al., 2019). The cause is attributed to the enhanced nutrient availability by virtue of the porous structure of the compost which allowed easy migration of soluble nutrients to the plant roots.

Ofori-Amanfo et al., (2018)found that the application of pelletized dried FS and compost resulted in an increase in lettuce production in terms of height, number of leaf, width, wet and dry weight. Pradhan et al., (2019) in a study showed that an even higher amount of lettuce can be produced using SDFS based fertiliser (sawdust and DFS co-compost) compared to other tested fertilisers. Results from a study conducted by Torgbo et al., (2018) showed that dried FS and municipal waste co-compost has the potential to help farmers increase their yield in cabbage production. This was because the co-compost performed far better than the mineral fertiliser and the control in terms of yield. Another study investigated the productivity of hydroponic tomato production in which the experiment was designed with three nutrient conditions: decentralised wastewater treatment system (DWTS) effluents, nitrified urine concentrate (NUC), and commercial hydroponic fertiliser mix (CHFM). Tomato plants treated with commercial fertiliser mix showed higher growth performance compared to those from human excreta derived materials. However, both DEWTS and NUC treated plants indicated specific positive effects: improving plant height, photosynthetic rate, fruit mass, harvest index, and nutrient uptake in DEWTS. In NUC, the result showed improved growth performances in terms of shoot dry matter, leaf area index, and chlorophyll content. Therefore, nutrient balances and the maximization in excreta could be adjusted and supplemented to enhance the production of tomatoes in the hydroponic system, which could meet the same nutrient quality level as mineral fertilisers (Magwaza et al., 2020) .

The application of the FDF pellets resulted in high maize (*Zea mays* cv. Abeleehi) and cabbage (*Brassica olereacea* var *capitata* cv. Oxylus) yields in a greenhouse trial conducted in Ghana (Pradhan et al., 2016). Karanja and Kamau (2013) reported significant increase in maize and Irish potato yields on using Peepoo compost. Nartey et al. (2021) concluded that crop yield was significantly higher ($p \leq 0.05$) in the FDF plots compared to the mineral fertiliser plots for all the selected crops (tomato, rice, maize, and pepper). The yield was 12% higher for tomato, 27% for rice and maize and 30% for pepper under the FDF plots. Increase in grain and straw yields from the combined application of compost and mineral fertilisers could be attributed to better crop growth, due to the readily available nutrients from the inorganic fertiliser sources and the improved nutrient availability and controlled release of nutrients from the co-compost (Seran et al., 2011; Suge et al., 2011) . Hossain et al. (2012) also reported a significantly higher grain protein (10.08 %) in maize from applying 22.5 t compost/ha and N–P–K (30–15–20 kg/ha), respectively as compared to the protein content in the control (4.85 %). Islam et al. (2016) found that production of hybrid rice in boro season gave more yields under the treatment of 10 t/ha composted DFS and ½ of the recommended mineral fertiliser dose compared to sole mineral fertilisers.

2.7.2 Residual Effect of The Application of Faecal-Derived Fertiliser on Soils and Different Crops.

The long-term cultivation of most tropical soils has resulted in severe depletion of organic carbon, the disintegration of soil aggregate stability, serious soil erosion and deterioration of soil fertility (Ding et al., 2016). Studies show an increase in the organic N pool in soils with a long history of animal manure application, which may increase the residual effect of N in soil (Bacca et al., 2020). This could reduce the need for additional applications of nitrogen sources (Zhang et al., 2012). As fertiliser rates are increased, the efficiency of fertiliser nutrient use decreases leaving behind in the soil an increasing proportion of the added nutrient (Loks et al., 2014). As previously reported in literature in section

2.8.1, FDF and other compost derivatives have been used to cultivate different types of crops with positive effects on the physico-chemical properties of soil, growth and yield of crops (Moya et al., 2019; Sommer et al., 2013).

In many of these studies, the direct mineralisation of one-off application of FDF were assessed on growth and yield components of crops and soils properties after a single cultivation cycle (Magwaza, et al., 2020; Nartey et al., 2021; Torgbo et al., 2018). So far, only one previous study had attempted to assess the residual effect of FDFs but limited it to two cultivations cycles (Pradhan et al., 2019). Hence, there are still gaps in knowledge on the residual effect of applications of FDF on crops and mineralisation in soils over successive cultivations. Studies on the long-term use of animal manure have shown great influence on N availability and the ability of these areas to supply N to crops, even years after the end of the applications. This is especially due to the changing dynamics of N (Müller et al., 2011). Part of the manure N is present in organic fractions and gradually made available over time (Petersen et al., 2012). There are variations in N availability among the different types of manure. Liquid manure (slurry) contains most total N in ammonium form (NH_4^+) (Webb et al., 2013), some of which may be immobilized and become part of the residual organic pool and only available to subsequent crops (Suarez-Tapia et al., 2018). It is beneficial for soil fertility to use nutrients from organic sources in different years and for different crops. Residual effects of organic fertilisers on chemical properties of soil were studied by (Tabibian et al., 2012) and found significant increase in soil organic matter and electrical conductivity. Reeve et al., (2012) reported a 1.6-fold higher total organic carbon (1.43 vs. 0.89 %), in a soil that was amended with compost 16 years before, compared to a soil that was not amended. Aside residual nutrients and organic matter, pathogens are also of great concern. The most important of these pathogens being *E. coli* O157:H7, *Salmonella* and helminths because they have been linked to many foodborne outbreaks especially *E. coli O157:H7*, *Salmonella* or Listeria which have been associated with the consumption of contaminated vegetables (Oliveira et al., 2011). Somehow, the application of FDF to soils have been recognised as a potential route for pathogen contamination to the soil biome with some reported studies actually confirming the transfer of pathogens from soil amended with contaminated compost/manure or co-composted sludge (Major et al., 2020; Oliveira et al., 2011, 2012). Outbreaks of infections caused by bacterial pathogens such as *E. coli* O157:H7, *Salmonella,* and *Listeria monocytogenes* on fruit and vegetable commodities consumed raw emphasize the importance of minimizing the risk of pathogenic contamination on produce commodities (Gurtler et al., 2018).

2.8 Summary Discussion and Conclusions

The current 1.2 billion population in SSA still experiences food shortages as the continent is not able to feed itself due to low and decreasing agricultural productivity. A major challenge to improving agricultural productivity is the declining soil fertility and low fertiliser application rates in the region. The review reveals that there is great potential to recover nutrients and organic matter from FS and FW at relatively low cost via decentralised treatment systems in SSA. Quantities of these waste generated are quite high and would continue to grow with population growth offering a win – win for improved sanitation and improved agriculture. However, the review has revealed that there is little or no information on how much of the major nutrients (N, P and K) are recovered from decentralised FS and FW treatment systems in SSA and the efficiency of recovery to aid treatment and protection of the receiving catchment. This information when obtained will aid in forecasting nutrients quantities that are recoverable from waste treatment to complement mineral fertiliser sources for sustainable crop production. It will also help in estimating pollution footprint/potential of such treatment systems within the entire catchment management. This aspect of research or knowledge is addressed in Chapter 3. The review also espoused the dearth of information on the pathogen reduction profiles of the decentralised FS treatment system in SSA. Very little information exists on the end-to-end assessment of the effect of treatment process on *E. coli* reduction and reduction efficiency in the treatment chain. The reduction profile and efficiency in a decentralised FS treatment is critical to minimize or eliminate public health risks to humans and the environment. The knowledge gap is addressed in Chapter 4.

Generally, not much information had been published on the storage of composted animal-based waste contrary to the large amount of information published on the storage studies on raw manure. Even so, most of the published research were on food borne pathogen reduction in composted manure and sewage sludge as influenced by storage conditions. There were conflicting views on the survival and regrowth of pathogens in storage. Some studies showed that storage of biosolids did not seem to be an effective means for decreasing *Salmonella* in biosolids, because mature biosolids seemed to support growth of pathogens. Other studies suggested that storage alone can create safe biosolids products if maintained under specific moisture conditions, as moisture and temperature are critical in the survival or inactivation of enteric bacteria in biosolids and compost. There was comparatively less information on the changes non-microbial aspects of composted animal waste during storage. Very few studies have looked at how storage conditions influenced changes in pH, EC, nutrients and heavy metals or the evolution of greenhouse gases during storage. From the review carried out, it can be deduced that not much research had been conducted extensively on FDF in storage. It could be argued that FS can be likened to manure or sewage sludge however, there exist differences between these wastes based on their sources of generation making them possess unique characteristics.

As FS is fast becoming an alternative source of fertilizing material in SSA, there is very little or no information on storage properties of FDF during storage in recent literature. More research is needed to determine the actual re-growth potential of indigenous pathogens, especially *E. coli* and *Salmonella* in FDF after die-off and subsequent storage under different conditions. It is still not understood if these pathogens die, or if they enter a "viable but non – culturable" state. Understanding this phenomenon is crucial in determining if regrowth of indigenous pathogens is possible in storage. There is little or no information on the influence of storage conditions on the antimicrobial sensitivity patterns of food borne pathogens, *E. coli* and *Salmonella* in FDF in long-term storage. These gaps are addressed in Chapter 5. The effect of continuous application of FDF on direct soil properties, crop and the soil environment have not been fully investigated. Similarly, little is known about the residual effects of one-time application of FDF on specific crop types following successive cultivation on soil nutrient, nutrient uptake dynamics, crop yields and microbial safety. In Chapters 6 and 7, the gaps in knowledge on the residual effect of applied FDF on crops and the associated economic benefits of the residual effect are addressed respectively.

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Chapter 3 Nutrient dynamics and recovery efficiencies in a faecal sludge and food waste treatment system

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Abstract

Nutrients in wastewater treatment pose significant challenges to the environment and public health. The purpose of treatment is to reduce amount of nutrients in effluent going into the receiving environment. In this study, we determined the total nitrogen (N), phosphorus (P), and potassium (K) fluxes, losses, and recovery efficiencies in a faecal sludge (FS) and food waste (FW) treatment system in Ghana. Total NPK recovery was estimated based on the amount of waste and nutrient concentration. The NPK dynamic at each treatment step was calculated using the mass balance principle. Results indicates that, between 270.4 – 1066.5, 122.4 – 300.3, and 214.9 – 456.8Kg of total N, P, and K respectively, entered the decentralised treatment system via FS and FW. Out of these amounts, the range of total NPK recovered in co-compost was 25.2 – 66.0 kg N, 43.9 – 61.8 kg P and 30.0 – 95.3 kg K while in the treated effluent leaving the system was 2.2 – 3.8 kg N, 0.6 – 1.0 kg P and 7.5 – 9.9 kg K. Approximately more than 50% of total N originating from FS is lost in all cycles at the dewatering stage of treatment. The treated effluent quality after the stabilisation ponds did not meet the minimum discharge standards for nutrients due to high solid content associated with higher nutrient content of FS compared with wastewater. It is recommended that the data obtained be used to build future predictive models for FSTP in sub-Saharan Africa.

Keywords: faecal sludge; nutrients; food waste; mass balance; treatment efficiency; decentralised; treatment plant

3.1 Introduction

Nutrient management from waste treatment plants pose significant challenges to the local environment and public health. One major objective of waste treatment is to reduce amount of nutrients, especially nitrogen (N), phosphorus (P) and trace elements in effluent going into the receiving environment. Excessive levels of nutrients and trace elements causes pollution of surface and groundwater bodies and could lead to bioaccumulation in living tissues. While cities are sinks for nutrient-rich food products, they also represent sinks for wastes including faecal sludge (FS) and uncooked food waste (FW) which makes them potential locations for resource recovery (Keraita et al., 2015). Relatively little research has been conducted on FS treatment processes, as non-sewered sanitation has only recently been acknowledged as a long-term sustainable solution (Strande et al., 2014, 2018). FS is quite different from wastewater sludge; for example, FS can be comprised of any range of fresh excreta to products of anaerobic digestion from storage in containment, and can include soil, sand, and municipal solid waste (Strande *et al.*, 2014; van Eekert *et al.*, 2019). In contrast, sewage sludge is relatively fresh, not stabilised, easily settleable solids from raw wastewater (Tchobanoglous et al., 2014), and activated sludge is mainly composed of bacterial cells and metabolic products generated during aerobic secondary treatment (Nielsen *et al.*, 2004). The demand for infrastructure to manage FS is increasing, and reliable methods are being developed and validated to estimate total accumulated quantities and qualities of FS to support management and treatment technology solutions (Englund et al., 2020; Strande et al., 2018).

As the supply of mined P dwindles (Cordell & White, 2014), and environmental and economic costs of synthetic N increases (Canfield *et al.*, 2010; Pikaar *et al.*, 2017), urban FS constitutes an untapped opportunity for nutrient recycling. FS contains approximately 50% carbon (C), 4 – 5% N, 2 – 3% potassium (K), and 2 – 3% P (Onabanjo et al., 2016; Rose et al., 2015). However, most of these nutrients are lost, mostly in low-income countries due to poor sanitation and solid waste mismanagement resulting in widespread pollution of waterways, intensified by unplanned urban expansion (Currie et al., 2015). Water pollution by N and P has been a major driver of efforts towards more sustainable nutrient recovery (Cordell et al., 2011). Options that allow for the recovery of nutrients, water, and organic matter from FS such as co-composting are desirable, especially where such an option is low-cost, efficient, and appropriate to the local socio-economic and environmental context.

There have been many attempts at implementing conventional wastewater treatment technologies (e.g., rotating biological contactors, activated sludge), but they have not been well adapted to the SSA context for several reasons, among which the high cost of installation, the availability of a reliable energy supply, and local skills and human resources are prominent. Part of the solution is implementing decentralised FS treatment systems, as expanding centralised sewer networks in established urban areas is not feasible (Nansubuga et al., 2016; Semiyaga et al., 2015). Hence, more "natural" or passive systems also termed low-cost technologies (Strauss *et al.*, 1997) provide a promising alternative in SSA and in the other developing regions. There are growing examples of successful innovative decentralized treatment systems that treat FS either on-site or off-site from on-site sanitation systems (Semiyaga et al., 2015). These innovations focus on resource-recovery as a way to help offset costs, but also to create win-win scenarios between sanitation and other sustainability goals like clean energy, sustainable consumption and production, and food security (McConville et al., 2020). Examples of decentralised FS treatment include co-composting, vermicomposting, conversion to animal protein via black soldier flies, pyrolysis, and larger scale natural systems such as waste stabilisation ponds and constructed wetlands (Semiyaga et al., 2015). One of such commonly used decentralised treatment technology is the innovative FS treatment system comprising of FS dewatering on sand drying beds and coupled with aerobic (open air) co-composting with municipal solid waste (Nikiema et al., 2013).

This system entails various treatment stages. It is expected that there will be a flow and transformation of specific nutrients (nutrient fluxes) in the system from start (feedstock input) to end (finished products). Nutrient fluxes in a FS treatment can be quite a complex mechanism involving many biological and physicochemical transformational processes such as mineralisation into leachate, volatilisation into the atmosphere, assimilation into living tissues of microorganisms, remineralisation in substrates, adsorption to surfaces, sedimentation etc. These transformations could also be affected by changing climatic conditions (raining and dry season) on feedstock availability and quality. It is therefore important to assess how this innovative FS treatment system recovers nutrients after treatment and the efficiency of the recovery in a developing country context. The quality of such end product meant to be used as faecal derived fertiliser should be able to meet quality guidelines to promote wider acceptance, and to access regional and international markets (lucrative export business) (Moya *et al.*, 2019).

While the main purpose for the decentralised FS treatment system is to recover valuable nutrients and organic matter back into soils, there is very little or no information on end-to-end assessment of the nutrient fluxes from the feedstock to the finished product as well as the efficiency the recovery. Earlier studies have only looked at individual components of the FS transformation such as dewatering and co-composting separately. These studies focused on dewatering efficiency of different FS types using different drying/dewatering methods (Getahun et al., 2020; Gold et al., 2018; Stefanakis & Tsihrintzis, 2011; Ward et al., 2019). Others studied pollutant removal efficiencies of FS dewatering systems such as constructed wetlands/vertical – flow constructed wetlands (Kengne et al., 2009, 2011; Koottatep et al., 2006). Research studies on FS co-composting have been on the optimization of the with various organic wastes (Al-Muyeed et al., 2017; Berendes et al., 2015; O. Cofie et al., 2009; Hashemi et al., 2019; Koné et al., 2007; Mengistu et al., 2018; Nakasaki et al., 2011; Nartey et al., 2017; Thomas et al., 2018) and the effectiveness on FS sanitation during open air composting (Al-Muyeed et al., 2017; Fidjeland et al., 2013; Manga et al., 2021; Mengistu et al., 2018; Nartey et al., 2017).

However, no studies have been conducted on the end-to-end assessment of the nutrients fluxes and nutrient recovery efficiency of the decentralised FS treatment system. This information on assessing the nutrient fluxes in a developing country setting is crucial to understanding the potential for nutrient capture and reuse, as well as protecting the local environment and supporting decision making on sustainable sanitation planning. Therefore, the main objective of this study is to determine the NPK fluxes, losses, and recovery efficiencies in a faecal sludge and solid waste treatment system.

3.2 Materials and methods

3.2.1 Study location description

The study was carried out in field scale at Akorley, Somanya (latitude 6° 6' 0" N and 3° 3' 0" N and between longitude 0° 0' 30" W and 0° 0' 10" W, Sadiq, 2016) at the Jekora Ventures Limited (JVL) – Yilo Krobo Municipal Assembly (YKMA) Recycling Plant, in the YKMA of Ghana (Figure 3-1). The annual rainfall of the area ranges from 750 to 1,600 mm and it's spans from May to October (bimodal). Average temperatures range between 24 and 30°C while relative humidity ranges between 60 and 90% (Sadiq, 2016). The major soil type is savanna ochrosol (Eastern Regional Co-ord Council, 2016). It has low nutrient reserves, with the topsoil consisting of dark greyish brown humus sandy or clay loams (Eastern Regional Co-ord Council, 2016). For the entire municipality, the potential/estimated FS generation is 15,000 m^3 per year from major on-site sanitation systems (OSS) (Nikiema et al., 2016). The main productive activity in YKMA is agriculture (e.g., mango plantations, food crop farming and livestock rearing), which employs around 60% of the population.

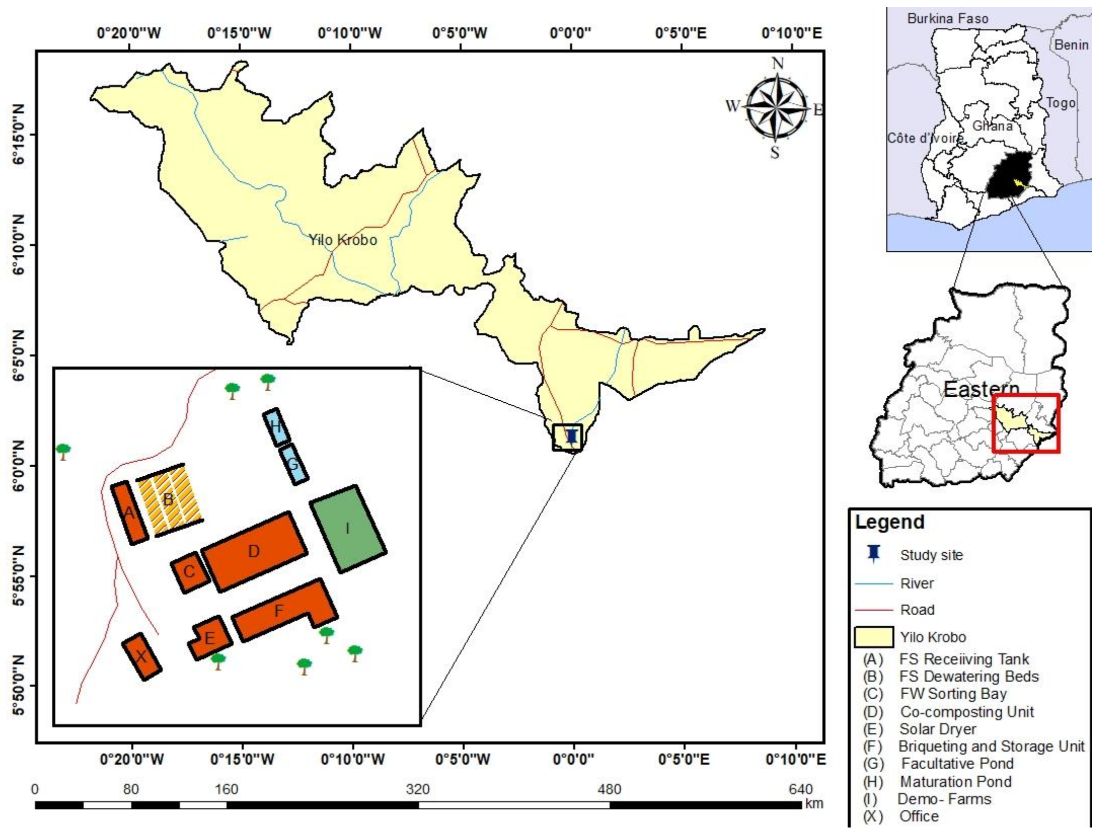


Figure 3-1. Map of study location showing study site

3.2.2 Study site description (operational conditions)

The study site was the FS and FW co-composting system (Figure 2) of the JVL - YKMA Recycling Plant. The treatment system treats both FS and uncooked FW into faecal derived fertiliser. The units include FS dewatering on sand drying beds, effluent and/or leachate treatment, and then co-composting and storage. The treatment chain is described in the following steps:

a) Receiving of FS from vacuum trucks from either pit latrines or septic tanks and dewatering on sand drying beds. Receiving source separated uncooked FW from local markets.
b) Removal of DFS from drying beds and co-composting with source separated FW at a ratio of 1:3 w/w.
c) Treatment of effluent in facultative and maturation ponds for reuse of irrigation.
d) Storage of mature co-compost.

3.2.3 Sampling approach

3.2.3.1 Field sampling plan and sample collection

The sampling plan was developed following the treatment process from start to end (end -to -end) (Figure 3-2). The points at which the material transformed or was expected to transform was delineated as sampling points (SP) and the justification provided as follows:

a) Sampling FS from trucks - SP 1

FS emptied from pit latrines and septic tanks by vacuum trucks and brought to the treatment plant (SP 1). SP 1 was the point at which all FS entered the treatment system.

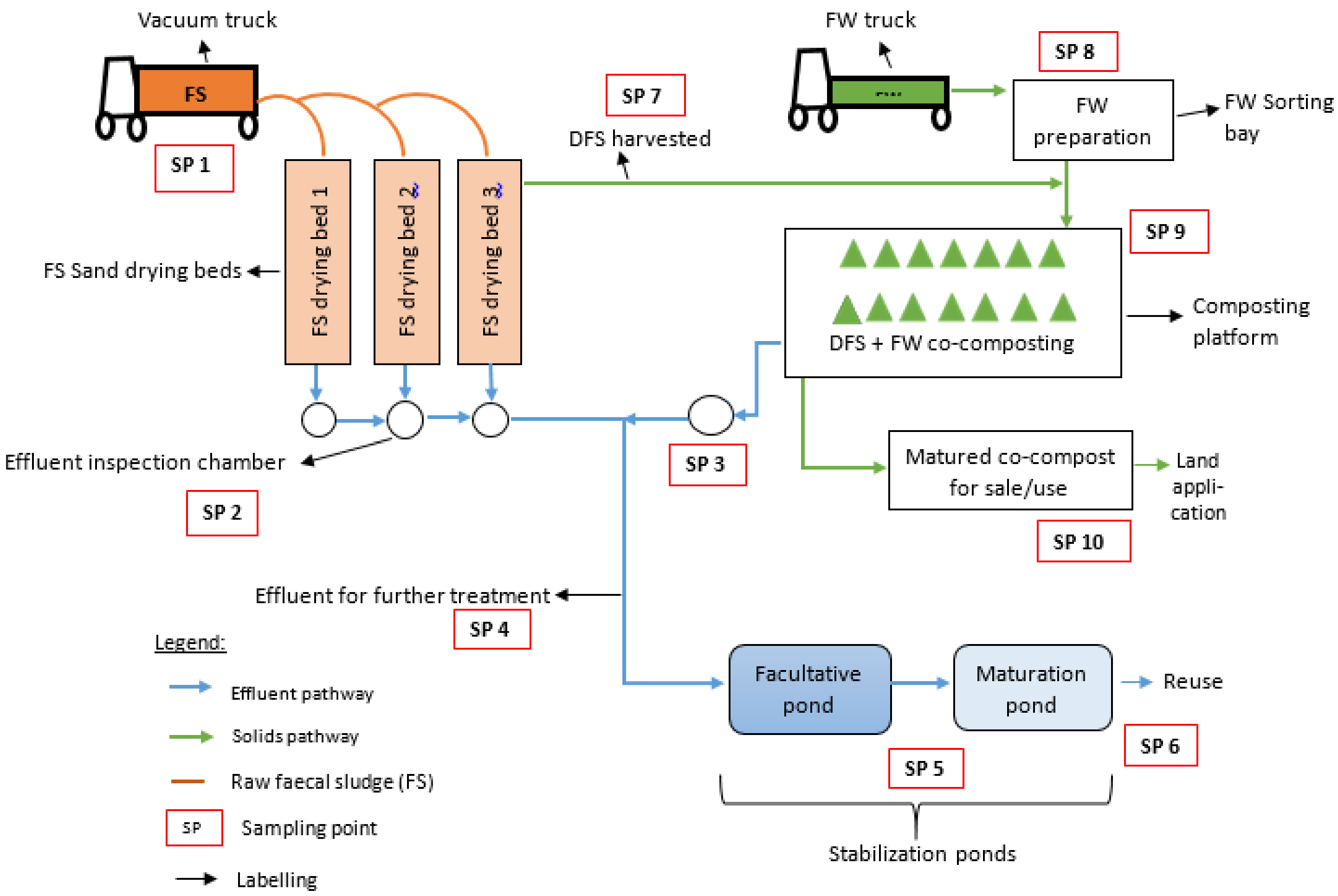


Figure 3-2. Schematic of the decentralised FS and FW treatment system

To characterise the FS that entered the treatment system, each truck was sampled at the point of discharging the FS onto the drying beds. Samples were collected directly from each truck following sampling methods described by Klingel et al. (2002) and Bassan et al. (2013). Equal volume of FS was collected during the following points of discharge:

(a) The beginning (valves opened)

(b) Middle of discharge

(c) Ending of discharge when pressure is low.

The samples were mixed/bulked together, and a composite sample taken. All the samples from all the trucks that fed the drying bed were again bulked together and a composite sample taken in triplicates to the labs for analyses.

b) Sampling percolate from dewatering beds - SP 2

On the drying bed, the FS underwent solid – liquid separation where free water filtered through the media and the dewatered FS (DFS) remained on the sand layer. The percolate, which was accessible in the inspection chamber (SP 2), was sampled at the same time every day from the start of loading until day 5 when active filtration was done (when 50% volume of sludge is dewatered). The samples collected after the 5 days were bulked together and a composite sample taken and analyzed in triplicate.

c) Sampling leachate from co-composting platform - SP 3

Leachate production is not regular on the composting platform. As and when it was produced, it was channeled through the inspection chamber (SP3). The leachate at this point was collected on a daily basis and bulked together to form a composite sample, which was analyzed in triplicate.

d) Sampling influent to facultative pond - SP 4

The percolate from the dewatering process (SP2) and the leachate from the composting platform (SP3) meets and mixes at this point before going into the facultative pond. This point (SP 4) became the influent of the facultative pond. The influent was sampled at the time the drying beds and composting platforms were actively being used. The sampling procedure followed the same description for SP 2.

e) Sampling effluent from facultative pond - SP 5

The treated effluent leaving the facultative pond became the influent to the maturation pond (SP 5). The effluent from the facultative pond was collected from four points and bulked into one sample. This was collected once a week.

f) Sampling effluent from maturation pond - SP 6

The treated effluent leaves the plant ideally at this point. From this point, all pollutants including nutrients may been recovered and/or reduced in the effluent to meet discharge guidelines. The effluent from the facultative pond was collected from four points and bulked into one sample. This was collected once a week.

e) Sampling of DFS from drying bed - SP 7

After the solid-liquid separation of the FS at SP 1, the harvested DFS (SP 7) is what goes into the co-composting process. Grab samples of DFS was taken from five points along the diagonal lines of the drying bed. These grab samples were bulked together to form a composite sample from which representative samples were taken for analyses.

f) Sampling of uncooked FW - SP 8

The source-separated FW arrives at the sorting bay (SP 8) of the treatment plant as the second feedstock to be used for the co-treatment of FS. As such, several grab samples of FW were taken and bulked together to form a composite sample from which samples were taken to the lab for analysis.

g) Sampling at the start of FS and FW co-composting - SP 9

This is the point where the DFS and FW were mixed in a known ratio to begin the composting process (SP 9) in Figure 2. Samples of the heap were taken at the initial mixing of the feedstock to form the heap. Grab samples at different points of the mix were collected and bulked together to form a composite sample, which was analysed in triplicate.

h) Sampling of matured faecal derived fertiliser - SP 10

A composite sample of the matured heap following the bulking up of several grab samples was taken and analysed in triplicate.

3.2.3.2 Duration and frequency of sampling

In this study, a phase is defined as one complete FS dewatering operation per drying bed while a cycle is defined as one complete treatment of FS received at the treatment plant until maturation pond is full. The duration for a complete P is on average 3 – 4 weeks while one complete C of treatment on average is from 4 – 6 months. The cycle was repeated three times.

3.2.3.3 Parameters for analyses

The nutrient parameters analysed were total N, total P, and total K.

3.2.4 NPK mass balance of FS and FW treatment system

The NPK flow in each treatment step was calculated using the mass balance principle, similarly as described in the previous works of (Laner & Rechberger, 2016). The mass of all input NPK (imports) into the treatment process (system) equals to the mass of all output NPK (exports) of this process (system) plus the change of stock (ΔS) that considers accumulation (ΔS > 0) or depletion (ΔS < 0) of material in the process (system), as shown in equation 1:

$$InputNPK = OutputNPK + \Delta S \qquad 1$$

The total NPK flows for the entire treatment system is shown by equation 2:

$$\Sigma feedstockNPK = \Sigma dewateringNPK + \Sigma co - compostingNPK + \Sigma effluent\ NPK \qquad 2$$

The total NPK dynamics around the FS dewatering treatment stage is shown in equation 3:

$$InputNPK\ (raw\ FS) = OutputNPK(DFS) + OutputNPK(effluent) + NPK(losses) \qquad 3$$

The total NPK dynamics around the FS and FW co-composting is shown in equation 4:

$$InputNPK(initial\ compost) = OutputNPK(final\ compost) + NPK\ (losses) \qquad 4$$

The total NPK dynamics around the effluent treatment stage is shown in equation 5:

$$InputNPK(influent) = OutputNPK(treated\ effluent) + NPK(losses) \qquad 5$$

3.2.5 NPK recovery efficiency and loss calculation

The efficiency of NPK recovery at stages of the treatment process was calculated based upon the influent concentrations and the effluent concentrations (Gross, 2005):

Treatment capacity and removal efficiency of the treatment

$$\text{Removal Efficiency} = ([(\text{Cin} - \text{Cout})]\ \text{x}\ 100\%)/\text{Cin}$$

Where:

C_{in} = Influent concentration (mg/L)
C_{out} = Effluent concentration (mg/L)
100% = Efficiency expressed as a percentage

The amount of NPK lost to the treatment was calculated from the differences between the initial nutrient concentration and the final concentrations at the different stages in the treatment chain.

3.3 Results and Discussions

3.3.1 FS loading of the treatment system

The treatment plant under study serves a population of 122,705 according to the 2021 census in the Yilo Krobo municipality (Ghana Statistical Service, 2021). The municipality has 36,371 households that are served by private vacuum truck operators for their FS contained in the various on-site sanitation systems (OSS) (Ghana Statistical Service, 2021). The study lasted from August 2021 – June 2022 over three cycles. The system received FS based on client demand in this case the vacuum trucks and availability of empty drying beds. The number of trucks received at the plant during the study period with their corresponding volume of FS discharged is shown in Table 3-1. Because the sizes of the trucks varied, the higher number of trucks received at the plant did not necessarily correspond to higher volumes of FS received. On average, 9 trucks were received per drying bed in Cycle 1, 5 trucks in Cycle +2 and 6 trucks in Cycle 3. Cycle 1's four phases recorded respective volumes of 55m^3, 52m^3, 45m^3, and 50m^3 (Table 3-1). A total of 202, 155 and 128m^3 of FS were received and treated for Cycles 1, 2 and 3, respectively (Table 3-2).

Table 3-1. Volumes of FS received and processed during the various phases and cycles of treatment.

Treatment Cycle	Phase	No. of FS trucks received	Volume of FS received (SP1) (m^3)	Volume of collected percolate after dewatering (SP2) (m^3)	Total volume treated for discharge (SP6) (m^3)	Fresh weight of DFS collected (SP7) (Kg)	Number of heaps formed with FW
Cycle 1	Phase 1	10	55	25.67	84.71	2927.4	1
	Phase 2	9	52	8.04		1962.7	1
	Phase 3	10	45	28.59		1400.9	1
	Phase 4	6	50	22.41		1693.9	1
Cycle 2	Phase 5	7	35	16.29	61.91	1075.1	1
	Phase 6	6	44	24.21		1383.8	1
	Phase 7	4	34	7.28		1097.0	1
	Phase 8	4	42	14.19		442.7	1
Cycle 3	Phase 9	7	54	23.32	61.82	2206.0	1
	Phase 10	5	25	10.78		2545.6	1
	Phase 11	6	31	20.68		3080.0	1
	Phase 12	7	39	7.04		5930.0	1

For each of the three cycles, the study found that the average properties of FS varied in characteristics such as volume and strength/texture by visual appearance. This affected the desludging (filling a drying bed) and the dewatering times as reflected in the period of desludging measured in Table 3-2. Sludge properties may vary depending on where it is produced, how it is generated, the seasonal variation, the type of OSS in place etc. (Yaser et al., 2022). The sources of FS for the study were mostly from homes (predominantly using septic tanks) and public toilets (using pit latrines). This corroborates that, septic tanks, and Ventilated Improved Pits (VIP) are the two types of OSS that are frequently used in towns and peri-urban communities in Ghana (Fanyin-Martin et al., 2017; Nayebare et al., 2020). The public toilets use little flush water in contrast to septic tanks, which employ flush systems like water closets that need water for their operation (Oppiah-Mensah, 2019). As a result, FS from the public toilets were usually more fresh, thick, and concentrated (high strength) while the FS from the septic tanks were more stable, watery, and more diluted (low strength).

While the type of OSS may have contributed to the properties of the FS received, the climate may have contributed as well. The climate of the area is in two seasons, rainy and dry season. The rainy season is bimodal with the major season occurring from March – July and minor season occurring from September – November each year. The dry season, characterised by cold dry weather occurs from November – February (Ministry of Food & Agriculture, 2023). In this study, cycles 1, 2 and 3 coincided with the minor rainy season, dry season, and major rainy season, respectively. The seasonality effect on OSS, is such that, septic tanks take relatively shorter time than usual to fill during the wet or rainy seasons (between the months of April and July), leading to more frequent desludging following the season.

Table 3-2. FS loading time and volume during the various phases and cycles of treatment.

Treatment Cycle	Phase	FS loading time	Volume (m^3)	Total volume (m^3)
Cycle 1	Phase 1	21 – 28 August 2021 (7 days)	55.0	202.0
	Phase 2	22 – 28 September 2021 (6 days)	52.0	
	Phase 3	29 September – 7 October 2021 (8 days)	45.0	
	Phase 4	14 – 16 October 2021 (2 days)	50.0	
Cycle 2	Phase 5	23 – 26 October 2021 (3 days)	35.0	155.0
	Phase 6	20 November – 22 December 2021 (32 days)	44.0	
	Phase 7	31 December 2021 – 15 January 2022 (14 days)	34.0	
	Phase 8	12 – 14 February 2022 (2 days)	42.0	
Cycle 3	Phase 9	11 March – 20 April 2022 (40 days)	54.0	128.0
	Phase 10	2 – 6 May 2022 (4 days)	25.0	
	Phase 11	8 – 22 May 2022 (14 days)	22.0	
	Phase 12	3 – 13 June 2022 (10 days)	27.0	

3.3.2 Total NPK Treatment efficiency in the FS and FW Treatment system

3.3.2.1 Treatment efficiency of the treatment stages

FS has very high oxygen demand due to readily degradable organic matter but also has substantial concentration of nutrients. The average total N, P and K concentrations of the FS arriving at the treatment facility showed high variability between phases in each treatment cycle (Table 3-3). This variability was dependent on the pH value, storage length, presence of oxygen and type of FS/OSS installation. Similar high variability of FS was observed in Yaounde by Kengne et al., (2011). Total N is the sum of organic nitrogen and ammoniacal nitrogen. The total N values were between 912.3 – 5119.1 mg/L (Table 3-3). This observed total N is higher than the total N values between 350 – 400 mg/L reported by the NIUA (2022) in India for FS and septage. Phosphorus is present as phosphate either in the acidic or basic form of orthophosphoric acid or as organically bound phosphate. The measured values ranged between 66.8 – 724.6 mg/L, higher in comparison to the 140 – 350 mg/L reported in India (National Institute of Urban Affairs (NUIA), 2022). There were significant differences at $p<0.05$ between total N concentrations of the FS between cycles. However, differences in total P and K concentrations were not statistically significant between the three cycles of treatment (Table 3-3). The highest N concentrations observed in cycle 3 of treatment and may have been due to the high volume of public toilet sludge received in comparison to sludge from septic tanks which were relatively more stable and contained relatively less nutrients. Kuffour et al., (2019) reported similar observations that, public toilet FS was usually fresh and undigested. It had very high concentration for all parameters while the FS from septic tanks had comparatively lower concentrations of the parameters analysed.

The high variability of FS observed in this study between phases is corroborated by results of other studies in West Africa (Heinss *et al*, 1999; Bassan *et al.*, 2013). The nutrient variability is due to factors such as storage duration, climate, type of OSS, and pump capacity of emptying truck (Bassan *et al.*, 2013). After dewatering, the nutrient concentration of the percolate is shown in Table 3-3. The differences in total N, P and K concentrations in the percolate after dewatering were not statistically significant at $p<0.05$ between cycles. The concentration of total K in the percolate after dewatering did not significantly differ from the FS at $p<0.05$ for all the cycles. Similarly, total N concentrations in cycle 1 and cycle 2 did also differ significantly between the FS and the dewatered percolate. Only in cycle 3 that total N and P concentrations significantly reduced in percolate after dewatering. This observation simply points to the fact that the dewatering process on sand drying beds essentially only focuses on solid – liquid separation of the FS. Harvesting the solids for further reuse and not so much on percolate polishing.

Table 3-3. Mean ± SD concentrations of NPK in FS before and after dewatering on sand drying beds.

Treatment Cycle	**FS (mg/L)**			**Percolate after dewatering (mg/L)**		
	Total N	Total P	Total K	Total N	Total P	Total K
Cycle 1	821.0 ± 72.8	599.8 ± 85.1	894.8 ±290.6	719.0 ± 88.6	50.1 ± 5.4	467.5 ± 104.8
Cycle 2	1346.3 ± 266.2	366.5 ± 30.3	857.0 ± 479.2	1546.0 ± 342.8	22.2 ± 3.9	1032.0 ± 212.7
Cycle 3	5055.6 ± 74.9	519.0 ± 59.4	1449.0 ± 1474.9	1255.8 ± 131.3	41.0 ± 19.1	1569.5 ± 1333.7

SD: standard deviation

The estimation of the treatment efficiency of the sand drying beds during the dewatering process for the different treatment cycles is shown in Table 3-4. The treatment efficiency for total K between the cycles was not significant at $p<0.05$ and seems to achieve the least removal from the FS. The values show an accumulation of total K concentration in the percolate. This may have been due to enrichment from the sand filtering medium. There was no significant difference in P treatment efficiency between cycles. And for total N, there was significant difference in efficiency between cycle 1 and cycle 3. This treatment efficiency for N in this study for cycles 1 and 2 is lower compared to planted drying beds (constructed wetland systems), which achieved pollutant removal rates of 80 - 90% for total N (Kengne et al., 2011; Koottatep et al., 2005) and the 75.9 – 81.6% achieved by unplanted drying bed (Kuffour et al., 2019). Only cycle 3 falls within the reported rates for unplanted drying beds. Nevertheless, the percolate remained of relatively poor quality, with concentrations close to those of domestic wastewater and therefore necessitated additional treatment before discharge into the environment (Kengne et al., 2011).

Table 3-4. Mean ± SD efficiency of sand drying beds in FS and FW treatment system.

Treatment Cycle	Treatment efficiency (%)		
	Total N	**Total P**	**Total K**
Cycle 1	12.3 ± 8.9	91.6 ± 0.8	41.2 ± 27.8
Cycle 2	-15.3 ± 19.3	93.9 ± 1.1	-46.4 ± 62.5
Cycle 3	75.2 ± 2.3	91.7 ± 4.6	-528.2 ± 953.6

The percolate after the dewatering stage was further polished in the facultative and maturation ponds. The treatment efficiency of the stabilisation ponds is described in Table 3-5. The treatment efficiencies exhibited by the ponds are quite variable. The negative efficiencies observed around the FS dewatering process in Table 3-4 indicate negative contaminant removal i.e., higher concentrations in effluent than in influent. According to Kumar et al., (2022) negative removal is a commonly found phenomenon in all wastewater treatment facilities irrespective of the pollutants, amount of wastewater, capacity of the treatment plants, and regions. Even the most efficient pollutants separation techniques will have negative removal efficiency for more than one pollutant (Kumar et al., 2022). The negative elimination or removal of total N, P, and K observed at various stages of treatment in this study can be attributed to factors such as accumulation from treatment components (e.g., sand drying bed), operating parameters such as varying retention times and climatic conditions (ambient temperature, rainfall etc).

Table 3-5. Mean ± SD efficiency of stabilisation ponds in FS and FW treatment system.

Treatment Cycle	Treatment efficiency of facultative pond (%)			Treatment efficiency of maturation pond (%)		
	Total N	**Total P**	**Total K**	**Total N**	**Total P**	**Total K**
Cycle 1	87.9 ± 0.0	64.3 ± 0.1	65.8 ± 0.0	40.8 ± 0.9	13.2 ± 4.0	16.2 ± 17.1
Cycle 2	94.5 ± 0.1	21.0 ± 0.0	72.8 ± 0.5	24.2 ± 2.7	10.1 ± 3.6	54.3 ± 0.5
Cycle 3	95.5 ± 0.1	55.5 ± 0.2	85.3 ± 0.4	18.9 ± 32.7	42.4 ± 1.5	27.0 ± 0.1

Mofokeng et al., (2022) in their study found that nitrates removal ineffectiveness may be attributed to operating temperatures (minimum average range of 10.5-13.5°C) that were not optimal for the activity of the microbial communities driving the treatment process in a Sludge Process Reduced Activated Sludge (SPRAS) plant. Thus, climatic conditions may influence treatment efficiency with the efficiency reducing when air temperatures are below optimal temperatures for growth of microbial communities.

3.3.2.2 Treatment performance against standards

Nutrients, especially N and P which are discharged into aquatic environments cause eutrophication, which in turn lead to high accumulation of dead biomass and depletion of oxygen in water bodies (Schellenberg et al., 2020). While nutrients are beneficial and required for plant growth, they can cause water contamination if found in excessive amounts and in areas with low groundwater table (Schellenberg et al., 2020). The total N, P and K characteristics of the final treated effluent leaving the maturation pond over the different treatment cycles is described and compared with various discharge standards in Table 3-6. The final concentrations did not meet any of the minimum discharge standards for release into receiving waters in Table 3-6. In cycle 1 (C1) for instance, the N, P and K concentrations were 51.4, 15.5 and 133.0 mg/L respectively. These concentrations were higher suggesting the potential of causing eutrophication in nearby water bodies downstream if released. Fortunately, this may not be the case as this FS treatment plant was designed to reuse the treated effluent internally for crop irrigation, hence promoting circular economy. In a similar study by Abagale *et al.,* (2020) of FS stabilisation ponds in Ghana found that the concentrations of various forms of N (NH_3, NO_3^- and NO_2^-) were also higher than the allowable limit of Ghana EPA.

Table 3-6. Comparison of treated effluent of FS and FW treatment system to discharge guidelines.

Nutrient (mg/L)	**Treatment cycles**			**Ghana EPA**	**EU**	**Uganda**	**USEPA**
	Cycle 1	Cycle 2	Cycle 3				
Total N	51.4 ± 1.6	65.0 ± 1.0	37.8 ± 0.7	-	15.0[a] 10.0[b]	-	3.0 – 5.0
Total P	15.5 ± 0.8	15.8 ± 0.1	10.5 ± 0.5	2.0	2.0[a] 1.0[b]	10.0	1.0
Total K	133.0 ± 20.2	128.1 ± 0.9	168.0 ± 5.0	-	-	-	-

Source: (Ghana EPA, 2000); Preisner *et al.* (2020), a = 10,000 – 100,000 PE; b = <100,000 PE

3.3.3 Mass balance of total N, P and K in the FS and FW Treatment

3.3.3.1 Total NPK mass distribution in Cycle 1

Quantifying nutrients in terms of their recovery and losses from end-to-end FS treatment is crucial to optimizing nutrient recovery efficiency along the treatment chain. Nutrients also need to be managed to minimize the eutrophication potential on surface water resources. Prior to the treatment, the main input sources of total N, P and K were from the FS and FW. Nutrient inputs from rainwater were considered as insignificant. The total N, P, and K received from the FW for cycle 1 was 76.17, 102.97 and 56.67 Kg, and for FS was 194.22, 139.94 and 208.85Kg, respectively (Table 3-7). However, results after the FS dewatering (primary treatment) showed a cumulative total N mass of 60.22kg and 80.00kg in the percolate and dewatered FS (DFS), respectively. This corresponded to an average recovery of 41.2% of total N in the DFS and an average loss of 58.8% due to the percolate (31.0%), losses attributed to gaseous escape and other unaccountable losses like adsorption to media surfaces (27.8%) (Figure 3-3). This reveals that, more than half of the total N received from the FS is lost in the system around the FS dewatering stage either as percolate, gaseous losses or adsorption to the filter media surfaces. Manga et al., (2016) observed similar losses of total N attributed to the higher NH_3-N volatilisation, which was due to longer dewatering times and extended exposure of dewatering FS to high ambient temperatures.

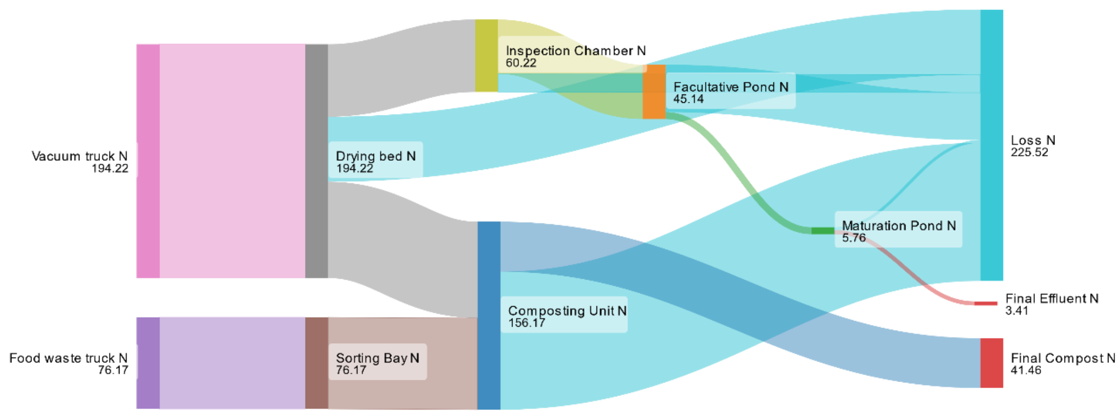
Table 3-7. Total N, P and K distribution in the different stages of FS and FW treatment system during cycle 1 (Kg).

Nutrient	**Input (kg)**		**FS dewatering (kg)**			**Co-composting (kg)**			**Effluent treatment (kg)**		
	FS	**FW**	**DFS**	**Percolate**	**loss**	**Initial**	**Final**	**loss**	**Influent**	**facultative**	**Maturation**
Total N	194.22	76.17	80.00	60.22	54.00	156.17	41.62	114.71	45.14	5.76	3.41
Total P	139.94	102.97	94.18	4.23	41.53	197.15	60.99	136.16	4.14	1.19	1.03
Total K	208.85	56.67	45.22	42.09	121.54	101.89	53.56	48.33	40.43	10.63	8.83

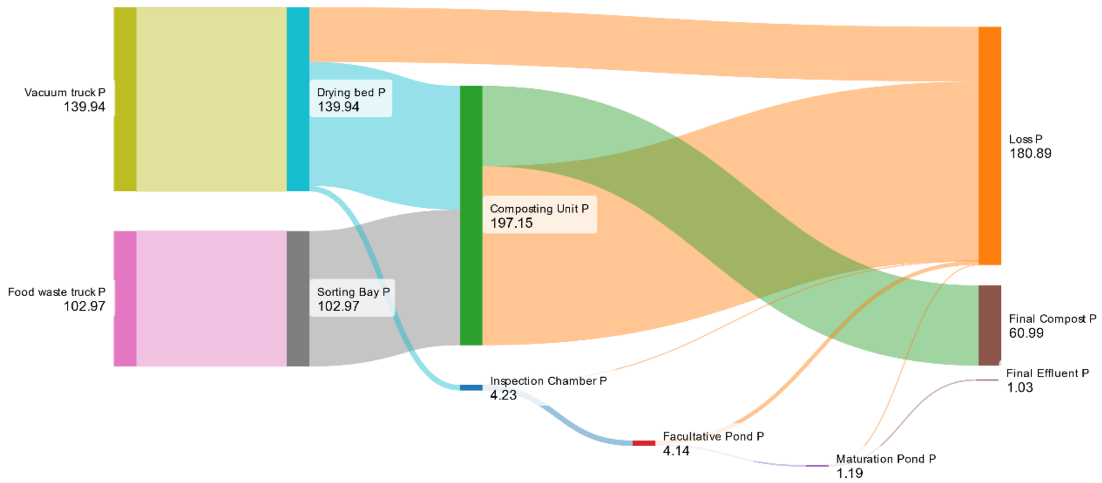
Similar high losses for total P and K were observed in this study. Sand drying beds (unplanted sludge drying beds) have proven to be technically feasible FS treatment technology with the recovery of nutrient and biosolids for agriculture reuse. However, the current design and operational criteria have been associated with some limitations such as generation of low quality dewatered solids in terms of NPK and organic matter; percolate with high contaminant loads; longer dewatering periods and high required footprint (Heinss et al., 1998; Cofie *et al.*, 2006; Manga *et al.*, 2016). Previous studies have addressed the longer dewatering periods (Gold et al., 2016; Seck et al., 2015) but not so much on the quality of the solids and the very high percolate contamination load (Manga et al., 2016).

There were further losses of total N (25.0%) in the percolate as it mixed with leachate in channel before entering the facultative pond for the final polishing (Secondary treatment). The final amount of total N leaving the treatment system as treated effluent is 3.41Kg which could be used for irrigation. A total of 41.73Kg of N had been treated from the effluent by the collective actions of the facultative and maturation ponds. The 80.00Kg total N recovered in the DFS was co-composted with FW as secondary treatment. The initial co-composts had total 156.17kg N to begin with and the final co-composts had 41.62Kg N indicating a loss of 114.71Kg (73.5%). More than half of the N is lost during the co-composting process (Figure 3-3).

The total P transformation in the treatment system showed that 139.94Kg and 102.97Kg was inputted by the FS and FW respectively (Table 3-7). About 94.18Kg of total P was recovered in the DFS during FS dewatering with an amount of 45.76Kg, lost constituting 32.70% of loss. A total of 1.03Kg P was retained in the treated effluent leaving the maturation pond (Figure 3-3). The total K input during cycle 1 treatment was 265.47Kg (208.80Kg DFS + 56.67Kg FW) and this was distributed along the treatment chain as shown in Table 3-7. The total amount of K leaving the treatment system is 53.56Kg in co-compost and 8.83Kg in the effluent leaving the maturation pond. The transformation of total N, P and K for cycle 1 treatment is shown in Figure 3-3.



(a)



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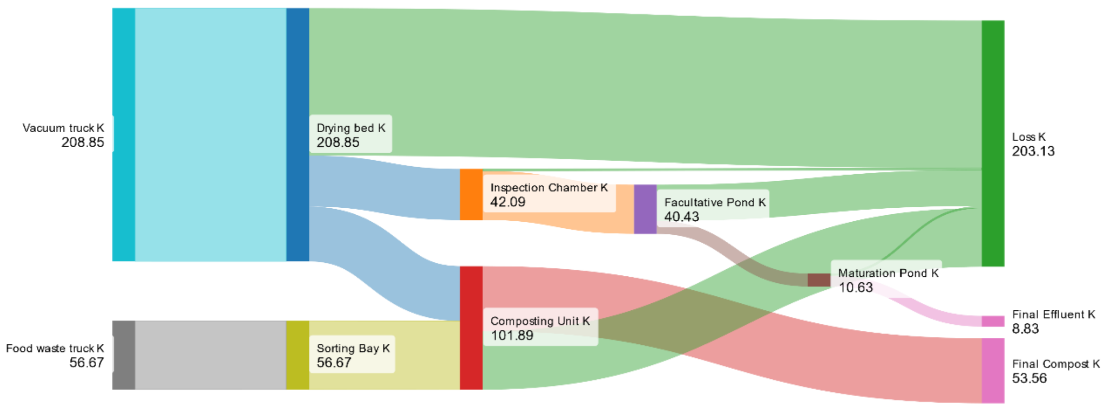


(c)

Figure 3-3. Total N (a), P (b) and K (c) distribution in the different stages of FS and FW treatment system during cycle 1 (Kg dry weight).

3.3.3.2 Total NPK mass distribution in Cycle 2

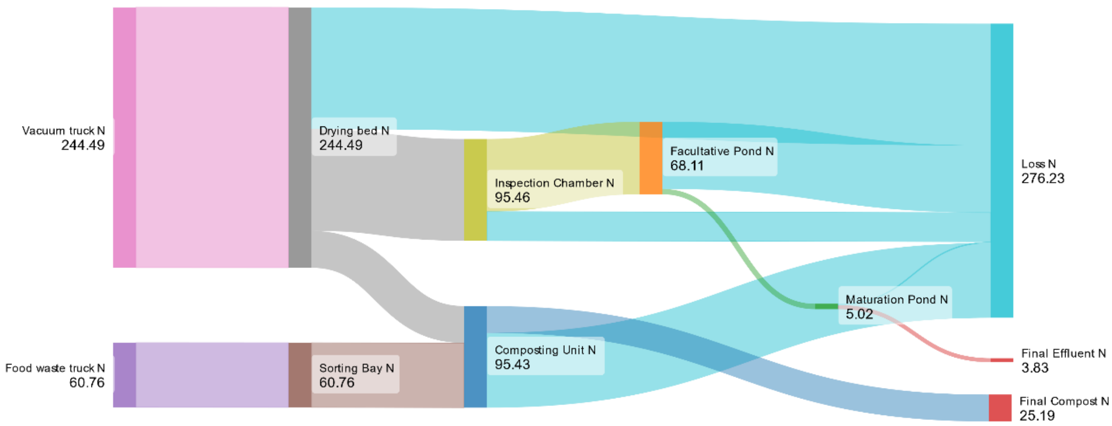
In cycle 2, the total input of total N, P, and K received from FS and the source separated FW into the treatment system was 305.25, 122.43 and 214.88kg, respectively in an approximate ratio of 2.5:1.0:1.8 (Table 3-8). The total output of N, P and K recovered in the form of co-compost and treated effluent exiting the treatment system was 29.02, 44.89 and 37.58kg, respectively in an approximate ratio of

1.0:1.5:1.3. There were significant changes in the amounts of nutrients throughout the treatment system. The total N received from the FS was 244.49kg. After going through the dewatering process, only 34.67kg (14.18%) out of 244.49kg was retained in the DFS to be used for the co-composting (Table 3-8). This indicates a much greater loss for N during dewatering in cycle 2 compared with cycle 1 (Figure 3-4). This was probably due to NH_3-N volatilisation from longer dewatering times. Total P retained in the DFS was 60.75% of the 65.73Kg from the FS and 39.25% counted as loses.

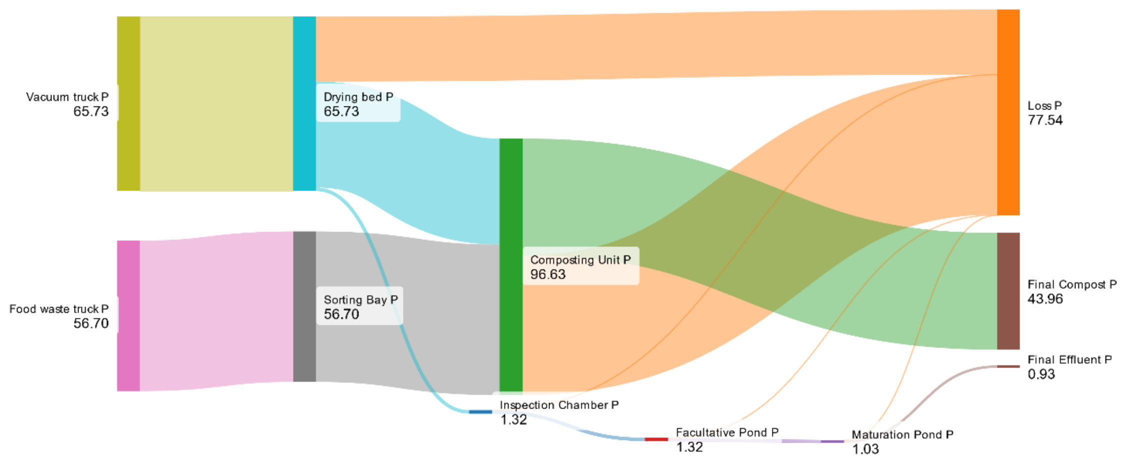
Table 3-8. Total N, P and K distribution in the different stages of FS and FW treatment system during cycle 2 (Kg).

Nutrient	**Input (Kg)**		**FS dewatering (Kg)**			**Co-composting (Kg)**			**Effluent treatment (kg)**		
	FS	**FW**	**DFS**	**Percolate**	**Loss**	**Initial**	**Final**	**loss**	**Influent**	**facultative**	**Maturation**
Total N	244.49	60.76	34.67	95.46	114.36	95.43	25.19	70.24	68.11	5.02	3.83
Total P	65.73	56.70	39.93	1.32	24.48	96.63	43.96	52.67	1.32	1.03	0.93
Total K	153.45	61.43	30.35	61.25	61.85	91.78	30.04	61.74	54.68	16.50	7.54

The mass transformation of N, P and K for cycle 2 treatment is shown in Figure 3-4.



(a)



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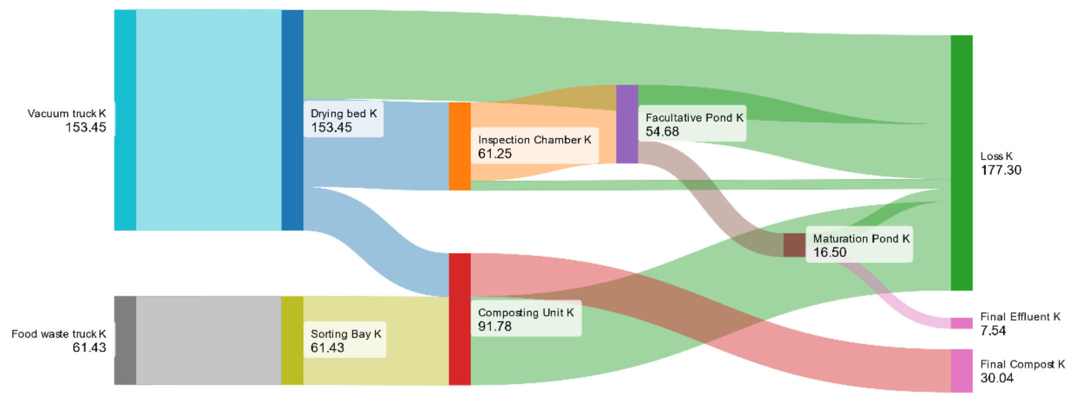


(c)

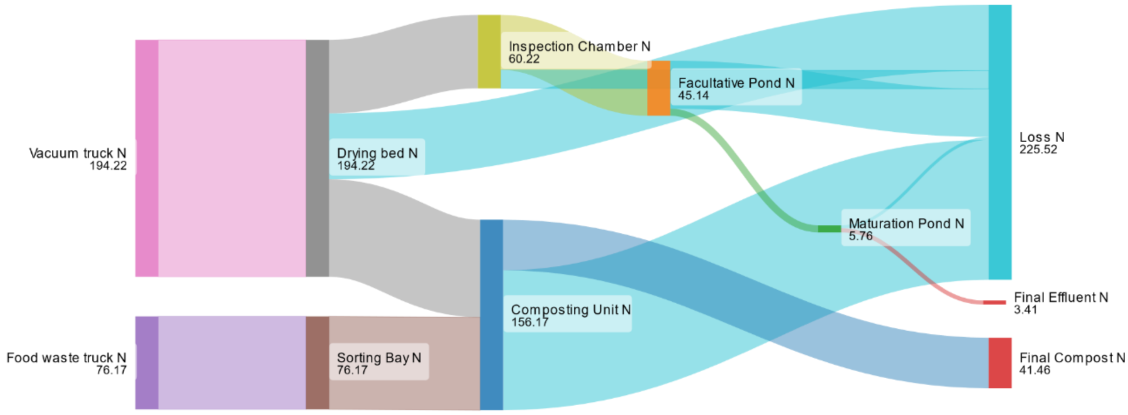
Figure 3-4. Total N (a), P (b) and K (c) distribution in the different stages of FS and FW treatment system during cycle 2 (Kg dry weight)

3.3.3.3 Total NPK mass distribution in Cycle 3

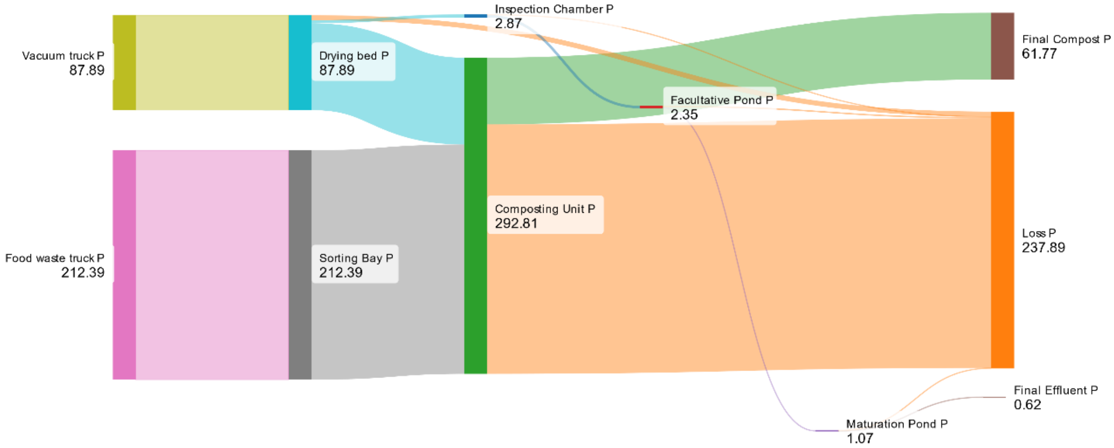
For cycle 3 and similar to the previous two cycles, the amount of total N losses around the FS dewatering processes was high (85.00%). The percentage loss was higher than in cycles 1 (58.80%) but similar to cycle 2 (85.81%) (Table 3-9). The mass transformation of N, P and K for cycle 3 treatment is shown in Figure 3-5.

Table 3-9. Total N, P and K distribution in the different stages of FS and FW treatment system during cycle 3 (Kg).

Nutrient	Input (Kg)		FS dewatering (Kg)			Co-composting (Kg)			Effluent treatment (kg)		
	FS	FW	DFS	Percolate	Loss	Initial	Final	loss	Influent	facultative	Maturation
Total N	879.77	186.74	132.31	77.46	670.00	319.05	66.03	253.02	74.53	5.10	2.22
Total P	87.89	212.39	80.42	2.87	4.60	292.81	61.77	231.04	2.35	1.07	0.62
Total K	242.04	214.72	123.43	74.40	44.21	338.15	95.29	242.86	70.83	13.52	9.88



(a)



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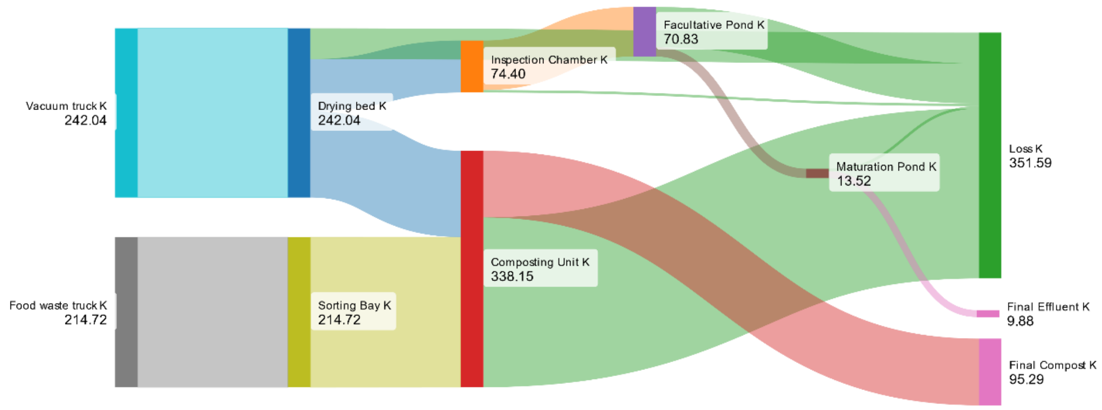

(c)

Figure 3-5. Total N (a), P (b) and K (c) distribution in the different stages of FS and FW treatment system during cycle 3 (Kg dry weight)

There is limited published data on nutrient dynamics focusing total N, P and K to compare these FS treatment plant (FSTP) results with, as few studies have been conducted on faecal sludge treatment from end to end (Levira et al., 2023). The results of this study can provide information on major nutrient dynamics in a FSTP to build future predictive models on nutrient recoveries and losses in FS treatment in SSA. This is not just from a point of view of protecting adjacent waterways from pollution but also give a better understanding and characterisation of the nutrient losses that can be minimised to the maintain the nutrients in flow. Presently, there is very limited information on the removal efficiencies of FSTP operating in lower and lower-middle income countries (Geetha Varma et al., 2022). FSTP are designed to treat FS which has a much higher total solids (TS) concentration compared to sewered domestic wastewater(Strande et al., 2014). The higher solids density of FS corresponds with increased nutrients and microbial loads relative to sewerage (Strande et al., 2014). Hence, FSTP removal efficiencies and effluent quality cannot be directly compared to wastewater treatment plants that treat sewerage.

3.4 Conclusions and recommendations

Based on the results presented, the study has successfully characterised the total N, P and K dynamics in the decentralised FS and FW treatment plant. It can be concluded that, the treatment efficiency of the various components of the treatment system is variable and one of the major factors is the variable characteristics of the FS. Significant losses of total N, P and K were observed around the dewatering stages. More than 50% of total N is lost after dewatering stages which suggests dewatering using sand drying beds may not be effective for nutrient recovery. Final effluent characteristic was higher than the minimum discharge guidelines for many countries, however it provides a great resource for reuse in crop irrigation. The removal efficiency of FSTP cannot be directly compared with wastewater treatment plants.

It is recommended that, the real time data and characterisation of nutrient dynamics obtained in this study can support the building future predictive models on nutrient recoveries and losses in FS treatment in SSA.

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Chapter 4 Assessing the effect of the decentralised faecal sludge treatment process on *E. coli* inactivation.

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Abstract
The objective of this study was to assess *E. coli* deactivation and prevalence of antimicrobial resistance in the end-to-end faecal sludge treatment system. The study was carried out at field scale at the Jekora Ventures Limited (JVL) – Yilo Krobo Municipal Assembly (YKMA) Recycling Plant, in Somanya Ghana. Results show that the average *E. coli* and total coliform levels in the FS ranged from 2.6 x 10^5 – 1.5 x 10^6 and 4.1 x 10^5 – 4.7 x 10^6 CFU/100ml, respectively for the three cycles. The inactivation efficiency of the dewatering process saw *E. coli* removal to be minimal (0.0 – 14.3%) in the percolate. The DFS on the other hand observed higher inactivation efficiencies of 88.2 – 98.0% (1 – 2 log reductions) for *E. coli* across the three cycles and 0.0 – 90.3% (up to 2 log reductions) for total coliform after dewatering. The retention in the maturation pond resulted in complete inactivation (100.0%) of *E. coli* during cycles 2 and 3 while cycle 1 observed 98.6% inactivation efficiency of *E. coli* (2 log reductions). The process significantly reduced *E. coli* levels as 100% pathogen inactivation was attained. As such, the *E. coli* levels observed in Cycles 2 and 3 were below the maximum limits set by the Ghana EPA and EU. mass balance estimations around the FS dewatering reveal that between 0.1 – 0.3% of all incoming *E. coli* is retained in the DFS per cycle. A range of 25 – 55% of the *E. coli* is retained in the percolate moving on to the effluent polishing in the facultative and maturation ponds. At the end of each treatment cycle, it is estimated that a total of 0.4 – 2.9 million CFU of *E. coli* from a population of 122,705 is treated and diverted out of pollution pathway.

Keywords: *E. coli; Inactivation; Treatment efficiency; Faecal sludge; Co-compost*

4.1 Introduction

In the Global South, there are growing examples of successful decentralized treatment systems that treat faecal sludge (FS) from on-site sanitation systems (OSS) (Semiyaga et al., 2015). Examples of decentralised FS treatment include co-composting, vermicomposting, conversion to animal protein via black soldier flies, pyrolysis, and larger scale natural systems such as waste stabilisation ponds and constructed wetlands (Semiyaga et al., 2015). FS contains nutrients required for plant growth, in addition to organic matter, and could thus be used as a fertilizer or soil conditioner. However, it also contains pathogen that can lead to occupational risks and to an increased risk of food- and feed-borne disease transmission if not managed properly (Jimenez et al., 2006; Trang et al., 2007). The type of pathogens mostly found in FS depends on the state of health of the population, as well as on the OSS containment practices. But the most commonly used indicator organisms are the *Escherichia coli* O157:H7, *Salmonella sp*. (Guan & Holley, 2003) and Helminths (Berendes et al., 2015; Koné et al., 2007). *E. coli*, *Shigella, Salmonella* and *Vibrio spp*. have been the predominant pathogens linked to waterborne outbreaks in Africa (Bessong *et al.*, 2009; Olaniran *et al.*, 2011), whereas certain clonal strains of these pathogens have been reported to survive conventional wastewater treatment processes (Adefisoye & Okoh, 2016; Anastasi et al., 2012; Cañigral et al., 2010).

Aerobic co-composting can stabilize organic matter in the matured/stable compost as well as minimize the adverse effects of indigenous bacteria and the fungi in FS (Awasthi et al., 2017; Duan, Awasthi, Liu, Chen, et al., 2019). Hence treatment plays a critical role, particularly where FS is likely to be used for crop cultivation and food production. An understanding of the effectiveness of treatment options on pathogens is critical for the assessment of appropriate management strategies (Manga et al., 2021). Pathogen survival or regrowth is a problem for certain bacteria such as *Salmonella spp*. and *E. coli*, especially where do not require a host organism for reproduction, unlike some other pathogen species like viruses, protozoa and helminths (Haug, 1993; Wichuk & McCartney, 2007). Poorly treated wastewater or FS from on-site and decentralised treatment plants often infiltrates into groundwater or surface water, either intentionally or inadvertently (Headley et al., 2013) or end up on land when used as soil amendment. The microbial safety of organic amendments and fertilizers used in agriculture is required to prevent colonization by food-borne pathogens, like *E. coli O157:H7* and *Salmonella spp*. (Lemunier et al., 2005). The study of these pathogen behavior and reduction profiles in a FS treatment chain in sub – Saharan Africa (SSA) context is thus critical to minimize or eliminate public health risks to humans and the rest of the environment. And with the ongoing COVID-19 pandemic, traces of the SARS-CoV-2 genetic material have been found in wastewater and faecal material of infected populations (Balboa et al., 2021; Chavarria-Miró et al., 2021). There is very little or no information on the effect of the treatment process on pathogen reduction profile along the treatment stages in the decentralised treatment system. Hence there is a dearth of information on the end-to-end performance of decentralised treatment systems for *E. coli* reduction efficiency.

Unfortunately, adequate FS treatment is not a common practice in urban SSA where sanitation service delivery in the form of sustainable treatment facilities is still greatly lacking (Manga *et al.*, 2020). Consequently, untreated FS is often reused or indiscriminately disposed of into the environment via canals, open drains, and surface water bodies. This results in environmental damage as well as serious public health risks, leading to high occurrences of excreta related diseases, and hence high morbidity and mortality (Peal et al., 2014). The demand for infrastructure to manage FS is increasing, and reliable methods are being developed and validated to estimate total accumulated quantities and qualities of FS to support management and treatment technology solutions (Englund et al., 2020; Strande et al., 2018). The socio-cultural and scientific perspectives on treated FS reuse in agriculture are now being revisited with the development of new technologies which facilitate safe and sanitary repurposing of FS (Grant *et al.*, 2012; López-Rayo *et al.*, 2016; Tobias *et al.*, 2017; Simha *et al.*, 2018). One of such decentralised systems is the FS treatment system comprising of FS dewatering on sand drying beds and coupled with aerobic (open air) co-composting with municipal solid waste (Nikiema et al., 2013). This FS treatment system, which provides a win-win situation for sustainable sanitation and agriculture, has not been thoroughly explored especially in urban Africa for its effectiveness to sanitize pathogens.

There is little evidence on the effectiveness of the holistic FS treatment system on pathogen reduction in tropical climate in literature. The study hypothesizes that the treatment process will reduce *E. coli* loads and suppress its antimicrobial resistivity. Therefore, this study aimed to conduct an end-to-end assessment of the effect of treatment on the inactivation of *E. coli* present in FS. The specific objective was to assess *E. coli* inactivation in the end-to-end faecal sludge treatment system.

4.2. Materials and methods

4.2.1 Study location description

The study was carried out at field scale at Akorley, Somanya (latitude 6° 6' 0" N and 3° 3' 0" N and between longitude 0° 0' 30" W and 0° 0' 10" W, Sadiq, 2016) at the Jekora Ventures Limited (JVL) – Yilo Krobo Municipal Assembly (YKMA) Recycling Plant, in the YKMA of Ghana (Figure 4-1). The annual rainfall of the area ranges from 750 to 1,600 mm and it's spans from May to October (bimodal). Average temperatures range between 24 and 30°C while relative humidity ranges between 60 and 90% (Sadiq, 2016). The major soil type is savanna ochrosol (Eastern Regional Co-ord Council, 2016). It has low nutrient reserves, with the topsoil consisting of dark greyish brown humus sandy or clay loams (Eastern Regional Co-ord Council, 2016). For the entire municipality, the potential/estimated FS generation is 15,000 m^3 per year from major on-site sanitation systems (OSS) (Nikiema et al., 2016). The main productive activity in YKMA is agriculture (e.g., mango plantations, food crop farming and livestock rearing), which employs around 60% of the population.

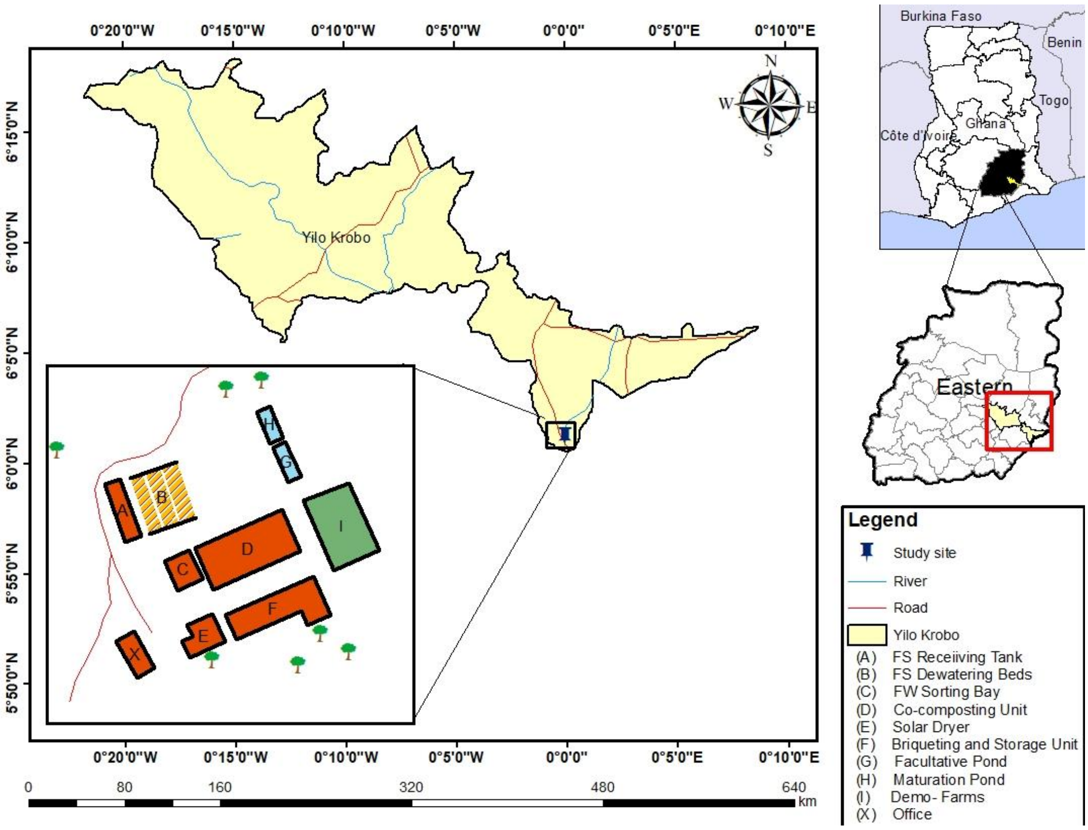

Figure 4-1. Map of study location showing study site

2.1.1 Study site description (operational conditions)

The study site was the FS and FW co-composting system (Figure 4-2) of the JVL - YKMA Recycling Plant. The treatment system treats both FS and uncooked FW into faecal derived fertiliser. The units include FS dewatering on sand drying beds, effluent and/or leachate treatment, and then co-composting and storage. The treatment chain is described in the following steps:

i) Receiving of FS from vacuum trucks from either pit latrines or septic tanks and dewatering on sand drying beds. Receiving source separated uncooked FW from local markets.
j) Removal of DFS from drying beds and co-composting with source separated FW at a ratio of 1:3 w/w.
k) Treatment of effluent in facultative and maturation ponds for reuse of irrigation.
l) Storage of mature co-compost.

4.2.2 Sampling approach

4.2.2.1 Field sampling plan and sample collection

The sampling plan was developed following the treatment process from start to end (end -to -end) (Figure 2). The points at which the material transformed or was expected to transform was delineated as sampling points (SP) and the justification provided as follows:

a) Sampling FS from trucks - SP 1

FS emptied from pit latrines and septic tanks by vacuum trucks and brought to the treatment plant (SP 1). SP 1 was the point at which all FS entered the treatment system. To characterise the FS that entered the treatment system, each truck was sampled at the point of discharging the FS onto the drying beds. Samples were collected directly from each truck following sampling methods described by Klingel et al. (2002) and Bassan et al. (2013). Equal volume of sludge was collected during the following points of discharge:

(i) The beginning (valves opened)

(ii) Middle of discharge

(iii) Ending of discharge when pressure is low.

The samples were mixed/bulked together, and a composite sample taken. All the samples from all the trucks that fed the drying bed were again bulked together and a composite sample taken in triplicates to the labs for analyses.

b) Sampling effluent from dewatering beds - SP 2

On the drying bed, the sludge underwent solid – liquid separation where free water filtered through the media and the dewatered FS (DFS) remained on the sand layer. The effluent, which was accessible in the inspection chamber (SP 2), was sampled at the same time every day from the start of loading until day 5 when active filtration was done (when the 50% volume of sludge is dewatered). The samples collected after the 5 days were bulked together and a composite sample taken and analyzed in triplicate.

c) Sampling leachate from co-composting platform - SP 3

Leachate production is not regular on the composting platform. As and when it was produced, it was channeled through the inspection chamber (SP3). The leachate at this point was collected on a daily basis and bulked together to form a composite sample, which was analyzed in triplicate.

d) Sampling influent to facultative pond - SP 4

The effluent from the dewatering process (SP2) and the leachate from the composting platform (SP3) meets and mixes at this point before going into the facultative pond. This point (SP 4) became the influent of the facultative pond. The influent was sampled at the time the drying beds and composting platforms were actively being used. The sampling procedure followed the same description for SP 2.

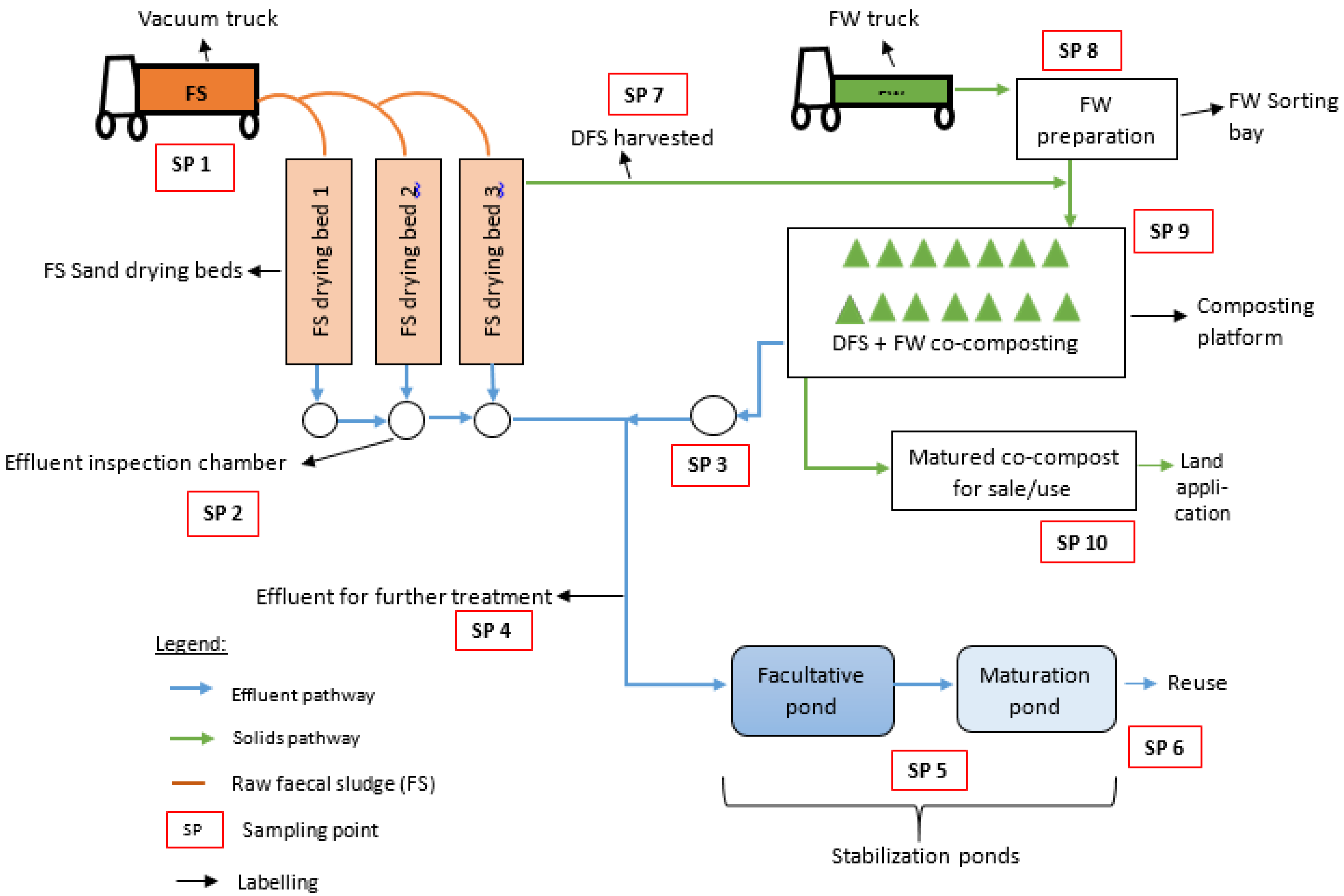

Figure 4-2. Schematic of the decentralised FS and FW treatment system

e) Sampling effluent from facultative pond - SP 5

The treated effluent leaving the facultative pond became the influent to the maturation pond (SP 5). The effluent from the facultative pond was collected from four points and bulked into one sample. This was collected once a week.

f) Sampling effluent from maturation pond - SP 6

The treated effluent leaves the plant ideally at this point. From this point, all pollutants including nutrients may been recovered and/or reduced in the effluent to meet discharge guidelines. The effluent from the facultative pond was collected from four points and bulked into one sample. This was collected once a week.

g) Sampling of DFS from drying bed - SP 7

After the solid-liquid separation of the FS at SP 1, the harvested DFS (SP 7) is what goes into the composting process. Grab samples of DFS was taken from five points along the diagonal lines of the drying bed. These grab samples were bulked together to form a composite sample from which representative samples were taken for analyses.

h) Sampling of uncooked FW - SP 8

The source-separated FW arrives at the sorting bay (SP 8) of the treatment plant as the second feedstock to be used for the co-treatment of FS. As such, several grab samples of FW were taken and bulked together to form a composite sample from which samples were taken to the lab for analysis.

i) Sampling at the start of FS and FW co-composting - SP 9

This is the point where the DFS and FW were mixed in a known ratio to begin the composting process (SP 9) in Figure 2. Samples of the heap were taken at the initial mixing of the feedstock to form the heap. Grab samples at different points of the mix were collected and bulked together to form a composite sample, which was analysed in triplicate.

j) Sampling of matured faecal derived fertiliser - SP 10

A composite sample of the matured heap following the bulking up of several grab samples was taken and analysed in triplicate.

4.2.2.2 Duration and frequency of sampling

In this study, a phase (P) is defined as one complete FS dewatering operation per drying bed while a cycle (C) is defined as one complete treatment of FS received at the treatment plant until maturation pond is full. The duration for a complete P is on average 3 – 4 weeks while one complete C of treatment on average is from 4 – 6 months. The cycle was repeated three times.

4.2.2.3 Parameters for analyses

E. coli, and antimicrobial sensitivity of *E. coli*.

4.3 Results and Discussions

4.3.1 FS loading of the treatment system

This decentralised plant serves a population of 122,705 according to the 2021 census in the Yilo Krobo municipality (Ghana Statistical Service, 2021). This population consists of 36,371 households that are served by private vacuum truck operators (Ghana Statistical Service, 2021). In this study, the assessment of the plant lasted between August 2021 – June 2022 for three cycles of operations. FS received was based on household demand for vacuum trucks and the plant's readiness to receive sludge. Table 4-1 shows the number of trucks received at the plant during the assessment and the corresponding volumes of FS discharged. The number of trucks received at the plant did not directly correspond to volumes of FS discharged. This was due to the different sizes of the vacuum trucks used by the private operators. On average, 9 trucks were received per drying bed for Cycle 1, 5 trucks for Cycle 2 and 6 trucks for Cycle 3. Cycle 1's Phase 1, 2, 3, and 4 recorded 55m^3, 52m^3, 45m^3, and 50m^3, respectively (Table 4-1). A total of 200, 155 and 128m^3 of FS were received and treated for Cycles 1, 2 and 3, respectively (Table 4-2).

Table 4-1. Volumes of FS received and processed during the various phases and cycles of treatment.

Treatment Cycle	**Phase**	**No. of FS trucks**	**Vol. of FS (SP1) (m^3)**	**Vol. of percolate after dewatering (SP2) (m^3)**	**Vol. of final effluent (SP6) (m^3)**	**Fresh weight of DFS (SP7) (Kg)**	**No. of compost heaps**
Cycle 1	Phase 1	10	53	25.7	84.7	2927.4	1
	Phase 2	9	52	8.0		1962.7	1
	Phase 3	10	45	28.6		1400.9	1
	Phase 4	6	50	22.4		1693.9	1
Cycle 2	Phase 5	7	35	16.3	61.9	1075.1	1
	Phase 6	6	44	24.2		1383.8	1
	Phase 7	4	34	7.3		1097.0	1
	Phase 8	4	42	14.2		442.7	1
Cycle 3	Phase 9	7	54	23.3	61.8	2206.0	1
	Phase 10	5	25	10.8		2545.6	1
	Phase 11	6	31	20.7		3080.0	1
	Phase 12	7	39	7.0		5930.0	1

In each of the cycles, the study found that the average properties of FS varied in volume and texture by physical appearance. This ultimately affected the desludging and dewatering times as reflected in the period of desludging measured in Table 4-2. Sources of FS for this study were mostly from homes (predominantly using septic tanks) and public toilets (using pit latrines). This corroborates previous findings that, septic tanks and ventilated improved pits (VIP) are the two types of on-site sanitation systems (OSS) that are frequently used in towns and peri-urban communities in Ghana (Fanyin-Martin et al., 2017; Nayebare et al., 2020). FS properties may vary depending on where it is produced, how it is generated, the seasonal variation, the type of OSS in place etc. (Yaser et al., 2022). The public toilets use little flush water in contrast to septic tanks, which employ flush systems like water closets that require water for their operation (Oppiah-Mensah, 2019). As a result, FS from the public toilets were usually more fresh, thick, and concentrated (high strength) while the sludge from the septic tanks were more stable, watery and ore diluted (low strength).

While the type of OSS may have contributed to the properties of the FS received, climatic conditions may have contributed as well. The climate of the area is in two seasons, rainy and dry season. The rainy season is bimodal with the major season occurring from March – July and minor season occurring from September – November. The dry season is characterised by cold dry weather occurring from November – February (Ministry of Food & Agriculture, 2023). In this study, cycles 1, 2 and 3 coincided with the minor rainy season, dry season, and major rainy season, respectively. The climatic effect on OSS is such that, septic tanks take relatively shorter time than usual to fill during the wet or rainy seasons (between the months of April and July), leading to more frequent desludging following the season.

Table 4-2. FS loading time and volume during the various phases and cycles of treatment

Treatment Cycle	Phase	Desludging period (days)	Volume (m^3)	Total volume (m^3)
Cycle 1	Phase 1	21 – 28 August 2021 (7 days)	55.0	200.0
	Phase 2	22 – 28 September 2021 (6 days)	52.0	
	Phase 3	29 September – 7 October 2021 (8 days)	45.0	
	Phase 4	14 – 16 October 2021 (2 days)	50.0	
Cycle 2	Phase 5	23 – 26 October 2021 (3 days)	35.0	155.0
	Phase 6	20 November – 22 December 2021 (32 days)	44.0	
	Phase 7	31 December 2021 – 15 January 2022 (14 days)	34.0	
	Phase 8	12 – 14 February 2022 (2 days)	42.0	
Cycle 3	Phase 9	11 March – 20 April 2022 (40 days)	54.0	128.0
	Phase 10	2 – 6 May 2022 (4 days)	25.0	
	Phase 11	8 – 22 May 2022 (14 days)	22.0	
	Phase 12	3 – 13 June 2022 (10 days)	27.0	

4.3.2 Pathogen inactivation efficiency in the FS and FW treatment system

4.3.2.1 Treatment efficiency of the treatment stages

The levels of *E. coli* and total coliform in the FS received at the decentralised treatment is shown in Table 4-3. The average *E. coli* and total coliform levels in the FS ranged from 2.6 x 10^5 – 1.5 x 10^6 and 4.1 x 10^5 – 4.7 x 10^6 CFU/100ml, respectively for the three cycles. In the treatment chain, the FS undergoes solid – liquid separation on sand drying beds, a process of dewatering to make the solids available for co-composting. The pathogen inactivation efficiency of the dewatering process saw *E. coli* removal to be minimal in the percolate ranging from 0.0 – 14.3%. The average concentration of *E. coli* in the percolate from cycles 1, 2 and 3 were 1.5 x 10^6, 2.4 x 10^5, and 2.4 x 10^5 CFU/100ml, respectively (Table 4-3). The DFS on the other hand observed higher inactivation efficiencies of 88.2 – 98.0% (1 – 2 log reductions) for *E. coli* across the three cycles and 0.0 – 90.3% (up to 2 log reductions) for total coliform after dewatering. The mechanism behind the higher inactivation observed in the DFS was probably due to pathogen desiccation, following loss of percolate (moisture) during the dewatering process by evapotranspiration (Stefanakis & Tsihrintzis, 2011). In addition to loss of moisture, solar radiation may have also played a role of reducing pathogen numbers (Obianyo, 2015). Similar inactivation efficiency of 85% for *E. coli* in DFS was reported by Mrimi et al., (2020).

Table 4-3. *E. coli* and total coliform inactivation efficiency during FS dewatering on sand drying bed.

Treatment Cycle	**FS (CFU/100ml)**		**Percolate after dewatering (CFU/100ml)**		**Dewatered FS (DFS) (CFU/100g)**		**Inactivation efficiency in percolate (%)**		**Inactivation efficiency in DFS (%)**	
	E. coli	Total coliform	*E. coli*	Total coliform	*E. coli*	Total coliform	*E. coli*	Total coliform	*E. coli*	Total coliform
Cycle 1	$1.5 \times 10^6 \pm 6.0 \times 10^5$	$4.7 \times 10^6 \pm 5.2 \times 10^6$	$1.5 \times 10^6 \pm 4.2 \times 10^5$	$1.8 \times 10^6 \pm 1.0 \times 10^6$	$2.9 \times 10^4 \pm 2.1 \times 10^4$	$1.6 \times 10^6 \pm 1.9 \times 10^6$	0.0	61.7	98.0	65.9
Cycle 2	$2.6 \times 10^5 \pm 9.9 \times 10^4$	$4.1 \times 10^5 \pm 1.6 \times 10^5$	$2.4 \times 10^5 \pm 1.0 \times 10^5$	$8.7 \times 10^5 \pm 2.3 \times 10^5$	$3.0 \times 10^4 \pm 1.6 \times 10^4$	$4.1 \times 10^5 \pm 2.1 \times 10^5$	7.7	-112.2	88.5	0.0
Cycle 3	$2.8 \times 10^5 \pm 1.1 \times 10^5$	$3.5 \times 10^6 \pm 1.8 \times 10^6$	$2.4 \times 10^5 \pm 7.4 \times 10^4$	$6.7 \times 10^5 \pm 1.4 \times 10^5$	$3.3 \times 10^4 \pm 8.7 \times 10^3$	$3.4 \times 10^5 \pm 3.5 \times 10^5$	14.3	80.9	88.2	90.3

Table 4-4. *E. coli* and total coliform inactivation efficiency of stabilisation ponds in a FS and FW treatment system.

Treatment Cycle	**Influent (CFU/100ml)**		**Effluent after facultative pond (CFU/100ml)**		**Effluent after maturation pond (CFU/100ml)**		**Inactivation efficiency of facultative (%)**		**Inactivation efficiency of maturation (%)**	
	E. coli	Total coliform	*E. coli*	Total coliform	*E. coli*	Total coliform	*E. coli*	Total coliform	*E. coli*	Total coliform
Cycle 1	$1.5 \times 10^6 \pm 0.0$	$1.9 \times 10^6 \pm 0.0$	$7.8 \times 10^4 \pm 1.3 \times 10^5$	$6.2 \times 10^6 \pm 4.3 \times 10^6$	$1.1 \times 10^3 \pm 1.9 \times 10^3$	$3.0 \times 10^6 \pm 1.0 \times 10^6$	94.8	-226.3	98.6	51.6
Cycle 2	$2.7 \times 10^5 \pm 0.0$	$8.1 \times 10^5 \pm 0.0$	$6.9 \times 10^4 \pm 6.3 \times 10^4$	$5.0 \times 10^5 \pm 2.9 \times 10^5$	0.0 ± 0.0	$1.9 \times 10^5 \pm 1.5 \times 10^5$	74.4	38.3	100.0	62.0
Cycle 3	$2.6 \times 10^5 \pm 0.0$	$6.3 \times 10^5 \pm 0.0$	$8.2 \times 10^4 \pm 6.2 \times 10^4$	$3.4 \times 10^5 \pm 4.0 \times 10^5$	0.0 ± 0.0	$1.3 \times 10^5 \pm 8.8 \times 10^4$	68.5	46.0	100.0	61.8

Table 4-5. *E. coli* and total coliform inactivation efficiency during composting in a FS and FW treatment system.

Treatment cycle	**Composting at start (CFU/100g)**		**Composting at end (CFU/100g)**		**Inactivation efficiency of composting (%)**	
	E. coli	Total coliform	*E. coli*	Total coliform	*E. coli*	Total coliform
Cycle 1	$2.1 \times 10^9 \pm 2.2 \times 10^9$	$7.5 \times 10^{10} \pm 9.5 \times 10^{10}$	0.0 ± 0.0	$8.1 \times 10^6 \pm 1.2 \times 10^6$	100.0	99.9
Cycle 2	$4.4 \times 10^9 \pm 4.7 \times 10^9$	$3.2 \times 10^8 \pm 3.1 \times 10^8$	0.0 ± 0.0	$1.6 \times 10^7 \pm 2.8 \times 10^7$	100.0	95.0
Cycle 3	$3.1 \times 10^7 \pm 3.4 \times 10^7$	$2.9 \times 10^8 \pm 2.7 \times 10^8$	0.0 ± 0.0	$3.3 \times 10^7 \pm 3.4 \times 10^7$	100.0	88.6

The percolate captured from the drying beds and leachate from the food waste was treated further in the facultative and maturation ponds in Table 4-4. The retention in the maturation pond resulted in complete inactivation (100.0%) of *E. coli* during cycles 2 and 3 while cycle 1 observed 98.6% inactivation efficiency of *E. coli* (2 log reductions). This shows that the maturation pond is a significant treatment step in inactivating faecal indicator organisms (Table 4-4). These results obtained falls in line with the range of efficiencies of maturation ponds studied by Sheludchenko et al., (2016).

E. coli and total coliform inactivation efficiency of the co-composting phase of the treatment is shown in Table 4-5. The process significantly reduced *E. coli* levels as 100% pathogen inactivation was attained during the all the cycles of monitoring. Allowing the end-product i.e., the faecal derived fertiliser exiting the plant to be free of *E. coli.* The total coliforms on the other hand achieved inactivation efficiencies ranging from 88.8 – 99.9% signifying up to 3 log reductions during the co-composting phase of the treatment process (Table 4-5). These results confirm earlier reports of the effectiveness of the co-composting in sanitizing FS in tropical climate (Al-Muyeed et al., 2017; Manga et al., 2021; Nartey et al., 2017).

4.3.2.2 Treatment performance against standards

The average *E. coli* levels in the final effluent of the FS and FW treatment system were 1.1 x 10^3, 0.0, and 0.0 CFU/100ml for cycle 1, 2 and 3, respectively. The *E. coli* levels observed in Cycles 2 and 3 were below the maximum limits set by the Ghana EPA (Ghana EPA, 2000) and EU (Truchado et al., 2021) in Table 4-6. The higher levels of *E. coli* above the guideline limits observed in cycle 1 were probably due to a case of overloading of the system as relatively a higher of FS was treated during cycle 1 (Table 4-2) or contamination from outside sources. However, further studies are recommended to ascertain if there are other factors which might have contributed to the failure of effluent to meet the discharge guideline in cycle 1. The final FDF (co-compost) characteristics described in Table 4-6 after treatment reveals average *E. coli* levels to be zero (0) in all cycles thus meeting the quality guidelines set by the USEPA (USEPA, 1993) and EU (Leifert, 2017). This is also an indication that all faecal pathogens are deactivated during the treatment process.

Table 4-6. *E. coli* and total coliform concentrations in final end-products of FS and FW treatment system in comparison to discharge guidelines.

Pathogen	**End products**	**Treatment cycles**			**Ghana EPA**	**USEPA**	**EU**
		Cycle 1	**Cycle 2**	**Cycle 3**			
Effluent (CFU/100ml)	*E. coli*	1.1 x 10^3	0.0	0.0	10	-	<10
	Total coliform	3.0 x 10^6	1.9 x 10^5	1.3 x 10^5	400	-	-
Co-compost (CFU/g)	*E. coli*	0.0	0.0	0.0	-	<1000	≤1000
	Total coliform	8.1 x 10^4	1.6 x 10^5	3.3 x 10^5	-	<1000	≤1000

Table 4-7. Total CFU of *E. coli* and total coliform calculated from the different stages of FS and FW treatment system for 3 cycles.

Pathogen	**Input (CFU)**		**FS dewatering (CFU)**			**Co-composting (CFU)**			**Percolate treatment (CFU)**		
	FS	**FW**	**DFS**	**Percolate**	**inactivated**	**Start**	**End**	**inactivated**	**Influent**	**facultative**	**Maturation**
						Cycle 1					
E. coli	2.9 Mil	nil	0.0009 Mil	1.6 Mil	1.3 Mil	429 Mil	nil	429 Mil	1.6 Mil	0.08 Mil	0.001 Mil
TC	9.4 Mil	1.7 Mil	0.1 Mil	1.9 Mil	7.3 Mil	27,000 Mil	0.8 Mil	27,000 Mil	1.9 Mil	2.4 Mil	0.6 Mil
						Cycle 2					
E. coli	0.4 Mil	nil	0.0005 Mil	0.2 Mil	0.2 Mil	2.0 Mil	nil	2.0 Mil	0.6 Mil	1.6 Mil	nil
TC	0.6 Mil	0.002 Mil	0.007 Mil	0.6 Mil	0.06 Mil	12.8 Mil	0.2 Mil	12.6 Mil	0.6 Mil	1.1 Mil	0.4 Mil
						Cycle 3					
E. coli	0.4 Mil	nil	0.001 Mil	0.1 Mil	0.3 Mil	4.3 Mil	nil	4.3 Mil	0.0002 Mil	0.04 Mil	nil
TC	4.7 Mil	0.9 Mil	0.01 Mil	0.3 Mil	4.4 Mil	44.0 Mil	2.4 Mil	41.6 Mil	0.3 Mil	0.2 Mil	0.06

Mil = million

4.3.3 *E. coli* and total coliform mass balance in the FS and FW treatment

Pathogen removal from decentralised FS treatment plants is important from a human health perspective, especially in the areas where these decentralised systems are the major treatment facilities available. In this study, between 0.4 – 2.9 million Colony forming units (CFU) per cycle of *E. coli* and 0.6 – 9.4 million CFU of total coliform were received at the treatment plant from the FS (Table 4-7). The FW did not contain any *E. coli* but had total coliform numbers ranging from 0.002 – 1.7 million CFU per cycle. The total coliform numbers in the FS and FW received as inputs in the treatment ranged from 0.60 – 11.1 million CFU per treatment cycle.

E. coli mass balance estimations around the FS dewatering reveal that between 0.1 – 0.3% of all incoming *E. coli* is retained in the DFS per cycle (Table 4-7). A range of 25 – 55% of the *E. coli* is retained in the percolate moving on to the effluent polishing in the facultative and maturation ponds. The remaining mass of *E. coli* is in the range 44 – 75% is considered as colonies treated (inactivated) plus losses in the system. Therefore, the percentage inactivation of E. coli during dewatering is 44 – 75% of the total colonies entering the decentralised system per cycle. This is value is quite significant reduction in the treatment of *E. coli*. The amount of *E. coli* retained in DFS is almost negligible due to unfavorable conditions created by the dewatering process in the DFS that allowed for inactivation.

Around the co-composting stage, the findings reveal an elevated number of colonies at the start of the FS and FW co-composting process (2.0 - 429.0 million CFU) per cycle of treatment (Table 4-7). Since the FW did not have *E. coli* (below detection limit), then it could be concluded that the DFS was the source of *E. coli* in the composting piles at start of composting. The elevated numbers were observed because the *E. coli* colonies surviving in the DFS multiplied under favorable conditions of moisture, nutrients etc. at the start of composting. There was complete inactivation (below detection limit) at the end of the process (Table 4-7). The mechanism for pathogen inactivation during co-composting was explained in Chapter 5a. Similar inactivation of *E. coli* were observed by Nartey et al., (2017) and Manga et al., (2021).

The percolate treatment in the facultative and maturation ponds revealed a 0.0002 – 1.6 million CFU inactivation of *E. coli* coming from the influent at the end of the treatment per cycle. The effluent had 0 – 0.001 million CFU *E. coli* per cycle. At the end of each treatment cycle, it is estimated that a total of 0.4 – 2.9 million CFU of *E. coli* from a population of 122,705 is treated and diverted out of pollution pathway. If these *E. coli* numbers had remained in the environment untreated, it would lead to a host of infections.

4.4 Conclusion

In conclusion, the average *E. coli* and total coliform levels in the FS ranged from 2.6 x 10^5 – 1.5 x 10^6 and 4.1 x 10^5 – 4.7 x 10^6 CFU/100ml, respectively for the three cycles. The inactivation efficiency of the dewatering process saw *E. coli* removal to be minimal (0.0 – 14.3%) in the percolate. The DFS on the other hand observed higher inactivation efficiencies of 88.2 – 98.0% (1 – 2 log reductions) for *E. coli* across the three cycles and 0.0 – 90.3% (up to 2 log reductions) for total coliform after dewatering. The retention in the maturation pond resulted in complete inactivation (100.0%) of *E. coli* during cycles 2 and 3 while cycle 1 observed 98.6% inactivation efficiency of *E. coli* (2 log reductions). The process significantly reduced *E. coli* levels as 100% pathogen inactivation was attained. As such, the *E. coli* levels observed in Cycles 2 and 3 were below the maximum limits set by the Ghana EPA and EU. mass balance estimations around the FS dewatering reveal that between 0.1 – 0.3% of all incoming *E. coli* is retained in the DFS per cycle. A range of 25 – 55% of the *E. coli* is retained in the percolate moving on to the effluent polishing in the facultative and maturation ponds. At the end of each treatment cycle, it is estimated that a total of 0.4 – 2.9 million CFU of *E. coli* from a population of 122,705 is treated and diverted out of pollution pathway.

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Chapter 5 a. Assessing consistency in the co-composting of faecal sludge and food waste in Ghana

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Abstract
A faecal sludge (FS) co-composting study assessed the extent of consistency in compost characteristics between and within batches. The study focused on the consistency of the co-composting process by measuring the variability of key parameters. The set up consisted of 12 FS and FW co-composting piles in three successive batches (1, 2 and 3). Between batch consistency was assessed in the three successive batches of co-composted FS and food waste (FW). Within batch consistency was assessed in each of the three batches by dividing the batch into four separate replicate piles. Characteristics of interest were *E. coli*, as well as selected physico-chemical (pH, EC, Mg, Ca, N, NH_4, NO_3, P, avail. P, and K) and heavy metals (Se, Fe, Cd, Cu, Hg, Ni, Pb and Cr). Data were subjected to analysis of variance (ANOVA) using SPSS. Results show that, *E. coli* levels were not consistent between the successive batches during the entire co-composting process. While variations between batches were only observed for EC and nutrient parameters, variations were evident for several measured characteristics within batches. The measured coefficient of variations (CVs) within batches ranged between 0 – 125% and 3 – 111% for heavy metals and nutrients, respectively. It is recommended that a threshold value be created for determining what is an acceptable level of variation in FS co-composting.

Keywords: Faecal sludge; Co-composting; Variation; Traceability; Batch productions; Consistency; Food Waste

5.1 Introduction

In recent years, composting has increasingly been promoted as a reliable and low-cost method for sanitizing faecal sludge (FS) from onsite sanitation systems, particularly where there are opportunities to use the recovered nutrients in agriculture (Wang et al., 2022). FS in developing countries remains one of the most challenging waste generated and the fast threatening pollutants as it facilitates the spread of pathogens (Coffey et al., 2017; Crocker et al., 2016; Velkushanova et al., 2021). Composting is considered one of the best options due to its sustainability and integration into circular bioeconomy concept, which is what the current European system is committed to (Razza et al., 2018). This process generates a safe and stable bioproduct, the compost, which can be used as organic fertilizer (Soobhany et al., 2017). Moreover, the high temperatures reached during the process eliminate possible pathogens and, in addition, could reduce ARGs present in the raw materials (Zittel et al., 2020).

Composting or co-composting is often preferred as a treatment method for FS in sub-Saharan Africa due to its low energy requirements and efficacy in terms of the recovery of critical nutrients (Nitrogen, Phosphorous and Potassium) (Manga et al., 2021) and organic matter which can be used for agriculture. Co-composting of FS and other organic solid waste streams particularly uncooked food waste (FW) from markets, food stores, restaurants etc. contributes as an efficient waste management tool and allows for recycling of nutrients and organic matter into agriculture thereby closing the nutrient loop (circular economy). The technologies chosen for co-composting usually depend on the geographical location, available capital, quantity, and type of feedstock to be used etc. There are generally two main types of aerobic co-composting systems namely open systems such as windrows and static piles, and closed systems such as vessel systems (Alamin, 2017).

According to Manga et al. (2021), the effectiveness of FS co-composting has not been thoroughly explored especially in urban Africa. Robust research studies conducted and published on FS in peer reviewed journals to date are few (Al-Muyeed et al., 2017; Berendes et al., 2015; O. Cofie et al., 2009; Hashemi et al., 2019; Koné et al., 2007; Mengistu et al., 2018; Mulec et al., 2016; Nakasaki et al., 2011; Nartey et al., 2017; Oarga-Mulec et al., 2019; Thomas et al., 2018). Most of these earlier studies have addressed the optimization of FS co-composting with various organic residues and the effectiveness of the different bulking agents on FS sanitization, during open-air composting in one-time experiments or trials (Al-Muyeed et al., 2017; Berendes et al., 2015; O. Cofie et al., 2009; Mengistu et al., 2018; Nartey et al., 2017). Doubtlessly, co-composting of FS has been, and it is being extensively practiced globally, both informally and formally (Manga et al., 2021). However, little or no information is available on the consistency of the FS co-composting process over successive batch productions to guarantee dependable quality of product for both producers, consumers, and regulators in the value chain and to instil confidence in its quality and acceptability. Understanding consistency would help producers communicate with certainty the quality of co-compost or faecal derived fertilisers (FDF), promote user's trust (farmers, landscapers etc), and facilitates replication and easy regulations.

The importance of consistency in FS co-composting especially in sub-Saharan Africa cannot be overemphasized. The characteristics are affected by the type of initial feedstock, the process of composting itself and the maturity of the final product (Alamin, 2017). Feedstock type and treatment processes play a critical role in the characteristics of final compost hence its quality. This is because FS which is the primary feedstock is very variable in nature because of the different user behaviour, on-site sanitation systems, sludge collection and transportation methods, etc. (Bassan, Tchonda, et al., 2013; Heinss et al., 1999; Ward et al., 2019). The characteristic of FW is equally affected by the types of foodstuff and vegetables available in time which in turn is influenced by the seasonality, local food trade etc. (Fisgativa et al., 2016). Treatment processes, particularly where it mostly consists of manual process steps like turning of heaps could introduce some variability.

Understanding the changes and extent of consistency in FS co-composting over the continuous production cycle is critical for instilling confidence in the quality and use of FDF as well as assessing appropriate management strategies to ensure quality and safe FDF are produced from such co-composting enterprises to meet international guidelines. To encourage public/consumer confidence in FDF quality and acceptability, the product and process must satisfy two criteria. Firstly, the FDF must meet key quality standards and secondly, the co-composting process must be consistent to guarantee dependable quality. In this study, we focus on the consistency of the co-composting process by measuring the variability of key parameters. The objective is to assess the degree of consistency between FDF batches and within batches over time.

5.2 Methodology

5.2.1 Experimental site, feedstock sourcing and pre-treatments

The study was carried out at field scale at Akorley, Somanya (latitude 6° 6′ 0″ N and 3° 3′ 0″ N and between longitude 0° 0′ 30″ W and 0° 0′ 10″ W, Sadiq, 2016) at the Jekora Ventures Limited (JVL) – Yilo Krobo Municipal Assembly (YKMA) Recycling Plant, in the YKMA of Ghana (Figure 5-1). The annual rainfall of the area ranges from 750 to 1,600 mm and it's spans from May to October (bimodal). Average temperatures range between 24 and 30°C while relative humidity ranges between 60 and 90% (Sadiq, 2016).

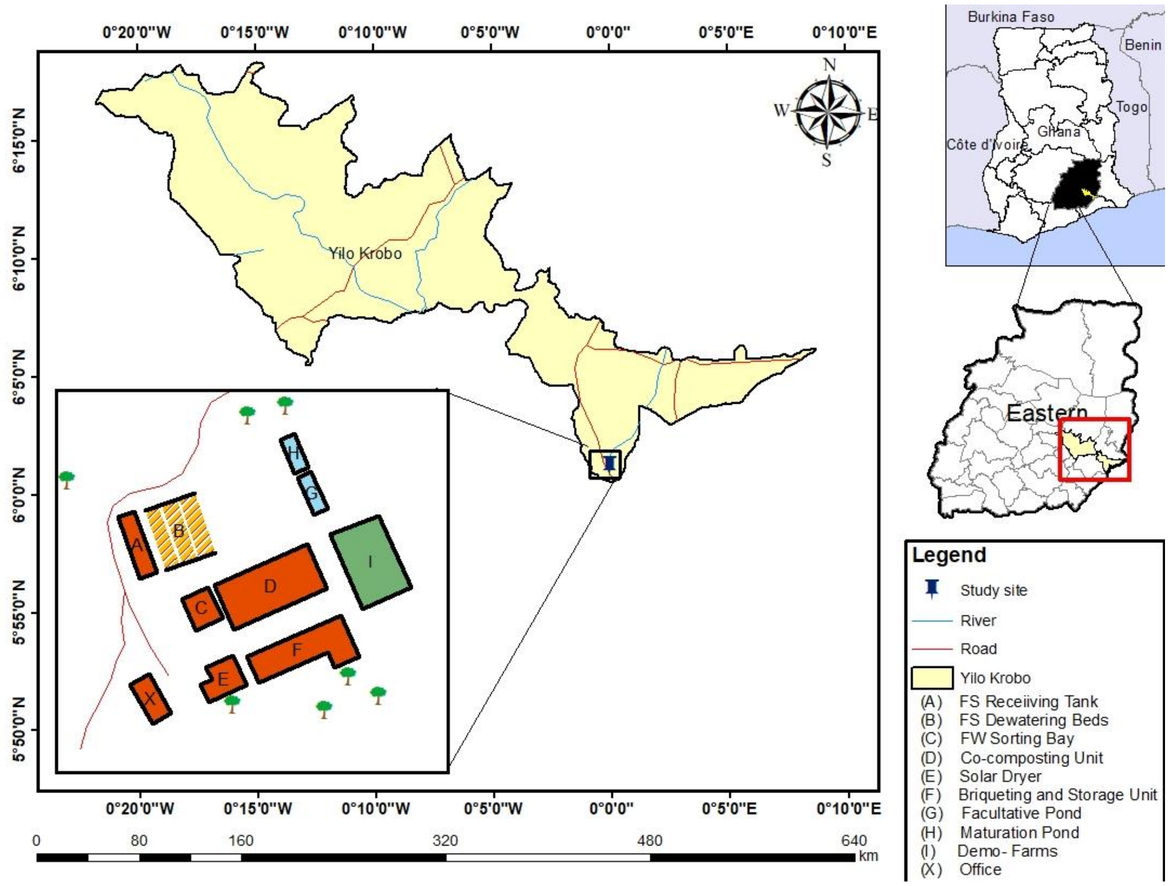


Figure 5-1. Map of study location showing study site

The FW was obtained from JVL's source segregation operations in major local markets and some institutions within YKMA, Ghana. The fraction of FW collected included source-separated fruit waste (citrus, watermelon, pineapple etc.), vegetable waste (cabbages, garden eggs, etc.) and foodstuff waste (plantain stalks, yam peels, potato etc.). At the recycling plant, FW was further sorted out to remove other foreign materials (mostly inert materials) that may have escaped the initial segregation at source. Larger sizes of the FW were cut into pieces of about 2 – 3cm to increase their surface area and allow for efficient aeration during the co-composting process. FW used for the study was characterised. The parameters that were considered for the characterisation are described in Table 5-1. Raw human excreta collected from various on-site sanitation systems and transported by vacuum trucks from in and around YKMA was dewatered on sand drying beds at the JVL-YKMA Recycling Plant. Mixtures of sludge from public toilets and households were loaded onto the drying beds at a ratio of 2: 1 v/v until the bed was full and allowed to dewater. The dewatered FS (DFS) was harvested manually from the drying beds and characterised. The DFS produced on each drying bed was treated as different sources and characterised differently.

5.2.2 Co-compost treatments

The set up consisted of 12 DFS and FW co-composting piles in three production batches (1, 2 and 3) for the between batch tests. Within each batch were four replications of co-composting piles of DFS: FW at 1:3 w/w for the within batch tests. The co-composting piles contained approximately 2.0 tonnes of materials of 1.5m high and 10m base circumference. The interval between batches production was 2 weeks in a successive process. The active composting phase took about 8 weeks, and the curing/maturation phase took about 4 weeks. Within this period, piles were monitored to ensure sanitization conditions were achieved. These included manually turning piles every 3 days for the first 2 to 3 weeks and then once a week afterwards. Moisture contents of the piles were adjusted to 50 – 60% during turning. Daily temperature recordings were taken with a compost thermometer.

5.2.3 Sampling and analysis

Composite samples were collected from the piles following methods described in USDA and USCC (2001) every two weeks from the start of co-composting for moisture content, *E. coli* and helminths as well as selected physico-chemical (pH, EC, Mg, Ca, N, NH_4, NO_3, P, avail. P, and K) and heavy metals and trace elements: Ca, Mg, Mn, Cu, Zn, Fe, Pb, Cd, Cr, Hg, Ni, and Se were determined before and after the co-composting. Total N was determined by the modified Kjeldahl method described in Black (1965). Ammonium (NH_4-N), Nitrate (NO_3-N), Total P and K were determined by methods, as described in Okalebo et al. (2002). The pH and EC were measured using 1:5 and 1:10 compost: water w/v ratios, respectively described in USDA and USCC (2001). Organic carbon (OC) was determined by the Walkley and Black (1934) method. The *E. coli* and total coliform counts were done using the spread plate method (APHA-AWWA-WEF, 2001). Helminth egg was determined by the flotation and sedimentation method following a modified USEPA method (Schwartzbrod & Gaspard, 1998). The heavy metals were analysed by atomic absorption spectrophotometer following methods described by Chapman and Pratt (1962).

5.2.4 Statistical analysis

Data were subjected to analysis of variance (ANOVA) using SPSS statistical package and Genstat 12th edition statistical package. Between batch treatment means found to be significantly different from each other at ($p < 0.05$) were separated by the Least Significant Differences (LSD) tests. Within batch tests were carried out by the comparison of the coefficient of variations (CVs).

5.3 Results and Discussions

5.3.1 Initial characteristics of feedstock and co-composting piles at start

Results indicate that, despite the DFS and FW coming from the same location/sources in this study, the physico-chemical characteristic showed wide variation from the mean (Table 5-1). The wide variation in the characteristics may have been due to factors surrounding the sources such as seasonality etc. as reported by other studies (Bassan et al., 2013; Fisgativa et al., 2016; Heinss et al., 1999; Ward et al., 2019). At the start of co-composting, it was generally observed that, there was consistency in characteristics between batches 1, 2 and 3 as characterised by no significant differences in the measured parameters between the batches except for Org C, available P, Cd, and total coliform

(Table 5-1). The level of variability within batches were wide. The CVs ranged from 9 – 13% for pH and between 14 – 30% for EC. The levels of consistency within batches for nutrients was wide ranging from 4 – 76% (Table 5-1). For example, there was relatively more consistency observed in the total P and N concentrations in batch 2 than in batches 1 and 3 (Figure 5-2).

Table 5-1. Feedstock and initial characteristics of piles for the different batches of FS and FW co-composting.

Parameter	**DFS**		**Food waste**		**Batch 1**		**Batch 2**		**Batch 3**	
	Mean	Std. Dev (±)	Mean	Std. Dev (±)	Mean	CV (%)	Mean	CV (%)	Mean	CV (%)
pH (1:5)	6.8	0.4	9.6	0.3	8.8^a	10	9.2^a	9	9.0^a	13
EC (1:10) (ms/cm)	3.34	1.30	6.59	2.81	4.06^a	14	5.24^a	30	5.07^a	27
Total N (%)	2.67	1.02	1.36	0.26	2.04^a	31	1.54^a	6	1.55^a	10
NH_4-N (mg/kg)	1513.91	255.60	652.64	290.80	232.34^a	23	192.83^a	45	336.55^a	33
NO_3-N (mg/kg)	1232.64	917.30	612.34	69.34	366.79^a	27	380.52^a	34	501.08^a	42
Org. C (%)	26.56	6.30	37.85	5.65	22.25^b	13	35.78^a	3	36.02^a	6
Total P (%)	3.77	1.14	0.43	0.08	2.93^a	41	1.92^a	14	1.98^a	60
Avail. P (mg/Kg)	9.88	0.65	11.46	1.75	9.95^a	11	9.93^a	16	5.26^b	36
Total K (%)	0.63	0.14	0.93	0.06	1.34^a	21	1.17^a	67	1.83^a	15
Avail. K (mg/Kg)	0.13	0.03	6.34	2.74	0.33^a	26	0.25^a	50	0.40^a	10
Ca (%)	2.53	0.38	0.76	0.32	3.52^a	16	2.14^a	76	2.66^a	21
Mg (%)	0.64	0.28	0.29	0.09	0.33^a	21	0.32^a	10	0.36^a	18
Mn(mg/Kg)	36.82	2.38	17.56	8.70	14.01^a	26	15.14^a	25	19.72^a	4
Cu(mg/Kg)	18.25	1.99	2.58	0.46	4.86^a	21	4.78^a	60	3.68^a	35
Zn(mg/Kg)	350.22	81.75	906.39	293.50	422.57^a	6	617.82^a	41	474.49^a	14
Fe(mg/Kg)	566.24	62.34	333.35	105.40	289.03^a	8	283.32^a	21	257.73^a	9
Pb(mg/Kg)	106.10	6.49	77.47	17.68	55.72^a	12	78.92^a	33	78.12^a	8
Cd(mg/Kg)	2.38	1.17	4.93	1.05	5.84^a	16	3.31^a	45	5.29^b	18
Cr(mg/Kg)	63.58	14.29	159.81	11.23	57.56^a	25	57.18^a	28	62.81^a	13
Hg(mg/Kg)	0.01	0.01	0.82	0.11	0.24^a	33	0.52^a	51	0.46^a	21
Ni(mg/Kg)	0.23	0.03	0.47	0.32	0.05^a	90	0.08^a	58	0.03^a	126
Se (mg/Kg)	0.18	0.06	0.36	0.09	0.09^a	32	0.12^a	84	0.03^a	153
E. coli (CFU/g)	8.1×10^3	1.1×10^4	1.3×10^3	8.0×10^2	5.2×10^{4a}	36	6.9×10^{6a}	128	1.7×10^{7a}	87
Total coliform (CFU/g)	9.6×10^4	1.3×10^5	3.3×10^3	2.4×10^3	3.8×10^{5b}	80	6.7×10^{7a}	24	6.0×10^{7a}	83

NB: Same letters on means in the column indicate no significant difference at 5%

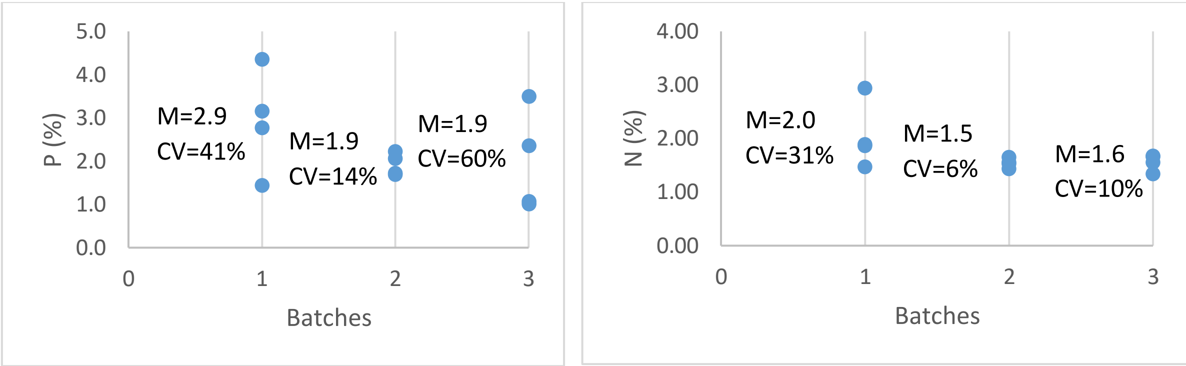
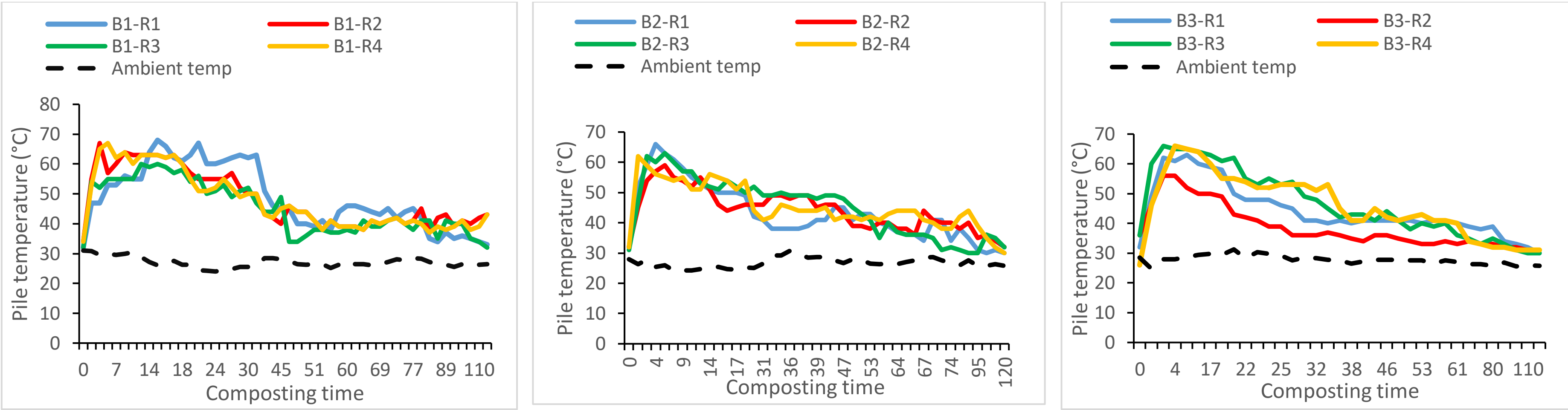


Figure 5-2. Consistency within batches for total P and N concentrations at start co-composting.

5.3.2 Temperature and moisture content of profiles

The key indicators of composting are temperature and pH (Cui et al., 2016). All piles within the batch achieved recorded temperatures above 50°C for weeks needed for pathogen deactivation during the thermophilic phases although the piles did not show identical or similar temperature trends throughout the thermophilic, mesophilic and maturation phases (less consistency in the trends of the pile temperatures) (Figure 5-3). While each pile achieved thermophilic temperatures within 2 – 5 days of co-composting, the period for the thermophilic phase varied within and between batches. An explanation for this phenomenon could be differences in the composition of the FW feedstock used as bulking agent which could have altered the substrate environment to bring an increase or decrease in temperatures. For instance, some components of FW like whole oranges constituted a suitable habitat for microbial proliferation by improving substrate properties like porosity, surface area that enhance microbial activities leading to increase in temperatures (Sánchez-García et al., 2015).

Another explanation can be unintentional inconsistencies introduced by workers during manual turning of piles and moisture adjustment. There can be inconsistency in the frequency of turning and the thoroughness of turning introduced by the workers. However, this was contrary to a previous study (Cofie et al., 2009) that found no significant effect of different turning frequencies on temperature changes. According to the time-temperature criteria provided by United States Environmental Protection Agency (USEPA)(USEPA, 2003), maintaining composting pile temperatures of >55°C for 3 days (or 15 days for windrow composting) reduced pathogen concentrations to non-detectable limits when the criterion is achieved.



a) Batch 1 b) Batch 2 c) Batch 3

Figure 5-3. Temperature profiles of batch piles (a, b, and c)

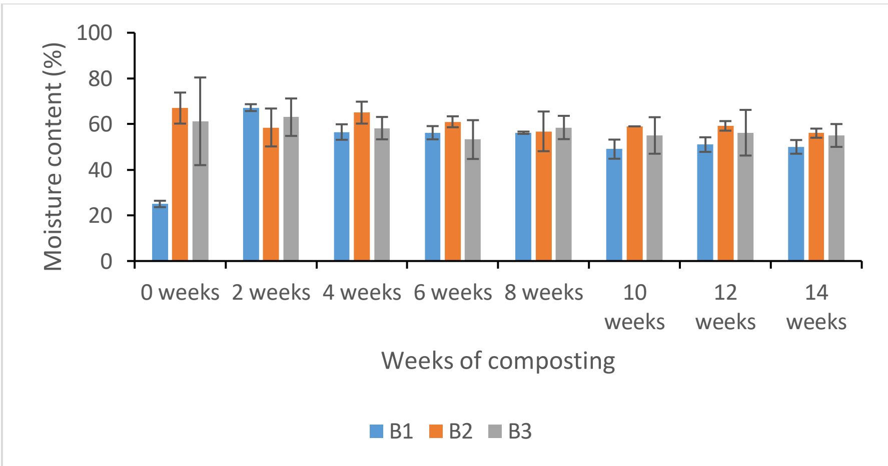


Figure 5-4. Changes in pile moisture content between batches. B1=Batch 1; B2=Batch 2; and Batch 3

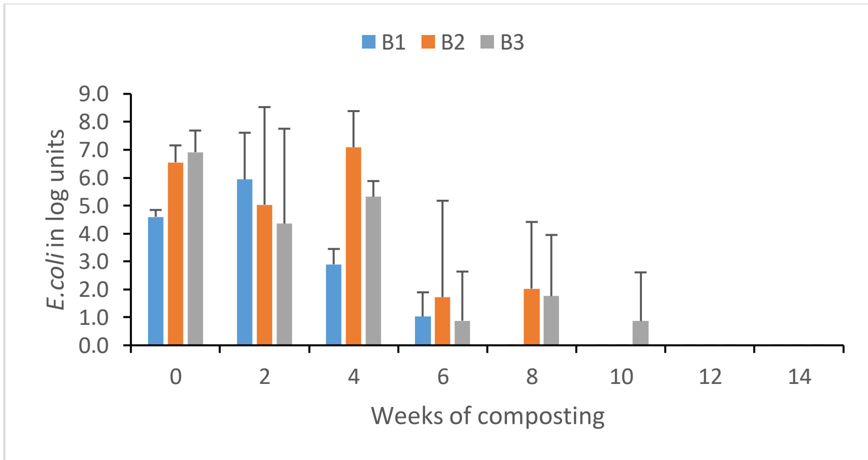


Figure 5-5. *E. coli* changes in batch piles in log units. B1=Batch 1; B2=Batch 2; and Batch 3

This criterion was fulfilled by most of the piles in this study. The effect on *E. coli* reduction is discussed in the next section. Moisture plays an essential role for the movement of microorganisms to move around to degrade the substrate and in the process generate heat to increase temperature. It is required that moisture content (MC) is maintained between 50 - 60% for optimal processes. Generally, there were no significant differences at $p<0.05$ in MC levels between batches indicating consistency between batches after the start of the co-composting process (Figure 5-4).

The significant differences observed between batches at composting week 0 was probably due to the uneven nature of particle sizes of the feedstock and the fact that feedstock may not have been thoroughly mixed at the start of the composting process. However, consistency within the batches indicated by the standard deviation was wide (Figure 5-4). There was generally high standard deviation within batches 2 and 3 than in batch 1 even from the start of co-composting and these could be linked to the specific composition of the FW in the piles (Sánchez-García et al., 2015).

5.3.3 *E. coli* and helminths

E. coli levels were not consistent between the successive batches during the entire co-composting process. There were statistically significant ($p<0.05$) differences in the mean *E. coli* concentrations between the batches (Figure 5-5). The differences could be due to the starting *E. coli* concentrations in the DFS feedstock and the differences in the temperatures generated by each pile. The different time-temperature regimes could have been the major cause of the differences in pathogen deactivation during the process inciting the differences in the *E. coli* levels (Manga et al., 2023). The consistency within the batches were also quite less, characterized by measured high standard deviations amongst the piles in a batch. This could be explained by differences in pile turning during the co-composting process. This process is largely carried out manually by workers with shovels and spades. As a result, there was a high possibility of some piles being thoroughly turned than others. Manga et al., (2023) found that turning frequency has a statistically significant ($p < 0.05$) effect on pathogen inactivation in FS compost. The 3 days turning frequency (TF) piles exhibited shorter pathogen inactivation periods (8 weeks) than 7 days TF and 14 days TF piles (10 weeks). Cofie et al., (2009) on the other hand found no significant effect of different turning frequencies on the temperature changes and the quality of mature compost. However, the degree of consistency may be less of importance to the FS co-compost producer, user, and regulator where complete deactivation of *E. coli* at the end of the co-composting period. If the final co-compost quality is meeting the standard, then the extent of consistency in the piles prior to co-compost maturity may be of less significance.

The time-temperature criteria were fulfilled by most of the piles even though there was complete deactivation of pathogen after the 10th week of co-composting. Similar findings of complete deactivation were observed by Manga et al (2019). and Evans et al (2015). Other studies (Cabañas-Vargas et al., 2013; Droffner & Brinton, 1995) on the contrary, found some pathogens in the final composts even after satisfying the time-temperature criteria for extended periods of time. No helminths were observed in the final co-compost characteristics in all batches.

5.3.4 Final characterisation of piles at the end of co – composting

Final product testing indicated that there were differences observed between batches (1, 2 and 3) for EC, and some nutrient (N, NH_4, NO_3, K, avail. K, Ca, Mn, and K) parameters indicating inconsistency in those characteristics (Table 5-2). A closer look at the consistency of measured parameters in replicated piles within batches showed coefficient of variations (CVs) ranging between 0 – 125% and 3 – 111%

for heavy metals and nutrients, respectively (Table 5-2). For instance, in Figure 5-6, there was relatively less consistency in batch 2 for Pb levels but more consistent in N% levels at the end of co-composting. The differences in nutrient levels were largely driven by the variable nature of the feedstock (Bassan et al., 2013; Fisgativa et al., 2016) and manual nature of the co-composting operations and the open-air composting method employed which allowed for different degrees of nutrient losses via gaseous escape and through leachate.

Table 5-2. Final characteristics of piles for the different batches of FS and FW co-composting.

Parameter	**Batch 1**		**Batch 2**		**Batch 3**		**ECN-QAS**	**ECOCERT Standard**
	Mean	CV (%)	Mean	CV (%)	Mean	CV (%)		
pH (1:5)	8.7^{a}	7	8.9^{a}	7	8.9^{a}	2	-	-
EC (1:10) (mS/cm)	5.3^{a}	15	6.3^{b}	2	6.4^{c}	2	-	-
Total N (%)	1.0a	20	1.3^{a}	3	1.7^{b}	12	-	-
NH_4-N (%)	0.13^{a}	8	6.65^{b}	7	7.04^{b}	5	-	-
NO_3-N (%)	0.12^{a}	8	11.17^{b}	76	6.69^{b}	3	-	-
Org. C (%)	13.6^{a}	7	14.2^{a}	23	12.2^{a}	8	-	-
Total P (%)	1.8^{a}	11	1.3^{a}	31	1.3^{a}	8	-	-
Total K (%)	1.8^{a}	11	3.0^{b}	17	2.6^{b}	8	-	-
Avail. K (mg/Kg)	0.14^{a}	14	0.19^{a}	111	0.48^{b}	2	-	-
Ca (%)	0.96^{a}	25	1.59^{a}	52	3.13^{b}	6	-	-
Mg (%)	0.33^{a}	24	0.57^{a}	40	0.56^{a}	9	-	-
Mn (mg/Kg)	14.94^{a}	18	23.08^{b}	18	24.06^{b}	5	-	-
Cu (mg/Kg)	6.21^{a}	38	7.07^{a}	22	6.98^{a}	15	300.0	70.0
Zn (mg/Kg)	531.46^{a}	17	701.08^{a}	29	500.87^{a}	8	600.0	200.0
Fe (mg/Kg)	289.32^{a}	4	307.89^{a}	24	273.56^{a}	4	-	-
Pb (mg/Kg)	66.72^{a}	15	64.61^{a}	40	71.28^{a}	7	40.0	25.0
Cd (mg/Kg)	6.08^{a}	18	4.80^{b}	45	3.12^{bc}	43	1.3	0.7
Cr (mg/Kg)	66.27^{a}	14	55.99^{a}	17	54.00^{a}	10	60.0	70.0
Hg (mg/Kg)	1.08^{a}	12	0.37^{a}	65	0.33^{a}	21	0.45	0.40
Ni (mg/Kg)	0.01^{a}	0	0.04^{a}	125	0.04^{a}	50	-	-
Se (mg/Kg)	0.02^{a}	0	0.09^{a}	44	0.06^{a}	33	-	-

NB: Same letters on means in the column indicate no significant difference at 5%

Physical parameters such as pH and EC had CVs ranging between 2 – 7% and 2 – 15%, respectively for replicated piles within batches, however significant consistency between batches (1, 2 and 3) were observed for co-compost pH. Showing that pH levels was the same between co-compost batches and relatively more consistent within the batches.

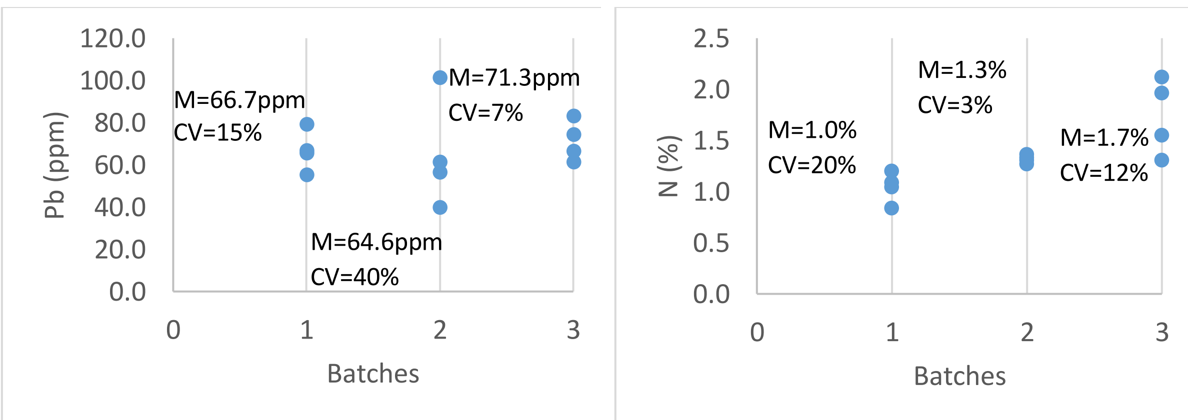


Figure 5-6. Consistency within batches for Pb and N concentrations at end of co-composting.

Heavy metals (Zn, Pb, Cd, Cr and Hg) levels did not consistently meet the standards allowable in Table 5-2 between batches. Only Cu levels consistently met the standards over successive batches. The sources of the heavy metals were from the feedstock, in this case FS and FW. The co – composting process itself did not seem to have significantly affected the levels of heavy metals between batches in this study. These findings can inform strategies to optimize the feedstock mixing ratio at the start of co-composting to ensure critical heavy metal standards are met.

Having information and an understanding of the level of consistency or variability in FS co-composting with FW in sub-Saharan Africa would help compost producers especially FS co-compost producers be more assertive about their product quality and be able to communicate with certainty the FS co-compost quality to their buyers. This would promote FS co-compost buyer's trust in the process and quality and therefore enhance the market and adoption options for FS co-compost that would ensure that FS are not dumped indiscriminately into the environment to pollute but rather transformed into a valuable product. The information and understanding of the level of consistency would also support policy makers and regulators in framing more responsive industry standards and regulations that are achievable and are reflective of the local context.

It is ideal that FS co-composting processes must have low CVs (consistent) to guarantee dependable quality. As at the time of discussing these findings, there is no known threshold or rule for determining what an acceptable level of CV for quality parameters should be for FS co-composting. But the question of how low we should go for that threshold value would be determined from future research supported by an exploration of causes of within and between batch variations. This must not only be left to academia but must be a joint dialogue, research and formulation by FS co-compost producers, users, regulators, and academia to create acceptable consistency levels.

5.4 Conclusion and Recommendation

In conclusion, the level of inconsistency or variations between FS and FW co-compost batches (1, 2 and 3) were only observed for EC, and the measured nutrient parameters at the end of co-composting. Replicate co-compost piles within batches exhibited coefficient of variation (CV) of measured parameters ranging between 0 – 125%. There was less consistency in nutrients between successive batches and CV within batches was wide. Consistency levels for *E. coli* may not be an issue if pathogen inactivation is complete. As at the time of discussing these findings, there is no known threshold or rule for determining what an acceptable level of CV for quality parameters should be for FS co-composting.

It is ideal that, FS co-composting processes and product must have low CVs (consistent) to guarantee dependable quality. It is therefore recommended that a threshold value be created for determining what is an acceptable level of CV for FS co-composting. This would be supported by a future exploration of causes of within and between batch variations. FS co-compost producers, users, regulators, and academia must dialogue to create acceptable consistency levels.

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Chapter 5 b. Effect of storage duration and temperature on pathogen load, heavy metals, and nutrient levels of faecal derived fertiliser.

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Abstract
Faecal derived fertiliser (FDF) is a promising alternative to mineral fertilisers in sub-Saharan Africa because of declining soil fertility and rising demand for renewable fertiliser sources. However, reasons for continuous debate regarding the safety of FDF is the notion that, even though treatment is achieved, metals, harmful toxins and pathogens may still be present at undetectable levels. A major concern is the recolonization of these undetectable indigenous pathogens to hazardous levels while undergoing storage awaiting field application. This study assessed the effect of various storage temperature and storage duration on previously undetected indigenous *E. coli* and N, P, K changes in two FDFs during a six-month storage. A 2x3x6 factorial design was used with factors: fertiliser, temperature, and storage duration. The factorial had 36 experimental conditions in a completely randomized design with three replications. FDF samples were collected monthly for six months and analysed for pH, EC, organic carbon, total N, NH_4-N, NO_3-N, total P, total K, *E. coli* and total coliform. Results show no detectable presence of *E. coli* in FDF at the end of the storage duration indicating no indigenous re-growth. After storage, total nitrogen (N) was affected by fertiliser type with higher N of 2.4% recorded for enriched co-compost (NECo) and 1.1% for co-compost (Co). However, storage temperature and duration did not have any effect on N. The available forms of nitrogen (NH_4-N and NO_3-N) were significant for the main effects of fertiliser, temperature, duration, and their interaction effects. An inverse relationship was observed between NH_4-N and NO_3-N concentrations in NECo under lower temperatures and longer duration of storage.

Keywords: faecal derived fertiliser; storage; *E. coli*; nutrients; temperature; faecal sludge; compost

5.1 Introduction

In developing countries, especially in sub-Saharan Africa (SSA), the demand for resources for food production is high, whilst reserves of non-renewable phosphorus resources are declining (Cordell & White, 2011; Magnone et al., 2022). Resource recovery from waste such as nutrients and water and their reuse hold great potential for meeting demand for food and renewable alternatives. Organic amendments or compost produced from organic waste materials must satisfy product safety and quality requirements to meet the needs of both organic and conventional agricultural farming. For excreta based amendments, even greater data and information is required to communicate product safety and quality due to potential pathogen transmission from farm products to humans (Bolton et al., 2014; Penakalapati et al., 2017). Excreta such as faecal sludge (FS) used as soil amendments is a promising alternative to mineral fertilisers in SSA as it contributes to sustainable agriculture and nutrient recycling (Adamtey et al., 2010; Amoah et al., 2017; Nartey et al., 2021; Pradhan et al., 2019). This is relevant in SSA because of soil fertility decline due to overexploitation, erosion, and climate change. However, there still could be chance of produce contamination arising from application of faecal derived fertilisers (FDF). FDF may still contain some inactive pathogens such as *Escherichia coli* O157:H7 and *Salmonella sp*. (Guan & Holley, 2003) which can become viable (infective) under favourable conditions. Aside pathogens, other constituents such as trace elements (heavy metals) etc. may accumulate in the soil environment to toxic levels and act as potential sources of bioaccumulation in crops and animal tissues.

Co-composting has proven as a reliable and low-cost technology for the treatment of organic waste including FS and food waste into FDFs prior to agricultural use (Bożym & Siemiątkowski, 2018; Uçaroğlu & Alkan, 2016; Wichuk & McCartney, 2007). Co-composting has been used as a practical and effective way for reducing human pathogen populations in the FS (Kim et al., 2009; Nartey et al., 2017; Wichuk & McCartney, 2007). One of the reasons for continuous debate regarding the safety of FDF is the notion that, even though treatment is achieved, metals, harmful toxins and pathogens may still be present at undetectable levels. The major concern for FDF is the possibility for re-contamination or recolonization of pathogens from either outside sources or subsequent re-growth of inherent pathogens to hazardous levels (Zaleski et al., 2005). There is a waiting period (storage duration) between when FDF is produced and when it is used/applied especially where conditions do not allow for immediate field application. Therefore, if co-composting ensures deactivation or reduction of pathogens to levels below detection limits and yet, there are reported cases of contamination at end use (Major et al., 2020; Oliveira et al., 2011, 2012), then it means the storage duration and conditions can play a critical role to the quality of the stored FDF. The storage period between when co-composting is complete and when the matured co-compost is applied to soils, exposes the co-compost to some changes in physico-chemical and pathogen characteristics. In this study, we refer to this storage period/duration and suggests that, the survival of *E. coli* and the availability of nutrients (and heavy metals) during this storage period, depend on abiotic (temperature, pH, humidity) (Pawłowska et al., 2019) and biotic (composition and diversity of the microbial community) factors. As at the time of this study, we are not aware of studies that have investigated the effect of storage conditions on inherent *E. coli* and N, P, K dynamics of FDF during storage. In SSA, where FDF is being promoted as an innovative solution to poor sanitation management and soil fertility, and in the context of resource recovery and reuse, there remains a knowledge gap on its quality characteristics during storage.

FDF just like any other co-compost material has crumbly, fine, brown, and light characteristic material; and this could change with time especially during storage awaiting field application. Stockpiling/ storage of compost is a common practice if field conditions are not suitable for immediate land application or in warehouses awaiting transportation and sale following production. During storage, compost continues to decompose and CH_4 and N_2O gases can be released (Hao, 2007). Pathogens and other microbial populations may also undergo some changes in characteristics and population. Enteric bacteria reduced to low or undetectable levels during the thermophilic process may re-grow to higher populations, under conducive environmental conditions, during storage of FDF prior to land application. *Salmonella* spp., in particular, may potentially pose a re-growth problem and health hazard (Skanavis & Yanko, 1994). The re-growth phenomenon is of concern because of the potential for additional treatment requirement, higher disposal costs and loss of consumer confidence associated with a compliance failure (Williams, 2014). Several studies carried out on pathogen re-growth in co-compost and compost-based amendments after the co-composting process analysed the role of abiotic intrinsic factors such as moisture, temperature and available nutrients (Soares et al., 1995). Other authors have exclusively focused on the role of biotic factors such as indigenous microflora (Sidhu et al., 2001; Williams, 2014) in the suppression of mostly seeded or inoculated pathogens.

However, the effect of extrinsic abiotic factors such as storage duration and temperature conditions on potential indigenous *E. coli* re-growth and changes in FDF nutrient as well as other properties during storage has received very little research attention (Adamtey et al., 2009; Fane, 2016). To contribute new knowledge to the ongoing debate regarding safety of using FDF, it is thus important to investigate how extrinsic abiotic factors such as storage duration and temperature conditions affect inherent *E. coli* and N, P, K changes. It is evident that, some composts can provide niches for pathogen survival and even re-growth, but the fate of inherent *E. coli* in FDF during storage is still not clear after treatment. Foodborne disease outbreaks linked to excreta-based fertiliser revealed gaps in knowledge on the microbiological safety of such fertiliser products and highlighted the need for implementing proper risk-reduction strategies during storage for them (Chen et al., 2018). Therefore, the aim of this study was to assess the effect of various temperature conditions and storage durations on indigenous *E. coli* and N, P, K changes of FDF during storage.

5.2 Materials and methods

5.2.1 Faecal derived fertiliser production

The FDFs for this study were prepared at the Jekora Ventures Limited (JVL) – Yilo Krobo Municipal Assembly (YKMA) Recycling Plant, in Akorley, Somanya (latitude 6° 6′ 0″ N and 3° 3′ 0″ N and between longitude 0° 0′ 30″ W and 0° 0′ 10″ W, Sadiq, 2016) in the YKMA of Ghana. The plant is a local commercial FS and solid waste treatment operation that had been producing FDF and biomass waste briquettes since 2020. A uniqueness of the JVL-YKMA Recycling Plant is that it is a 'end-to-end' from FS dislodging, treatment, to end product as compost (FDF) all happening in one place and allows ease of monitoring of the various processes. The FDF was prepared from FS collected by vacuum trucks (from households and other institutions) and source segregated food waste (FW) sourced from local markets and restaurants. The FS and FW were taken through pre-treatment process of dewatering and shredding, respectively before the co-compost piles were formed in replicates and in batches. In a batch, were four replicates of co-compost piles built with FW and dewatered FS (DFS) at a ratio of 3:1 w/w. Each pile was approximately 2.0 tons in weight and of the size 1.5 x 10 m (height x base circumference) at the start. The process lasted for 100 days and the matured co-compost post processed by spreading out the piles to air dry until moisture content was <15%. A detailed description of the co-composting process is described in Nartey et al (2022).

The FDF that is matured co-composts (Co) was divided into two parts and one part enriched with ammonium sulphate mineral fertiliser to attain 3% total N content (NECo), following methods described by Adamtey et al. (2009). The moisture content of the FDFs at the end of production was maintained between 10 – 15%. Both Co and NECo were filled into labelled 5kg sized plastic sacks lined with transparent polythene material (a reproduction of the real life 50kg sized sacks used for bagging FDF from the recycling plant). The sacks in each group were randomly further divided into two sub-groups before placing in storage.

5.2.2 Experimental Design

A 2x3x6 factorial design was employed. The factors were fertiliser type (F), storage temperature (T) and storage duration (D). There were two types of FDFs (F1 = NECo and F2 = Co), three levels of temperature conditions (T1 = 5°C, T2 = 25°C and T3 = ambient temperature), and six levels of the storage duration (D1 = 1 month, D2 = 2 month, D3 = 3 months, D4 = 4 months, D5 = 5 months, and D6 = 6 months). The factorial design table is shown in Figure 5-1.

Storage duration (months)

Temperature (°C)	D1	D2	D3	D4	D5	D6
T1	T1D1	T1D2	T1D3	T1D4	T1D5	T1D6
T2	T2D1	T2D2	T2D3	T2D4	T2D5	T2D6
T3	T3D1	T3D2	T3D3	T3D4	T3D5	T3D5

Co (F2)

Storage duration (months)

Temperature (°C)	D1	D2	D3	D4	D5	D6
T1	T1D1	T1D2	T1D3	T1D4	T1D5	T1D6
T2	T2D1	T2D2	T2D3	T2D4	T2D5	T2D6
T3	T3D1	T3D2	T3D3	T3D4	T3D5	T3D6

NECo (F1)

Figure 5-1. Factorial design table for storage study.

The 2x3x6 factorial had 36 experimental conditions in a completely randomized design (CRD) with three (3) replications. Thus, a total of 108 experimental treatments were conducted with the FDFs. The 5°C and 25°C storage temperatures were achieved by placing the bags in large storage cold-rooms at Noguchi Memorial Institute for Medical Research (NMIMR) and International Water Management Institute research labs in Ghana, respectively. The ambient storage conditions were achieved by placing the bags under a built shed over a concrete platform. The ambient temperatures for the duration of the entire experiment ranged from 24 - 30°C.

5.2.3 Sampling and analyses

The Co and NECo samples were collected on monthly basis (at the end of each month of storage) from each treatment factor and analysed for pH, electrical conductivity (EC), organic carbon, total N, NH_4-N, NO_3-N, total P, total K, *E. coli* and total coliform. Heavy metals and trace elements: Ca, Mg, Mn, Cu, Zn, Fe, Pb, Cd, Cr, Hg, Ni, As and Se were determined before and after the experiment. Total N was determined by the modified Kjeldahl method described in Black (1965). Ammonium (NH_4-N), Nitrate (NO_3-N), Total P and K were determined by methods, as described in Okalebo et al. (2002). The pH and EC were measured using 1:5 and 1:10 compost: water w/v ratios, respectively described in USDA and USCC (2001). Organic carbon (OC) was determined by the Walkley and Black (1934) method. The *E. coli* and total coliform counts were done using the spread plate method (APHA-AWWA-WEF, 2001). Helminth egg was determined by the flotation and sedimentation method following a modified USEPA method (Schwartzbrod & Gaspard, 1998).

5.2.4 Statistical analysis

Treatment effects were subjected to analysis of variance (ANOVA) using Genstat 12th edition statistical package. Treatment means found to be significantly different from each other at ($p < 0.05$) were separated by the Least Significant Differences (LSD) tests.

5.3 Results

5.3.1 Characteristics of faecal derived fertiliser before and after enrichment

The characteristics of the various FDFs (Co and NECo) used in the study is shown in Table 5-1 and compared with some international organic/compost standards. The concentrations of Avail. P, Ca, Cu, EC, Hg, Mg, Ni, Zn, total K and P did not change from what was observed in Co after enrichment to NECo (Table 1). As expected, total N, NH_4-N and NO_3-N were significantly higher in the NECo than in Co. The pH and available K concentration in the Co reduced after enrichment while heavy metals such as Pb, Cr, Cd and As concentrations also reduced significantly at $p<0.05$ after enrichment. *E. coli* levels were below detectable limits in the FDFs (Table 5-1).

Table 5-1. Physico-chemical characteristics of the faecal derived fertiliser types.

Parameter	Co	NECo	ECN-QAS	ECOCERT Standard
pH (1:5)	8.7^{a}	6.1^{b}	-	-
EC (1:10) (mS/cm)	5.3^{a}	5.6^{a}	-	-
Total N (%)	1.04^{b}	3.03^{a}	-	-
NH_4-N (mg/kg)	1300.0^{b}	509200.0^{a}	-	-
NO_3-N (mg/kg)	1200.0^{b}	231000.0^{a}	-	-
Org. C (%)	13.6^{b}	17.9^{a}	-	-
C:N ratio	13.3^{a}	5.9^{b}	-	-
Total P (%)	1.8^{a}	1.8^{a}	-	-
Avail. P (mg/kg)	6.5^{a}	4.0^{a}	-	-
Total K (%)	1.8^{a}	2.1^{a}	-	-
Avail. K (mg/kg)	0.14^{a}	0.02^{b}	-	-
Ca (%)	0.96^{a}	1.22^{a}	-	-
Mg (%)	0.33^{a}	0.35^{a}	-	-
Mn (mg/Kg)	14.9^{b}	27.3^{a}	-	-
Cu (mg/Kg)	6.2^{a}	7.9^{a}	300.0	70.0
Zn (mg/Kg)	531.5^{a}	567.2^{a}	600.0	200.0
Fe (mg/Kg)	289.3^{b}	362.7^{a}	-	-
Pb (mg/Kg)	66.7^{a}	35.7^{b}	40.0	25.0
Cd (mg/Kg)	6.1^{a}	1.3^{b}	1.3	0.7
Cr (mg/Kg)	66.3^{a}	29.6^{b}	60.0	70.0
Hg (mg/Kg)	0.29^{a}	0.13^{a}	0.45	0.4
Ni (mg/Kg)	0.01^{a}	0.02^{a}	-	-
As (mg/kg)	1.08^{a}	0.28^{b}	-	-
Se (mg/Kg)	0.02^{b}	0.04^{a}	-	-
E. coli (CFU/g)	$<1.0^{a}$	$<1.0^{a}$		
Total coliform (CFU/g)	1.1×10^{4a}	$<1.0^{b}$	-	-
Helminth eggs (Ova/10g)	$<1.0^{a}$	$<1.0^{a}$	-	-

ECN-QAS = European Compost Network-Quality Assurance Scheme. Sources: (Ecocert, 2013; Leifert, 2017)

The heavy metals, Pb, Cr, Cd and As concentrations were observed to have 46%, 55%, 79% and 74% lower concentrations, respectively in the NECo after enrichment while Fe, Mn and Se concentrations were observed to be 25%, 83% and 100% higher, respectively after enrichment. Some of the heavy metal levels measured in Co and NECo were below the limits set by ECN-QAS and Ecocert standards (Table 5-1).

5.3.2 Effects of fertiliser type, storage temperature and duration on indigenous E. coli and total coliform counts of stored faecal derived fertilisers

5.3.2.1 Main effects of fertiliser type

In all the FDF types, *E. coli* were not observed. This means that re-growth of inherent *E. coli* did not occur irrespective of the FDF type. An indication that, once the primary and secondary treatment processes are effective and *E. coli* have been fully deactivated or reduced below detection limits, the probability of inherent re-growth or multiplication is low. Total coliforms were however present in the all the FDFs and the differences were significant at $p<0.05$. The Co had a mean total coliform count of 2.4 log units while the NECo had a mean total coliform count of 0.4 log units.

5.3.2.2 Main effect of storage temperature

The main effect of storage temperature was significant at $p<0.05$. The mean total coliform count was in the order of 1.6 > 1.5 > 1.0 log units for 25°C, 5°C and ambient storage temperatures, respectively. There were no significant differences in total coliform counts between 25°C and 5°C storage temperatures, however differences were significant between ambient storage temperature and the other temperature conditions. Storage temperatures effect on *E. coli* was not observed as *E. coli* was absent (below detection limit).

5.3.2.3 Main effects of storage duration

The main effect of duration was also significant at $p<0.05$. The mean total coliform count for the various duration is shown in Figure 2. Mean total coliform counts were observed at all time points except for month 5 where the coliform levels were below detection. There was a general increase in coliform counts from 1.3 log units in month 1 to 2.8 log units in month 2 before gradually declining to 1.1 log units in month 6 (Figure 5-2). The total coliform counts were significantly different between month 2 and the other months as well as between month 5 and the other months.

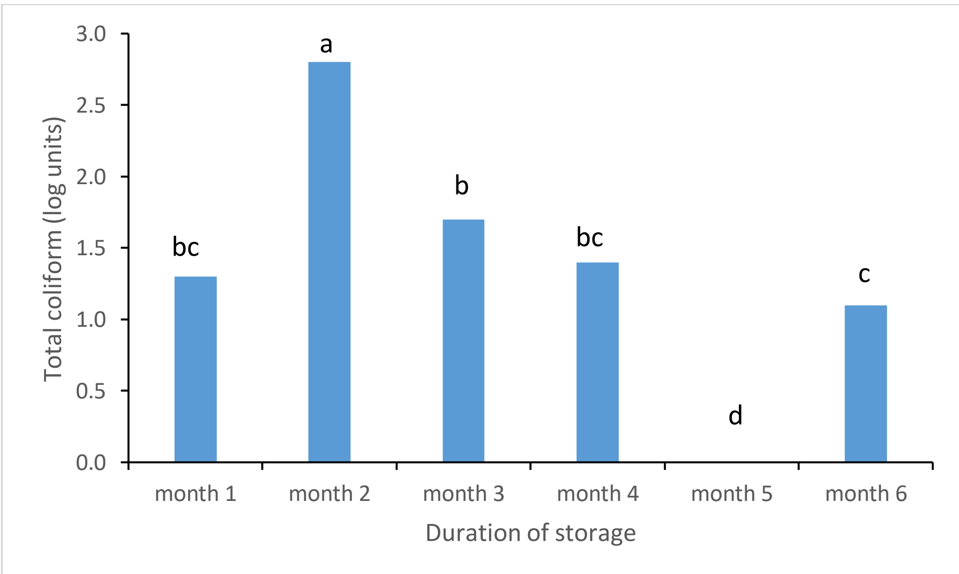


Figure 5-2. Main effect of storage duration on total coliform count

5.3.2.4 Interaction effects between fertiliser, storage temperature and duration

The interaction effects between fertiliser*duration and between temperature*duration were significant at $p<0.05$. The Co recorded higher levels of total coliform counts at earlier durations of storage than NECo (Table 5-2). The coliform levels reduced with longer duration of storage for both

Co and NECo. Decreasing storage temperatures (25°C to 5°C) encouraged more total coliform survival. However, their survival reduced with longer duration of storage (Table 5-3).

Table 5-2. Interaction effect between fertiliser type and storage duration on total coliform levels (log units/g)

Fertiliser type	Duration month 1	month 2	month 3	month 4	month 5	month 6
NECo	0.9^{e}	1.4^{de}	0.0^{f}	0.0^{f}	0.0^{f}	0.0^{f}
Co	1.8^{cd}	4.1^{a}	3.5^{a}	2.7^{b}	0.0^{f}	2.2^{bc}

Table 5-3. Interaction effect between storage temperatures and storage duration on total coliform levels (log units/g)

Temperature	Duration month 1	month 2	month 3	month 4	month 5	month 6
5°C	1.4^{cdef}	2.5^{a}	2.0^{bc}	1.6^{cde}	0.0^{g}	1.7^{bcde}
25°C	1.7^{cde}	3.7^{a}	1.8^{bcd}	1.2^{def}	0.0^{g}	1.0^{ef}
Ambient	0.9^{ef}	2.1^{bc}	1.3^{cdef}	1.2^{def}	0.0^{g}	0.7^{fg}

The interaction effect between fertiliser*temperature*duration on total coliform counts was also significant at $p<0.05$. The results show that, Co at lower temperatures (25°C and 5°C) and at shorter storage durations (≤ 3 months) had the highest counts of total coliform. This similar trend was also observed for the NECo.

5.3.3 Effects of fertiliser type, storage temperature and duration on nutrient and heavy metals characteristics of stored faecal derived fertilisers

5.3.3.1 Main effects of storage temperature, duration, and fertiliser type on nutrient characteristics

The pH, total N, NH_4-N and NO_3-N of stored FDFs under the different storage temperatures conditions were not significantly different at $p<0.05$ (Table 5-4). Total P has nearly doubled in all storage temperatures from the initial 1.8% in Table 5-1. The results also show that decreasing storage temperatures led to an increase in EC of fertiliser types. Organic carbon (OC) did not differ between 5°C and 25°C storage temperatures.

Table 5-4. Main effect of temperature on stored fertiliser nutrient characteristics

Temperature conditions	pH (1:5)	EC (1:10) (mS/cm)	Organic carbon (%)	Total N (%)	NH_4-N (mg/Kg)	NO_3-N (mg/Kg)	Total P (%)	Total K (%)
5°C	8.1^{a}	10.0^{a}	27.4^{a}	1.8^{a}	535^{a}	151^{a}	4.1^{a}	2.6^{b}
25°C	8.1^{a}	9.8^{b}	27.4^{a}	1.8^{a}	529^{a}	164^{a}	2.8^{b}	2.8^{a}
Ambient	8.1^{a}	9.8^{b}	26.7^{b}	1.7^{a}	509^{a}	154^{a}	3.2^{b}	2.5^{b}

The pH of FDFs generally showed an increasing trend with longer storage duration (Table 5-5). There were no significant differences in pH between months 3, 4, and 5. The duration of storage did not affect the total N levels of FDFs while total K showed a general 25% increase in levels from 2.1% in month 1 to 2.8% in month 6 (Table 5-5). The highest EC of 10.7 mS/cm was recorded in month 1 and the lowest EC of 9.1 mS/cm recorded for month 3. EC pretty much doubled in all months from the starting EC in Table 1 in response storage duration. However, OC content was significant at $p<0.05$ and showed a gradual increase with longer duration of storage albeit the values were not significantly different between months 1 and 2. The increase in OC in month 3 was almost double of the content in months 1 and 2.

Table 5-5. Main effect of storage duration on nutrient characteristics of FDFs

Storage duration	**pH (1:5)**	**EC (1:10) (mS/cm)**	**Organic carbon (%)**	**Total N (%)**	**NH_4-N (mg/Kg)**	**NO_3-N (mg/Kg)**	**Total P (%)**	**Total K (%)**
month 1	7.9^{e}	10.7^{a}	17.1^{c}	1.8^{a}	510^{bc}	151^{bc}	1.8^{b}	2.1^{c}
month 2	8.0^{d}	10.2^{a}	17.5^{c}	1.8^{a}	545^{b}	146^{bcd}	3.5^{a}	2.4^{a}
month 3	8.2^{a}	9.1^{e}	31.1^{b}	1.8^{a}	609^{ab}	113^{d}	3.7^{a}	2.6^{a}
month 4	8.2^{b}	9.9^{bc}	32.3^{a}	1.8^{a}	654^{a}	120^{cd}	3.8^{a}	2.8^{a}
month 5	8.2^{b}	9.5^{d}	32.7^{a}	1.8^{a}	408^{c}	176^{b}	4.1^{a}	2.8^{a}
month 6	8.1^{c}	9.8^{c}	32.3^{a}	1.8^{a}	421^{c}	231^{a}	3.5^{a}	2.8^{a}

The main effect of fertiliser type on nutrients after storage was significant at $p<0.05$. As expected, the levels were generally higher in the NECo than in Co except for total K (Table 5-6). The pH of NECo was lower than that of Co. The difference in EC was also significant between the FDF types.

Table 5-6. Main effect of fertiliser type on stored nutrient characteristics

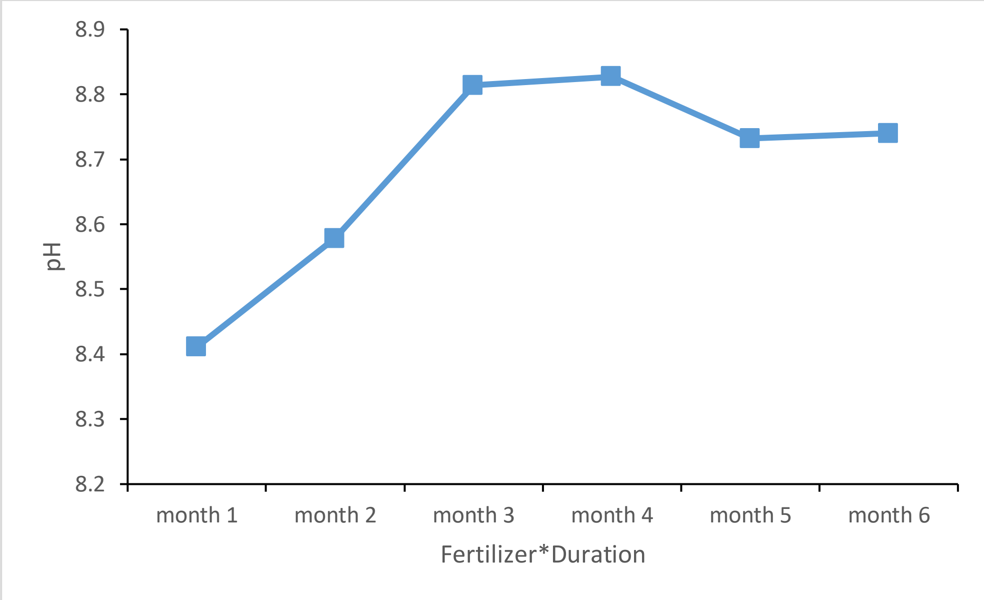
Fertiliser type	**pH (1:5)**	**EC (1:10) (mS/cm)**	**Organic carbon (%)**	**Total N (%)**	**NH_4-N (mg/Kg)**	**NO_3-N (mg/Kg)**	**Total P (%)**	**Total K (%)**
NECo	7.5^{b}	14.3^{a}	27.5^{a}	2.4^{a}	1013^{a}	191^{a}	4.0^{a}	2.3^{b}
Co	8.7^{a}	5.4^{b}	26.9^{a}	1.1^{b}	36^{b}	122^{b}	2.8^{b}	2.9^{a}

5.3.3.2 Interaction effect between fertiliser type, storage temperature and duration on nutrient characteristics

pH and EC

The interaction between the factors was only significant at $p<0.05$ for temperature*duration, and fertiliser*duration for pH. The interaction between fertiliser type and duration show, a more sigmoid-shaped relationship with higher pH values recorded during the mid-periods of the storage duration for both Co and NECo (Figures 5-3a and b).

a.



b.

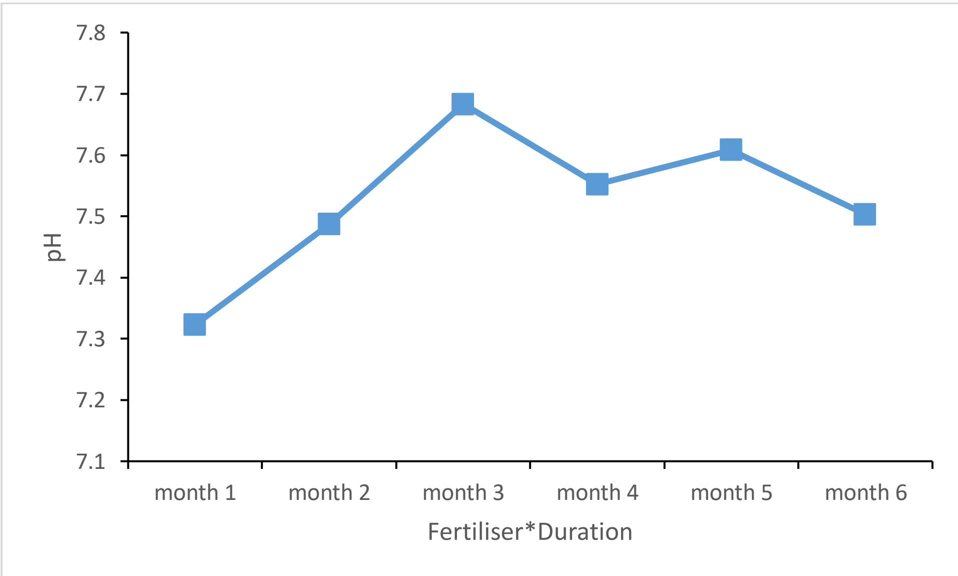


Figure 5-3. Interaction effect between fertiliser type and storage duration on pH (a = Co, b= NECo)

For EC, the interaction effect was significant for temperature*duration, fertiliser*duration, and fertiliser*temperature at $p<0.05$. The interaction between temperature and storage duration reveals that the EC increased in the first month of storage and generally under the colder temperature conditions (Table not shown). In terms of the interactions between fertiliser type and duration, it was also observed that EC of NECo almost tripled in month 1 from an initial 5.6 mS/cm (Table 5-1) before decreasing in value with longer duration of storage (Table 5-7). However, EC of Co did not generally change from the initial 5.3 mS/cm in Table 5-1 suggesting zero influence of storage duration on EC of Co (Table 5-7) i.e. higher EC values were recorded in the 1 month of storage (Table 5-7).

Table 5-7. Interaction effect between fertiliser type and storage duration on EC (mS/cm)

Fertiliser type	**Duration**					
	month 1	month 2	month 3	month 4	month 5	month 6
NECo	15.6^{a}	14.6^{b}	13.1^{d}	14.4^{b}	13.9^{c}	14.2^{bc}
Co	5.9^{e}	5.7^{e}	5.0^{g}	5.5^{ef}	5.0^{g}	5.3^{fg}

OC, N, P and K

The organic carbon (OC) content was significant at $p<0.05$ only for the interaction effect between temperature*duration. Higher OC concentrations were recorded for lower storage temperatures (5 and 25°C) ranges and with longer storage duration (Table 5-9). The interaction effects were not significant at $p<0.05$ for total N in this study, however the available forms of N (NH_4-N and NO_3-N) were significant for interactions between fertiliser*duration, temperature*duration, and fertiliser*temperature*duration. There were no significant changes in NH_4-N concentration in Co for the different duration of storage (Table 5-8). However, in the case of NECo, NH_4-N concentrations decreased with longer duration of storage (Table 5-8). In terms of interactions between temperature and storage duration on fertiliser NH_4-N concentration, the results generally show a decrease in NH_4-N concentrations with longer duration of storage under the lower temperatures (Table 5-9). Under ambient temperatures conditions, there was increase in NH_4-N concentration with longer duration of storage.

The interaction between fertiliser*duration on NO_3-N changes saw the concentrations of NO_3-N increase with longer duration of storage in NECo (Table 5-8). On the contrary, NO_3-N concentrations decreased with longer duration of storage in the Co. The highest NO_3-N concentration of 351 mg/Kg was recorded in NECo after 6 months of storage (Table 5-8).

Table 5-8. Interaction effect between fertiliser type and storage duration on available nitrogen

Fertiliser type	**Duration**					
	month 1	month 2	month 3	month 4	month 5	month 6
			NH_4-N (mg/Kg)			
NECo	994^{a}	1070^{bc}	1190^{ab}	1260^{a}	788^{d}	782^{d}
Co	27^{e}	24^{e}	31^{e}	46^{e}	29^{e}	59^{e}
			NO_3-N (mg/Kg)			
NECo	163^{cd}	166^{c}	105^{e}	117^{cde}	242^{b}	351^{a}
Co	139^{cde}	125^{cde}	121^{cde}	124^{cde}	109^{e}	111de

Table 5-9. Interaction effect between storage temperature and duration on organic carbon and available nitrogen

Temperature	**Duration**					
	month 1	month 2	month 3	month 4	month 5	month 6
			OC (%)			
5°C	16.1^{h}	17.3^{g}	31.7^{bcde}	33.0^{a}	33.2^{a}	33.1^{a}
25°C	17.5^{g}	18.0^{g}	31.3^{def}	32.3^{abcd}	32.8^{ab}	32.6^{abc}
Ambient	17.6^{g}	17.2g^{h}	30.4^{f}	31.6^{cde}	32.1^{abcde}	31.2^{ef}
			NH_4-N (mg/Kg)			
5°C	590^{abcd}	507^{bcde}	638^{abc}	646^{abc}	392^{efg}	439^{def}
25°C	698^{a}	657^{ab}	513^{bcde}	654^{ab}	333^{efg}	322^{fg}
Ambient	243^{g}	472^{cdef}	675^{ab}	661^{ab}	500^{bcdef}	501^{bcdef}
			NO_3-N (mg/Kg)			
5°C	160^{bcd}	140^{cde}	115de	134^{cde}	134^{cde}	220^{ab}
25°C	117de	123de	88^{e}	116de	262^{a}	279^{a}
Ambient	175^{bcd}	174^{bcd}	137^{cde}	111de	131^{cde}	194^{bc}

All factor interactions (fertiliser*temperature*duration) on NH_4-N and NO_3-N are described in Table 12. For the NECo, the highest NH_4-N concentration of 1370 mg/Kg was recorded after 1 month of storage and this gradually reduced to 612.0 mg/Kg after 6 months of storage (Table 5-10). The results also show that the NH_4-N concentration in stored FDF is affected by the type of fertiliser, temperature of storage and duration of storage. NECo experienced decreasing NH_4-N concentrations with longer storage duration under lower storage temperature (5 and 25°C). On the contrary, increasing NH_4-N concentrations were observed after long duration under ambient storage conditions (Table 5-10). In the case of NO_3-N, the reverse relationship was observed for NECo with NO_3-N concentrations increasing with longer storage duration under generally lower storage temperatures. This shows an inverse relationship between NH_4-N and NO_3-N concentrations in NECo under lower storage temperatures and longer duration of storage (Table 5-10).

Table 5-10. Interaction effect between fertiliser*temperature*duration on available nitrogen

Fertiliser	**Temperature**	**Duration**					
		month 1	month 2	month 3	month 4	month 5	month 6
		NH_4-N (mg/Kg)					
NECo	5°C	1140^{ab}	994^{bc}	1250^{a}	1270^{a}	748^{cd}	827^{cd}
	25°C	1370^{a}	1290^{a}	993^{bc}	1260^{a}	637^{de}	612^{de}
	Ambient	475^{e}	922^{bc}	1310^{a}	1260^{a}	979^{bc}	907^{bc}
Co	5°C	43^{f}	21^{f}	22^{f}	21^{f}	36^{f}	50^{f}
	25°C	27^{f}	29^{f}	32^{f}	50^{f}	29^{f}	32^{f}
	Ambient	12^{f}	21^{f}	37^{f}	67^{f}	22^{f}	94^{f}
		NO_3-N (mg/Kg)					
NECo	5°C	219^{cde}	160^{def}	118^{fgh}	147^{defg}	143^{efg}	334^{b}
	25°C	103^{fgh}	104^{fgh}	48^{h}	110^{fgh}	456^{a}	459^{a}
	Ambient	166^{def}	236^{cd}	150^{defg}	94^{fgh}	127^{fgh}	262^{bc}
Co	5°C	101^{fgh}	120^{fgh}	113^{fgh}	121^{fgh}	125^{fgh}	107^{fgh}
	25°C	132^{efgh}	143^{efg}	127^{fgh}	122^{fgh}	68^{gh}	99^{fgh}
	Ambient	183^{cdef}	112^{fgh}	124^{fgh}	129^{efgh}	135^{efgh}	127^{fgh}

For potassium (K), the interaction effect was only significant at $p<0.05$ for fertiliser*temperature. The K concentrations were significantly higher in the Co and at lower storage temperatures than at ambient temperatures (Table 5-11). In the NECo, the highest K concentrations were observed at 25°C followed by ambient and then 5°C indicating that lower temperatures may not favour K conservation during storage of NECo. Total phosphorus concentrations were not significant for any interaction effect between factors in this study.

Table 5-11. Interaction effect between fertiliser and temperature on potassium (%).

Fertiliser	**Temperature**		
	5°C	25°C	Ambient
NECo	2.0^{e}	2.6^{c}	2.2^{d}
Co	3.1^{a}	2.9^{ab}	2.8^{b}

After 6 months of storage, the Cd and Cr levels in both Co and NECo did not meet the ECN-QAS standards as indicated in Table 5-1 because their concentrations were higher. However, Cu, Zn, Pb and Hg levels conformed to the ECN-QAS standards.

5.4 Discussion

5.4.1 Faecal derived fertiliser characteristics before and after enrichment

Co-composting of DFS and uncooked FW or other bulking agents into FDF has demonstrated its effectiveness to inactivate completely or reduce faecal pathogens to undetectable levels while making nutrients and organic matter safe for crop growth (Manga et al., 2021; Nartey et al., 2017). According to Williams (2014) *E. coli* is used as an indicator for the microbiological quality of sludge-derived products destined for agricultural recycling and of the efficacy of the sludge treatment processes to demonstrate the presence of faecal and pathogenic bacteria and their removal. The results from this study showed the absence of *E. coli* or below detectable limit (<1 CFU/g) in the FDFs produced though total coliforms were quantified. The quality of FDFs did not meet the ECN-QAS or Ecocert standards for selected heavy metals (Leifert, 2017). The higher levels of Cd and Cr might be from the FS pit latrines and septic tanks used as feedstock. Users of these latrines also throw in rubbish/solid wastes which might have contained batteries, consumer electronics, newspapers, paints thus contributing to elevated levels of the Cd and Cr in the sludge.

This is further supported by Oghenerobor et al. (2014) who stated various sources of heavy metals that find their way into faecal matter sludge, and they include man-made sources like paint chips, used motor oils, batteries, ceramics, consumer electronics and natural sources like soil erosion. One of the ways to ensure FDFs meet the standards for heavy metals is to alter the feedstock mixing ratio at the beginning of co-composting. Further processing by enrichment with mineral N fertiliser in NECo further lowered the pH and reduced the total coliform numbers below detectable limits. This phenomenon was observed by Adamtey et al. (2009) in a similar study involving enrichment of FDF with various mineral nitrogen sources. In their study, Adamtey et al. (2009) found out that, after enrichment, the pH of compost increased in the urea-based products while a decrease in pH was observed in the ammonium sulphate-based products. A similar observation was made in this study where the pH of Co reduced from 8.7 to 6.1 after the ammonium nitrate-based enrichment to form NECo. The decrease in pH of the NECo could be attributed to nitrification, a set of processes that result in bacterially—mediated oxidation of ammoniacal nitrogen to nitrate with the associated the release of protons (Dworkin & Gutnick, 2012).

This lower pH probably led to the initial inactivation of total coliforms in Co as observed in this study. Organic carbon (OC) was observed to be higher in NECo than in Co after enrichment in this study. This was contrary to expectations, since the enrichment was targeted at increasing N concentration and thus expected not to affect the OC. However, this increase in OC could be explained from multiplication of microorganism numbers (biomass) and activities to mineralise of nutrients following enrichment. This could have accounted for the higher OC in the FDF. Some heavy metal levels of NECo after enrichment like Pb, Cr, Cd and As were reduced while Fe, Mn and Se levels increased. The reduction in heavy metal levels may be attributed to biological processes by the microbial community in NECo. Microorganisms such as bacteria, fungi, algae and yeast are known to tolerate and accumulate heavy metals (I. Ahmad et al., 2005). Extremely acidophilic bacteria (organisms thriving below pH of 3) species in the compost may have been responsible through a process referred as bioleaching (Pathak et al., 2009; Rohwerder et al., 2003). This is also supported by the fact that the enrichment process lowered the pH of the FDF to acidic ranges. The increase in concentrations may have been due to enrichment from the mineral fertiliser used for the enrichment. Similar findings were observed by John et al. (2010) when they utilised urea enrichment during the composting of cattle dung and poultry manure. They found increased concentration of Fe, Zn, Cu, and V in the enriched compost than in the non-enriched while Ni and Pb concentration reduced in the enriched compost compared with the non-enriched.

5.4.2 Effects of fertiliser type, storage temperature and duration on indigenous pathogen counts in stored faecal derived fertilisers.

A key condition for maintaining quality of a fertiliser in storage is ensuring its integrity and efficacy in terms of its chemical composition and physical structure is preserved (Pawłowska et al., 2019). In this study, all the FDFs placed in storage recorded *E. coli* below detectable limits. This simply means that there was no indigenous re-growth as the conditions of storage did not offer favourable environment for *E. coli*. It could also mean that, the co-composting treatment effectively and completely deactivated all viable *E. coli* prior to storage. This finding confirms the efficacy of co-composting as a viable treatment for producing safe FDF for sub-Saharan Africa. This is similar to earlier studies which found high temperatures to be responsible for the rapid inactivation of *E. coli* in co-composting piles (Christensen et al., 2002; Manga et al., 2021).

Total coliforms were present at different levels, and this was affected by the type of fertiliser. Though total coliforms are commonly found in the environment and as such are not directly associated with faecal contamination, they were found to be significantly lower in the NECo because of the enrichment process. Higher counts of total coliforms were found at lower storage temperatures (25°C and 5°C) compared to ambient conditions. This means the relatively higher temperature of ambient conditions accelerated the die-off of the total coliforms. This shows that temperature is an important factor affecting the survival of total coliforms (Fane et al., 2021). Chen et al. (Chen et al., 2018) observed similar findings that, *E. coli* and *S. enterica* survived for longer periods in composts at 5°C than at 22°C and greenhouse conditions. Wang et al. (2017) also reported similar findings for a non-O157 Shiga toxin–producing *E. coli* which survived better at 4°C than 22°C for 125 days in dairy compost. The main effect of storage duration on total coliform levels saw presence in all months of storage except at month 5. There was also an initial increase in levels in month 2 before gradually declining towards month 6. This implies that longer durations of storage suppressed coliforms level in FDF possibly due to competition for resources/nutrients running out or competition from other microorganisms. Contrary to our findings on coliforms, Sidhu et al. (2001) found out that long-term storage of the compost is not recommended as the inactivation rate of *Salmonella* decreased significantly with longer storage times, most likely due to the decline in indigenous microflora over time.

The interaction effect between fertiliser type, storage temperature and duration saw all the FDF had higher coliform counts at lower storage temperatures and at shorter storage duration. Implying that, regardless of the FDF type, lower storage temperature and shorter duration enhanced coliform levels. Generally, the storage of FDFs under ambient conditions for longer durations (at least 6 months) is more advantageous because it keeps total coliform levels lower and saves on energy required to store the fertiliser at lower temperatures. This will make it a less expensive option for both small and medium-scale enterprises (SMEs) who operate FS treatment plants to produce and store FDFs.

5.4.3 Effects of fertiliser type, storage temperature and duration on nutrient and heavy metals characteristics of stored faecal derived fertilisers

For the effect on nutrients, all factors affected the nutrient characteristics differently. The duration of storage did not have any effect on total N levels of the FDFs. However, the storage temperature did influence total N levels. The lowest temperature condition of 5°C retained the highest concentration of total N and least concentration under ambient conditions. As anticipated, this was due to exposure to higher temperatures under ambient conditions and fluctuating weather conditions substantially influencing losses through N volatilisation. The effect of storage duration saw no significant changes in NH_4-N for Co for the duration of storage probably because N mineralisation was happening at a much slower constant rate in the compost compared to the enriched compost (NECo). While the NH_4-N concentration of NECo decreased by 21% after storage, it coincided with a 115% increase in NO_3-N concentration after storage. This is strongly linked to nitrification process which converts ammonium to nitrite and then to nitrate (Adamtey et al., 2009). This similar trend of nitrification was also observed for the combined effect of both storage duration and temperature on the FDF where greater nitrification occurred at lower storage temperatures. Nitrification is important because plants most take up N from the soil in the form of NO_3-N. A combined effect of fertiliser type, storage duration and temperature on nutrient changes saw nitrification occurring in NECo fertiliser after 6 months of storage and at lower storage temperatures while NH_4-N and NO_3-N concentrations of Co did not significant change over the storage duration and temperature indicating a quite minimal N mineralisation of Co during storage.

Total P and K levels were affected by both temperature and duration of storage. There was a general increase in total P and K at lower temperatures of storage (5°C) compared to the ambient temperature condition of storage as well as increasing P and K concentration with longer duration of storage. This was contrary to the findings of Pawlowska et al. (2019) that, one year storage under different conditions did not significantly influence concentrations of P and K. The findings from this study clearly point to the fact that higher nutrient content and conservation were achieved at lower storage temperatures and with longer duration of storage. The heavy metal levels of the FDFs after the storage period of six months did not meet the ECN-QAS or Ecocert standards. This may imply that storage of FDF under different temperature conditions did not significantly affect heavy metal concentrations prior to storage. These elements find their way from the feedstock such as food waste or FS through contamination at the point of generation/containment. There must be mechanisms put in place along the sanitation value chain to minimize the contamination with potential sources of heavy metals.

In this study, we found that the interaction effect of all three factors (fertiliser type, storage temperature and duration) on pH and EC of FDF was not significant. While pH was on a slight increasing trend with longer duration of storage, there was however not a clear increasing or decreasing trend observed for EC with duration of storage. This trend with EC observed was similar to the findings of Kleawklaharn and Iwai (2014) who found out that, the chemical properties of the vermicompost were changing differently, and yet there was no clear trend in the changes of pH, EC, total potash and magnesium during the different duration (0, 1 and 3 months) of storage. The higher EC recorded for NECo than Co was due to the main factor effect of fertiliser type. The enrichment process (Adamtey et al., 2009) may have increased the EC of the NECo just like the enrichment also affecting the pH of the FDF by lowering the pH of the NECo (Dworkin & Gutnick, 2012).

5.5 Conclusion

The heavy metal levels in NECo following enrichment met the criteria set by ECN-QAS standard as safe compost/fertiliser while Co did not meet Cd and Cr criteria prior to the storage study. Storage temperature and duration of storage did not affect *E. coli* levels in any of the faecal derived fertilisers (FDFs) in this study indicating that re-growth of indigenous *E. coli* that had previously been treated below detection limits was unlikely barring extraneous contamination. Relatively higher NPK availability and conservation were achieved at the lower storage temperatures and with longer duration of storage implying that, longer storage periods do not necessarily influence losses in nutrient value, but the temperature of storage plays a bigger role.

5.6 References

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Chapter 6 The Direct and Residual Effect of Faecal Derived Fertiliser on Lettuce Nutrient Uptake, Quality, Yield and Soil Properties

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Abstract
Faecal derived fertilisers (FDFs) application on crops has positive effects on properties of soil, crop growth and yield. Earlier studies focused on a wide range of FDF uses including vegetable, cereal, and oil palm productions. In most cases, the direct mineralisation of one-off application of FDF were considered. This study evaluated the direct and residual effect of one-off application of FDF on soil nutrient, lettuce yield and nutrient uptake under field conditions for four successive cultivations in Ghana. A randomized complete block design was used for the field experiment. The fertiliser treatments included FDFs —co-compost (Co), N-enriched Co (NECo), mineral fertiliser as compared to a soil only control in 3 replications. The experiment was based on one-time application of fertiliser at a rate of 150 kg N/ha at the start of first cultivation period. Data on growth, yield parameters of lettuce and residual NPK in soil were collected. Results show that, for the direct effect (first cycle), the highest lettuce yield of 27.9 t/ha was recorded in Co fertilized plots followed by 11.1 t/ha for mineral fertilized plots. The residual effect of FDF improved lettuce yield by as much as 344% by the second cycle. *E. coli* was below the limit of detection on lettuce leaves after successive cultivations implying the safety of crops grown with FDFs. Findings from this study support the use of FDF as an alternative to mineral fertilisers with the added advantage of residual effects, which could be utilised for successive cropping.

Keywords: nutrient availability; crop contamination; soil pathogen; faecal sludge; co-compost; mineral fertiliser

6.1 Introduction

Matching temporal and spatial nitrogen (N) supply to demand in vegetable crops, without experiencing excesses or deficiencies is still a challenging task for growers (Di Gioia et al., 2017). Especially in the era of realisation that sources of mineral fertilisers such as phosphorus are finite, as well as geopolitical conflicts inducing high cost and acute shortages around the world (Cordell et al., 2012; Herebrand & Laborde, 2022). Secondly, what makes it more urgent to adopt circular economy for sustainability today is the convergence of several factors including climate change emergency, competition in the markets for natural resources and increasing pollution of soil and water environment from wastes (Smol, 2021; Velenturf et al., 2019; Vollaro et al., 2016). It has become imperative to explore alternative fertilising sources in the context of circular economy, to provide optimal N rate for crops that is safe. Faecal sludge (FS) has been touted and continues to be promoted as one of the useful materials for circular economy in developing countries (Semiyaga et al., 2015; S. Singh et al., 2017) because it contains abundant nutrient and organic matter, which if treated well could be used for safe crop production as an alternative or compliment to mineral fertilisers (O. Cofie et al., 2016; Otoo et al., 2015). This is even more relevant in the sub-Saharan African context where soils are inherently low in organic matter and nutrient depletion is high (Henao & Baanante, 2006; Tully et al., 2015). Proper and careful soil management as practiced under organic or ecological farming can reverse the decreasing trend in soil fertility and quality (Bhaskaran & Krishna, 2009; Sacco et al., 2015; Weber et al., 2007), and increase nutrient availability and crop yield (Nurhidayati et al., 2018).

Over the years, faecal derived fertilisers (FDFs) have been used to cultivate different types of crops e.g. vegetables and cereals (Magwaza, Magwaza, Odindo, Mashilo, et al., 2020; Nartey et al., 2017, 2021; Pradhan et al., 2016, 2019; Torgbo et al., 2018) with positive effects on the physico-chemical properties of soil, growth and yield of crops (Adamtey et al., 2010; B. Moya, Parker, Sakrabani, et al., 2019; Oyetunji et al., 2022; Sommer et al., 2013).The above studies assessed the direct effect of one-off application of FDF. Studies on the residual effect of FDF are few (Pradhan et al., 2019) and limited to two cycles of cultivations (Pradhan et al., 2019). Similarly, studies on safe use of FS and sewage sludge were focussed on composting (Hait & Tare, 2012; Koné et al., 2007; Krounbi et al., 2019; Nartey et al., 2017; Sreesai et al., 2013). The studies have demonstrated that, mature co-compost or FS is sanitized from pathogens (*Escherichia coli* O157:H7, Salmonella and helminths) with values below detectable levels and thus meet the threshold limit of various standards. However, some studies reported soil contamination with pathogens when treated with co-compost or composted FS (Major et al., 2020; Oliveira et al., 2011, 2012). Information on the medium-term residual effect (four or more successive cultivations) of one-off applications of FDF on soil available nutrients, crop nutrient uptake, yield, and soil properties and quality are crucial for sound decision making on its use and soil fertility and health management.

The study aimed at bringing the above-mentioned gaps using lettuce because it is widely cultivated (FAOSTAT, 2022) and consumed mostly raw. Intensive cultivation of the crop uses huge amounts of fertilisers and pesticides, which can lead to widespread contamination (Kundu & Mandal, 2009) and could post potential threat to producers, consumers and the environment (Kundu & Mandal, 2009). It is therefore important to provide alternatives inputs to improve the sustainability of such farming systems without compromising on yield and crop quality (Hernández et al., 2016). Therefore, this study aims to assess the suitability of FDF as an alternative to inorganic fertiliser in vegetable production. The objectives are to evaluate the direct and residual effect of a one-off application of FDF (under four successive cultivations) compared with mineral fertiliser on nutrient availability, crop nutrient uptake, yield, quality, and soil properties (chemical and pathogens). The study tested the hypothesis that, the direct effect from one-time application of FDF on crop and soil quality will differ from its residual effect.

6.2 Materials and Methods

6.2.1 Study site description

The study was carried out in field scale at Akorley, Somanya (latitude 6° 6′ 0″ N and 3° 3′ 0″ N and between longitude 0° 0′ 30″ W and 0° 0′ 10″ W, Sadiq, 2016) at the Jekora Ventures Limited (JVL) – Yilo Krobo Municipal Assembly (YKMA) Recycling Plant, in the YKMA of Ghana (Figure 6-1). The annual rainfall of the area ranges from 750 to 1,600 mm and it's spans from May to October (bimodal). Average temperatures range between 24 and 30°C while relative humidity ranges between 60 and

90% (Sadiq, 2016). The major soil type is Savanna Ochrosol (Eastern Regional Co-ord Council, 2016). It has low nutrient reserves, with the topsoil consisting of dark greyish brown humus sandy or clay loams (Eastern Regional Co-ord Council, 2016).

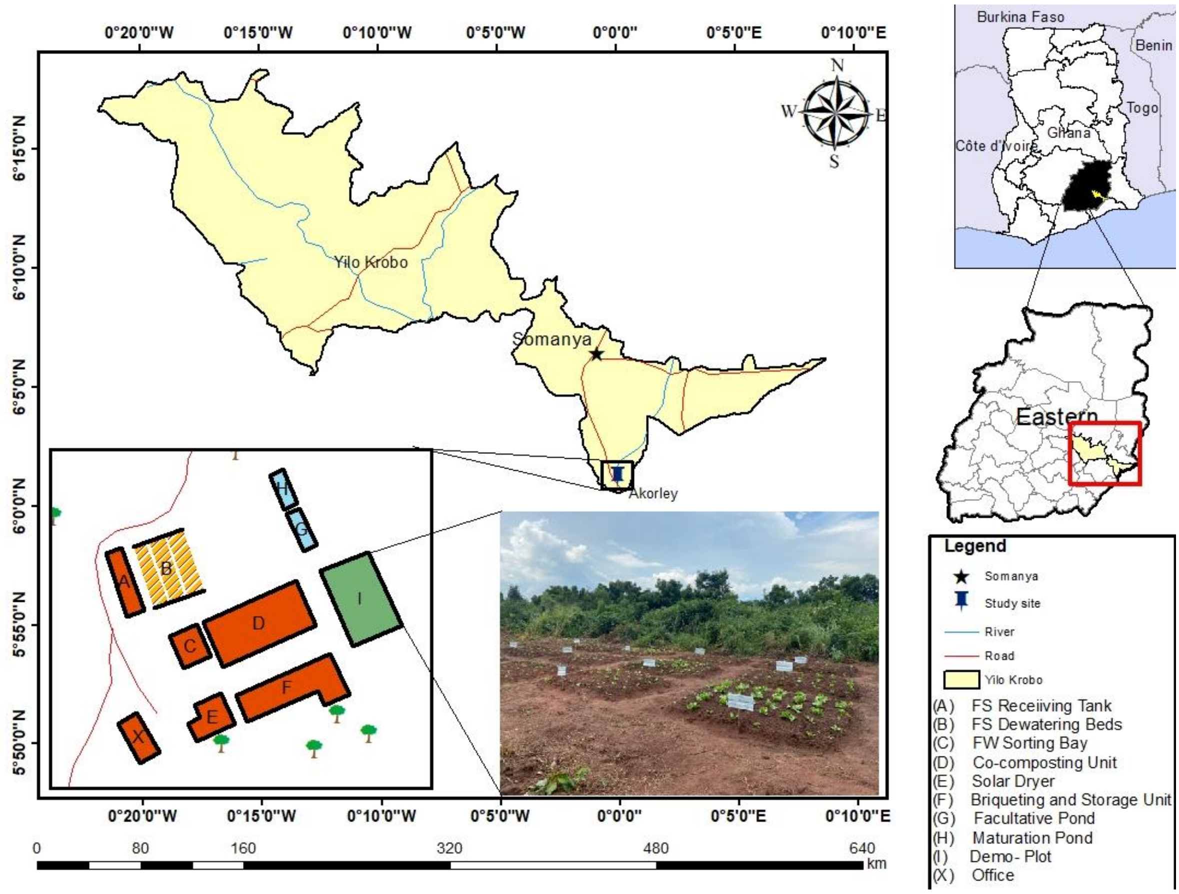

Figure 6-1. Map of study site

6.2.2 Soil characteristics, experimental design, and treatments

Initial soil samples were taken from a depth of 0 – 15cm from the study plots and analyzed for physico-chemical parameters (pH, EC, OC, NH_4-N, NO_3- N, N, P, K, Cu, Zn, Mg, Ca, Se, Fe, Cd, Cr IV, Hg, Ni and Pb) and pathogen parameters (*E. coli*, *total coliform*, helminth). The bulk density and water holding capacity (WHC) were also determined. Each experimental plot size was 2.4m x 1.8m covering a total area of 51.84m^2. The experimental plots were separated with paths of 0.6m.

The treatments, FDFs that is co-compost (Co) produced from food waste and dewatered FS and ammonium sulphate enriched co-compost (NECo) were obtained from the JVL - YKMA Recycling Plant in Somanya. The Co and the NECo were produced according to the methods and procedure described by Nartey et al. (2022) and Adamtey et al. (2009). The mineral fertilisers, ammonium sulphate and $NPK_{23\text{-}10\text{-}5}$ were purchased from the local agro-chemical shops. The characteristics of the fertiliser treatments are described in Table 6-1.

Table 6-1. Physico-chemical characteristics of Co, NECo and mineral fertilisers

Parameter	Co	NECo	**$NPK_{23\text{-}10\text{-}5}$**	**Ammonium Sulphate**	ECN-QAS	ECOCERT Standard

pH (1:5)	8.7^{a}	6.1^{b}	-	-	-	-
EC (1:10) (mS/cm)	5.3^{a}	5.6^{a}	-	-	-	-
Total N (%)	1.04^{b}	3.03^{a}	23.0	21.0	-	-
NH_4-N (%)	0.13^{b}	50.92^{a}	-	-	-	-
NO_3-N (%)	0.12^{b}	23.10^{a}	-	-	-	-
Org. C (%)	13.6^{b}	17.9^{a}	-	-	-	-
C:N ratio	13.3^{a}	5.9^{b}	-	-	-	-
Total P (%)	1.8^{a}	1.8^{a}	10.0	-	-	-
Avail. P (mg/kg)	6.5^{a}	4.0^{a}	-	-	-	-
Total K (%)	1.8^{a}	2.1^{a}	5.0	-	-	-
Avail. K (mg/kg)	0.14^{a}	0.02^{b}	-	-	-	-
Ca (%)	0.96^{a}	1.22^{a}	-	-	-	-
Mg (%)	0.33^{a}	0.35^{a}	-	-	-	-
Mn (mg/Kg)	14.9^{b}	27.3^{a}	-	-	-	-
Cu (mg/Kg)	6.2^{a}	7.9^{a}	-	-	300.0	70.0
Zn (mg/Kg)	531.5^{a}	567.2^{a}	-	-	600.0	200.0
Fe (mg/Kg)	289.3^{b}	362.7^{a}	-	-	-	-
Pb (mg/Kg)	66.7^{a}	35.7^{b}	-	-	40.0	25.0
Cd (mg/Kg)	6.1^{a}	1.3^{b}	-	-	1.3	0.7
Cr (mg/Kg)	66.3^{a}	29.6^{b}	-	-	60.0	70.0
Hg (mg/Kg)	0.29^{a}	0.13^{a}	-	-	0.45	0.4
Ni (mg/Kg)	0.01^{a}	0.02^{a}	-	-	-	-
As (mg/kg)	1.08^{a}	0.28^{b}	-	-	-	-
Se (mg/Kg)	0.02^{b}	0.04^{a}	-	-	-	-
E. coli (CFU/g)	$<1.0^{a}$	$<1.0^{a}$	-	-	$1.0 \times 10^{3*}$	
Total coliform (CFU/g)	9.2×10^{3a}	$<1.0^{b}$	-	-	$1.0 \times 10^{3*}$	-
Helminth eggs (Ova/10g)	$<1.0^{a}$	$<1.0^{a}$	-	-	-	-

ECN-QAS = European Compost Network-Quality Assurance Scheme. Sources: (Ecocert, 2013; Leifert, 2017), *USEPA part 503 class A standard (USEPA, 1993)

The above treatments were applied at the start of first cultivation period at a rate of 150 kg N/ha according to the recommendation of the Ministry of Food and Agriculture (MoFA) for lettuce (MOFA & MOAP, 2011). The treatments were applied in three replicates and arranged in a randomized complete block design (RCBD). The Co was incorporated into the 0 – 15cm topsoil a week before transplanting while the NECo and the mineral fertiliser ($NPK_{23\text{-}10\text{-}5}$ + ammonium sulphate in split) were applied a week after transplanting on the topsoil. This conforms to local smallholder farmers practise of applying organic and mineral fertilisers to vegetables in Ghana.

6.2.3 Nursery establishment, transplanting and crop management.

Lettuce seeds *(Lactuca sativa L. var. Great lakes)* were obtained from the local agro-chemical shop and sown in a nursery. Healthy and vigorous lettuce seedlings at true 4-leaf stage were transplanted after four weeks onto the experimental plots on raised beds at a spacing of 30cm × 30cm (Pradhan et al., 2019; Torgbo et al., 2018). The planting population was 48 plants/m^2 which amounted to 111,111 plants/ha. Lettuce was grown under rainfed with supplementary irrigation using handheld watering cans, which represent the most common irrigation method used in the area for leafy vegetable production. An integrated pest management approach such as pulling weeds before they establish roots and handpicking pest off plants was used to control weeds, and pests. Lettuce was harvested five weeks after transplanting. After harvest the plots were immediately prepared, and the second lettuce transplanting/cultivation was done within a week. The cultivations were repeated four times on the same experimental plots from June – December 2021 without additional fertiliser input. Care was taken not to contaminate the experimental plots during fertiliser application, irrigation, and husbandry practices by carefully washing hands, washing, and sanitizing the tools as well as the boots that were used in the field in and between each treatment plots.

6.2.4 Data collection

6.2.4.1 Lettuce nutrient uptake, growth, and yield parameters

Plant height (cm) and root length (cm) were measured with a ruler, number of leaves per plants, were measured by direct counting from 10 plants selected along the diagonals of each plot. Shoot fresh and dry weight (g), root fresh and dry weight (g) of the 10 selected plants were measured with a scale. The growth and yield parameters were measured at 5 weeks after transplanting. Dry weight was measured by drying fresh lettuce sample from each treatment in the oven at 65-70°C for about 48 hours until a constant weight was attained. Nitrogen uptake and utilisation efficiencies were calculated by methods described by Weih et al. (2018) and Congreves et al. (Congreves et al., 2021)

6.2.4.2 Pathogen analyses of lettuce and soil

Six lettuce heads were randomly selected from each plot and collected with sterile disposable gloves just before harvesting and were put into separate sterile polythene bags and transported on ice to the laboratory for analyses of *E. coli, total coliform,* and helminth eggs using the procedure and methods described by (APHA-AWWA-WEF, 2001; Schwartzbrod & Gaspard, 1998). Soil samples from each plot were also aseptically taken from points along the diagonals of each plot and bulked into composite samples. A grab sample was taken after every cropping cycle and analysed for the same microbial parameters as described above for the lettuce.

6.2.4.3 Compost and Residual assessment of soil properties

After every cycle of harvest, soil samples were taken from the topsoil (0 – 15cm) with an auger from each plot by aseptically taking soil from points along the diagonals of each plot and bulked into composite samples. A grab sample was then taken from the composite after every cropping cycle and analysed. N, P, K and bulk density were determined by methods described in Okalebo et al. (2002). pH and EC were measured using 1:5 and 1:10 compost: water w/v ratios, respectively described in (USDA and USCC, 2001). Organic carbon (OC) was determined by the Walkley and Black (1934) method. Water holding capacity were determined according to methods described in Okalebo et al. (2002) and Vengadaramana and Jashothan (2012), respectively. The heavy metals were analysed by atomic absorption spectrophotometer following methods described by Chapman and Pratt (1962). Exchangeable cations (Ca, Mg, Na and K), effective cation exchange capacity (ECEC), base salts, exchangeable acidity, and total exchangeable bases (TEB) were determined by methods described in Okalebo et al. (2002).

6.2.5 Statistical analysis

Data collected were analysed using Genstat statistical package (12^{th} edition). Simple ANOVA and two-way ANOVA were used to analyse treatment and cultivation cycle effect on growth, yield, nutrient uptake, nutrient recovery, and pathogens in lettuce. Same analyses were carried out on the soil parameters. Treatment means found to be significantly different from each other at ($P < 0.05$) were separated by Bonferroni tests.

6.3 Results and Discussion

6.3.1 Direct and residual effect of fertilisers on lettuce yield and growth parameters

6.3.1.1 Direct and residual effect on lettuce yield

Farmers in SSA conventionally use organic inputs in the form of manure and co-compost for fields and manually incorporate into the soils at least a week before planting. This is smallholder farmers' practise which was employed in this study under field conditions. Co-compost was applied at least a week before transplanting among other things to allow some form of nutrient mineralisation to take place and to make the needed nutrients readily available to the crops when planted. Mineral fertilisers were applied a week after planting to ensure that the plants were well established to utilize the nutrients to avoid losses. Where combined organic and mineral fertilizations are used, it usually follows the same convention of organic first before planting and mineral fertiliser after planting. The timing of these fertiliser applications has implications for transplant survival and for mineralisation and nutrient availability in soils for the crops as seen in this study. The mean growth parameters for the lettuce transplants used for the different cultivation cycles in this study is shown in Table 6-2. The transplants had on average four leaves and were 6.6 cm tall. The fresh shoot weight was 2.2 g per plant.

Table 6-2. Vegetative growth properties of lettuce transplants for the successive cultivations

Transplants	Plant height/ plant (cm)	Number of leaves/ plants	Fresh shoot weight /plant (g)	Dry shoot weight/ plant (g)	Fresh root weight/ plant (g)	Dry root weight/ plant (g)
1st cultivation	6.5	4.1	2.2	0.27	0.09	0.02
2nd cultivation	6.7	3.5	2.0	0.24	0.07	0.02
3rd cultivation	6.6	3.7	2.2	0.25	0.10	0.02
4th cultivation	6.5	3.5	2.2	0.25	0.09	0.02
Mean ± SD	6.6 ± 1.0	3.7 ± 0.3	2.2 ± 0.1	0.25 ± 0.01	0.09 ± 0.01	0.02 ± 0.00

The main fertiliser and cultivation cycle treatment effect on lettuce yield is described in Table 6-3. There was no difference in yield between Soil + NECo and Soil + mineral fertiliser at $p<0.01$ and $p<0.05$, meaning that NECo provided similar nutrient availability as mineral fertiliser. All the other differences in yield were however significant for the fertiliser treatment. In terms of cultivation cycles, the differences in yield were not significant between the third and fourth cycles. This could be an indication of the remaining available nutrients becoming limiting after the third successive cultivations in the various plots.

Table 6-3. Main fertiliser and cultivation cycle treatment effects on lettuce yield.

Treatment effect	Yield (t/ha)
Fertiliser	
Soil + Co	14.3^{a}
Soil + NECo	6.9^{b}
Soil + Mineral	5.1^{bc}
Soil only	3.0^{c}
Cultivation cycle	
First cycle	13.0^{a}
Second cycle	9.7^{b}
Third cycle	4.2^{c}
Fourth cycle	2.3^{c}

The interaction effect of fertiliser and cultivation cycle on direct and residual lettuce yield is shown in Figure 6-2. The direct effect on yield (first cycle) was highest (27.9 t/ha) in the Co fertilised plots followed by 11.1 t/ha for mineral fertilised plots (Figure 2). This difference in yield output could be attributed to improvement of the physical properties of the soil (lower bulk density) and timing of Co application. Co was applied a week before transplanting which probably enhanced mineralisation to release nutrients for lettuce roots to access. This is because, the Co was able to supply sufficient nutrients and right soil structure enabling superior yields over the mineral fertilised plots (Pradhan et al., 2019; Torgbo et al., 2018). The mineralisation process avails valuable micronutrients, which help to coordinate a range of physiological functions and promote growth of earthworms and other beneficial soil organisms including nitrifying bacteria (Kästner & Miltner, 2016; Ren et al., 2018). This finding is contrary to the findings of Hernandez et al (2016) who observed that, the first crop lettuce yields in the compost-treated soils did not significantly ($p<0.05$) differ from the yields of mineral fertiliser treated soils.

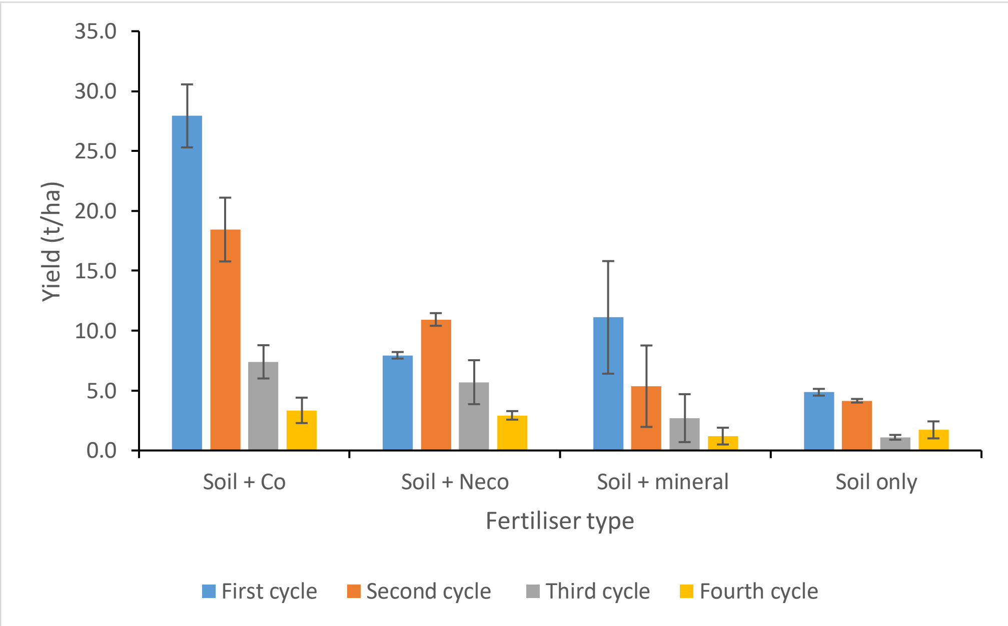

Figure 6-2. Interaction between fertiliser and cultivation cycle on lettuce yield

In NECo fertilised plots, the lower direct yield was contrary to expectation. This might have been due to delay in nutrient release rate following the timing of NECo application (a week after transplanting). The organic matter may have immobilised the available nutrients from the enrichment making NECo not a fast nutrient releaser as anticipated. According to Abedi et al. (Abedi et al., 2010), compost application slowly releases nutrients and prevents nutrient losses from mineral fertilisers by binding to nutrients and releasing with time. This finding is contrary to findings by Nartey et al. (2021) where higher direct yield of pepper and tomato were recorded for NECo formulations. However, this could also be influenced by crop under consideration in that lettuce is a short duration crop compared to tomato or pepper. The highest lettuce yield recorded in this study for the direct effect was higher than the average lettuce yield of 15 – 20 t/ha of this variety in Ghana as reported by MOFA-MOAP (2011).

The yield of lettuce differed significantly ($p<0.05$) between various fertiliser treatments for the residual effect (second, third and fourth cultivation) (Figure 6-2). The residual effect on second lettuce yield showed 33.4%, 51.3% and 11.0% drop in yields for Co, mineral and soil only, respectively. However, there was a 48.6% increase in yield in the NECo fertilised plots in the second cycle. The increase can be attributed to the release of nutrient from the mineral enrichment and mineralisation of nutrients from the compost which were immobilised in the first cycle (Seran et al., 2011; Suge et al., 2011). Pradhan et al. (2019) in their study of FDF effect on lettuce also recorded higher yields in lettuce in the Co treatment followed by NECo for the first cycle but, in their second cycle, only control, mineral and poultry manure fertilised plots recorded increases in yield. Hernandez et al. (2016) found lettuce yields in the compost-treated soils did not differ from the yields of mineral fertiliser treated soils in the first cropping. In the second crop, however, lettuce yields were higher in organically treated soils than in soils receiving only mineral fertilisation.

In this study, it was observed that applying FDF at the recommended rate gave between 163.6% - 344.4% increase in lettuce yield in the first residual effect (second cycle) compared to the control (Soil only) while the mineral fertilised recorded 29.3% increase in yield for the first residual effect. This showed that FDF provided lettuce with all the nutrients needed particularly N to grow. Hernandez et al. (2016) observed similar findings in their second cropping, where lettuce yields were higher in compost treated soils than in soils receiving only mineral fertilisation. The third and fourth cycle yields showed an overall decreasing trend in lettuce yield for all the treatments, though yields were significantly ($p<0.05$) higher in the Co fertilised plots (Figure 6-2). Similar trends were reported by Nurhidayati et al. (2018) for various vermicompost used to cultivate mustard over successive cycles. The results from this study showed 59.8%, 46.6%, 53.3%, and 73.6%, 53.3%, 46.6% and 59.8% further drop in third cycle yields for Co, NECo, mineral and soil only fertilised plots, respectively from the second cycle. At least 50% yield loss was observed after each cultivation cycle in the mineral fertilised plots. For the third cycle, the Co fertilised plots gave 586.1% increase in lettuce yield compared with control (soil only) while the NECo and mineral fertilised plots gave 430.9% and 151.9%, respectively providing evidence that farmers can benefit from the residual effect of FDF even after the third cycle. The losses in yield for the soil only plot was not significantly different between the first and second cycles but was significantly reduced in the third and fourth cycles.

This could be due to depletion of nutrients in the soil after sustaining growth of the lettuce for more than two cycles without replenishment. Results from this study show that there is residual nutrients, which could be utilised for the next cropping. Therefore, farmers who could not afford to buy or do not have steady access to fertiliser every cropping season could improve productivity and income by as much as 344.4% by applying FDF on lettuce after two cropping cycles without additional fertilisation. These results are in agreement with other results on cereals such as rice (Sarwar et al., 2007), wheat (Abedi et al., 2010; Sarwar et al., 2007) and sorghum (Ouédraogo, 2001).

6.3.1.2 Direct and residual effect on growth parameters of lettuce

The main effect of fertiliser on the growth parameters of lettuce after four cycles of cultivation is shown in Table 6-4. Root length was significantly longest in the soil + Co treatment at 9.1 cm compared to the other treatments. This could be explained by the high nutrient release rate of Co into the soil enticing root development as the roots explore different parts of the soil for nutrients and the reduced bulk density of the soil. The availability of nutrients in the soil made it worth investing for the roots to reach deeper. There was no significant difference in root length of lettuce between NECo and mineral fertiliser amended soils (Table 6-4), indicating that the NECo was comparable in terms of root development to the mineral fertiliser treatment. Lettuce height and fresh shoot weight did not significantly differ between the soil only and mineral fertilized soils (Table 6-4). There was a positive correlation between root length and root dry matter (0.59) as well as shoot dry matter (0.68), lettuce height (0.71) and number of leaves (0.67).

For the main effect of cultivation cycles, root length was not significantly different between the first, third and fourth cycles (Table 6-4). There was a general decreasing trend in values of all growth parameters measured with increasing cultivation cycles except for lettuce height, No. of leaves, root length and fresh root weight. Lettuce height was not significantly different between the second and third cycles. Number of leaves and fresh root weight of the second cycle was not significantly different from the first cycle. The interaction effect between fertiliser treatment and cultivation cycle on lettuce growth parameters showed that, plant height did not significantly differ between the first three cycles for the soil only treatment (Table 6-4). The number of leaves were significantly lower in the third cycle for all the fertiliser treatments except for NECo amended treatment. The leaf area was not significantly different between the second and third cycle except for Co amended soils. The below ground parameters show root length to be highest generally in Co amended soils for all cycles. The NECo amended soils showed significant increase in root length in the third cycle which was comparable to the Co amended ones. This meant that the roots in those treatments were able to extend deeper into the soil to reach more water and nutrients. This has reflected in the no significant difference in fresh and dry root weight in the third cycle between NECo and Co amended soils.

Enhanced food production in SSA is critically dependent on external nutrients inputs especially N and P (Chivenge et al., 2011). While mineral fertilisers are widely used globally to overcome nutrient deficiencies, their use remains very low in SSA with average application rates of 8 kg/ha/yr which is less than one tenth of the world average (Wanzala & Groot, 2013). Kenya, Ghana and Madagascar use 26, 15 and 4 kg/ha respectively (NEPAD-CAADP, 2015) indicating a wide range of fertiliser application from various parts of Africa. Reasons for this scarce use of fertiliser is its low availability and the high cost associated with it. For example in Ghana, mineral fertiliser shortages have been reported in various parts of the country in 2021 (Apubeo, 2021; Ashon, 2021). Organic resources, ranging from animal manures, household composts, crop residues, leguminous cover crops, to leguminous and non-leguminous trees and shrubs, are often used as major nutrient sources to crops (Chivenge et al., 2011). The findings of this study have supported the case for FDF to be used as an alternative and/or compliment to mineral fertilisers with the added advantage of residual effects, which could be utilised for successive cropping. This could contribute to enhancing fertiliser availability and affordability for smallholder farmers. Smallholder farmers who could not afford to buy and apply fertiliser every cropping season could improve productivity and income by utilising the residual effect.

Table 6-4. Main effects of fertiliser, cultivation cycle and their interaction effects on growth properties of lettuce at 5WAT.

Treatment effect	**Plant height (cm)**	**No. of leaves**	**Leaf area (cm^2)**	**Root length/ plant (cm)**	**Fresh shoot weight/ plant (g)**	**Fresh root weight/ plant (g)**	**Dry shoot weight/ plant (g)**	**Dry root weight/ plant (g)**
<u>Fertiliser</u>								
Soil + Co	15.8^{a}	13.0^{a}	209.8^{a}	9.1^{a}	121.9^{a}	3.5^{a}	13.5^{a}	0.6^{a}
Soil + NECo	11.9^{b}	8.2^{b}	107.5^{b}	6.6^{b}	65.3^{b}	1.8^{b}	6.4^{b}	0.4^{ab}
Soil + Mineral	10.1^{c}	7.0^{b}	89.8^{bc}	5.7^{bc}	39.6^{c}	1.2^{bc}	4.5^{bc}	0.3^{bc}
Soil only	9.8^{c}	6.7^{b}	73.3^{c}	5.6^{c}	26.4^{c}	1.2^{c}	2.7^{c}	0.2^{c}
<u>Cultivation cycle</u>								
First cycle	14.4^{a}	9.4^{a}	199.4^{a}	7.1^{a}	117.1^{a}	2.7^{a}	11.9^{a}	0.6^{a}
Second cycle	11.6^{b}	9.9^{a}	127.5^{b}	5.9^{a}	82.6^{b}	2.7^{a}	7.5^{b}	0.4^{b}
Third cycle	12.6^{b}	6.9^{a}	93.7^{c}	7.0^{a}	32.9^{c}	0.9^{b}	4.2^{c}	0.3^{b}
Fourth cycle	9.1^{c}	8.7^{b}	59.9^{d}	7.0^{b}	20.6^{c}	1.4^{b}	3.6^{c}	0.3^{b}
<u>Fertiliser x Cultivation cycle</u>								
Soil + Co x First cycle	18.9^{a}	13.2^{b}	391.6^{a}	10.7^{a}	238.0^{a}	5.1^{a}	28.5^{a}	0.6^{abc}
Soil + NECo x First cycle	13.4^{d}	7.5^{de}	131.3^{cd}	5.5^{ef}	99.0^{c}	2.1^{bc}	6.7^{cde}	0.5^{abcd}
Soil + Mineral x First cycle	13.9^{cd}	9.2^{cd}	162.7^{c}	7.1^{cd}	82.0^{cd}	2.0^{bc}	8.4^{cd}	0.8^{a}
Soil only x First cycle	11.3^{ef}	7.9^{de}	111.8^{def}	5.1^{f}	49.5^{ef}	1.6^{cd}	4.0^{efgh}	0.4^{cde}
Soil + Co x Second cycle	16.5^{b}	15.5^{a}	219.6^{b}	8.1^{bc}	152.7^{b}	4.9^{a}	12.2^{b}	0.7^{ab}
Soil + NECo x Second cycle	11.0^{efg}	8.7^{cd}	114.5^{def}	5.6^{ef}	96.9^{cd}	2.7^{b}	9.2^{bc}	0.5^{bcd}
Soil + Mineral x Second cycle	9.8^{fgh}	8.1^{cd}	90.3^{defg}	5.0^{f}	47.6^{efg}	1.7^{cd}	5.6^{defg}	0.2^{ef}
Soil only x Second cycle	9.1^{gh}	7.2^{def}	85.8^{efg}	4.9^{f}	33.1^{fghi}	1.6^{cd}	3.1^{fgh}	0.2^{ef}
Soil + Co x Third cycle	15.6^{bc}	10.3^{c}	112.6^{def}	8.7^{b}	66.7^{de}	1.6^{cd}	7.5^{cde}	0.5^{abcd}
Soil + NECo x Third cycle	13.9^{cd}	7.7^{de}	120.4^{de}	8.2^{bc}	39.0^{efgh}	1.0^{def}	5.1^{defgh}	0.4^{cd}
Soil + Mineral x Third cycle	10.3^{fg}	5.0^{fg}	77.9^{fg}	5.0^{f}	18.3^{ghi}	0.4^{f}	2.2^{fgh}	0.2^{f}
Soil only x Third cycle	10.6^{efg}	4.7^{g}	64.0^{gh}	6.0^{def}	7.7^{i}	0.4^{f}	1.7^{h}	0.2^{f}
Soil + Co x Fourth cycle	12.2^{de}	13.0^{b}	115.5^{def}	8.8^{b}	30.2^{fghi}	2.2^{bc}	5.7^{cdef}	0.5^{bcd}
Soil + NECo x Fourth cycle	9.4^{gh}	9.0^{cd}	63.9^{gh}	7.1^{cd}	26.4^{fghi}	1.4^{cde}	4.7^{efgh}	$0.^{3def}$
Soil + Mineral x Fourth cycle	6.6^{i}	5.7^{efg}	28.4^{h}	5.8^{def}	10.6^{hi}	0.7^{ef}	1.7^{h}	0.1^{f}
Soil only x Fourth cycle	8.3^{hi}	7.0^{def}	31.6^{h}	6.4^{de}	15.5^{hi}	1.0^{def}	2.1^{gh}	0.2^{ef}

6.3.1.3 Nitrogen uptake, and utilisation efficiency over successive lettuce cultivations

The highest nitrogen uptake by lettuce in all fertiliser treatments were observed for the direct effect (first cycle) (Table 6-5) and might be due to increased availability of nitrogen arising from the fertiliser application which gave higher lettuce yield. The residual availability of nitrogen in the soil decreased over succeeding cycles (second, third and fourth cycles) generally in all treatments as reflected in the decreasing N uptake. This meant that the residual availability of nitrogen was mostly from mineralisation of total N remaining in the soils after the initial fertiliser application. Since mineralisation of nutrients is expected to be slower in organic fertilisers and for that matter in FDFs, relatively higher N uptake were expected for the NECo and Co fertilised plots than in the mineral fertiliser and soil only plots. These were confirmed in this study in Table 6-5. As a concept, the nitrogen use efficiency (NUE) is expressed as a ratio of output (total plant N, rain N, biomass yield, grain yield) and input (total N, soil N or N-fertiliser applied) (Masclaux-Daubresse et al., 2010). NUE of a crop is considered as the product of two components: i) the N uptake efficiency (NupE), that expresses the ability of a crop to acquire nitrogen from the soil; and ii) the N utilization efficiency (NutE), that expresses the ability of a crop to use the N taken up to produce plant biomass (Di Gioia et al., 2017).

In this study, the NupE for the direct effect of applied fertiliser (first cycle) ranged between 8 – 36%. The NupE was generally higher in the successive second and third cycles with a decline in the fourth cycle compared to the direct effect for all treatments except the soil only (control) (Figure 6-3). This could mean that there was better NUE observed in the residual fertiliser effect on the lettuce yield requiring lower or no additional fertiliser input. This is both important to protect the environment from excessive application of N-fertiliser and improve sustainable and productive agriculture (Masclaux-Daubresse et al., 2010). The highest NupE were observed for lettuce cultivated in the Co fertilised plots. For the direct effect, the lowest NupE observed in cultivated lettuce were 9% and 8% for the mineral fertiliser and NECo fertilised plots, respectively. These were lower than the average 37% and 22% NupE reported by Di Gioia et al. (Di Gioia et al., 2017) for two cultivars of lettuce. However, the NupE of 36% and 25% obtained for Co and soil only fertilised plots, respectively for the direct effect were similar to those observed by Di Gioia et al. (Di Gioia et al., 2017) and Di Mola et al. (2020) confirming that lettuce crops are characterised by a low efficiency in recovering applied N.

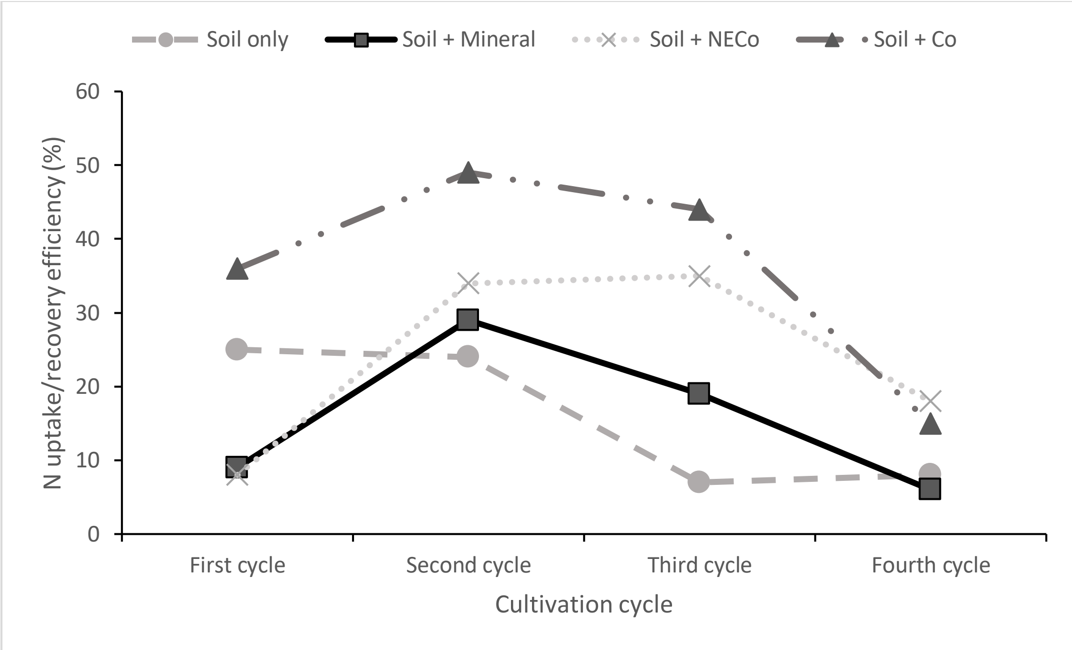

Figure 6-3. Nitrogen uptake/recovery efficiency (NupE) in successive lettuce cultivations

The nitrogen utilization efficiency (NutE) showed an increasing trend with the residual fertiliser effect on lettuce (Table 6-5), indicating that, improved N utilization was achieved with residual effect than with the direct effect (first cycle). This could be explained by the increasing lettuce root length observed for the residual effect allowing for more uptake of water and nutrients to be converted into shoot biomass.

Table 6-5. Nitrogen uptake and utilization efficiency in successive lettuce cultivations

Treatment	**Cultivation cycle**	**Dry yield (t/ha)**	**Nitrogen uptake in yield (Kg/ha)**	**Nitrogen utilization efficiency (NutE)**
Soil + Co	First cycle	3.17	65.2	48.6
	Second cycle	1.35	14.9	90.5
	Third cycle	0.83	9.3	89.4
	Fourth cycle	0.64	4.2	151.6
Soil + NECo	First cycle	0.75	13.6	54.9
	Second cycle	1.02	10.7	95.2
	Third cycle	0.57	5.7	99.4
	Fourth cycle	0.52	2.9	177.8
Soil + Mineral	First cycle	0.93	16.4	56.8
	Second cycle	0.62	8.1	76.9
	Third cycle	0.25	3.4	73.1
	Fourth cycle	0.19	1.9	99.5
Soil only	First cycle	0.44	7.5	59.0
	Second cycle	0.34	3.1	111.2
	Third cycle	0.19	2.3	85.4
	Fourth cycle	0.24	2.3	101.6

6.3.2 Soil properties after successive cultivation

6.3.2.1 Soil properties before and after cultivations

Some properties of the soils and the different fertiliser types at the beginning of lettuce cultivation are described in Table 6-6. Generally, there was no significant difference in the exchangeable cations, CEC and water holding capacity (WHC) of the different soil treatments. The mineral fertilized plots were more acidic than the FDF applied soils. The EC was significantly higher in the NECo plots than the mineral fertilized and soil only plots. *E. coli* was not significantly different between the treatments.

Table 6-6. Some physico-chemical and microbial properties of soil treatments at the start of cultivation.

Parameter	Soil + Co	Soil + NECo	Soil + Mineral	Soil only
pH (1:5)	6.7^{b}	5.8^{ab}	5.7^{a}	6.6^{ab}
EC (1:10) (ms/cm)	0.3^{ab}	0.6^{b}	0.2^{a}	0.1^{a}
Bulk density (g/cm³)	1.2^{a}	1.2^{a}	1.2^{a}	1.4^{b}
WHC (%)	71.3^{a}	74.3^{a}	74.0^{a}	57.7^{a}
Total N (%)	0.08^{a}	0.09^{ab}	0.11^{ab}	0.14^{b}
Org. C (%)	0.9^{ab}	0.9^{ab}	1.2^{b}	0.7^{a}
OM (%)	1.5^{ab}	1.5^{ab}	2.1^{b}	1.2^{a}
Ex. Acidity (meq/100)	0.1^{a}	0.1^{a}	0.2^{a}	0.2^{a}
Base salts (%)	98.5^{a}	98.6^{a}	97.6^{a}	98.0^{a}
Ca (meq/100g)	6.7^{a}	5.4^{a}	6.5^{a}	4.9^{a}
K (meq/100g)	0.2^{a}	0.5^{b}	0.2^{a}	0.1^{a}
Mg (meq/100g)	0.4^{a}	0.7^{a}	0.5^{a}	2.3^{b}
Na (meq/100g)	0.03^{a}	0.04^{a}	0.01^{a}	0.07^{a}
ECEC (meq/100g)	7.4^{a}	6.7^{a}	7.4^{a}	7.6^{a}
TEB (meq/100g)	7.3^{a}	6.6^{a}	7.2^{a}	7.5^{a}
E. coli (log units/g)	0.8^{a}	0.0^{a}	0.5^{a}	0.0^{a}
Total coliform (log units/g)	4.8^{b}	4.2^{ab}	3.5^{a}	4.9^{b}
Helminth eggs (Ova/10g)	0.0^{a}	0.0^{a}	0.0^{a}	0.0^{a}

WHC = Water holding capacity

The variables of soil nutrient showed that the different fertiliser treatment and cultivation cycles did not significantly influence the soil nutrient characteristics (Table 6-7). This was probably due to the short duration of the cultivation cycles of lettuce. The finding in this study is contrary to the findings of Nurhidayati et al. (2018) and Hernandez et al. (2016) who found that different fertiliser treatment significantly influenced soil nutrient and organic matter content. However, in this study, the residual effect of the treatments significantly affected the soil bulk density, water holding capacity and ECEC (Table 6-7). The bulk densities were lower in the FDFs in all the cultivation cycles indicating higher soil porosity and lower soil compaction. The differences in lettuce yield observed for the various treatments were supported by the significant differences in bulk density, water holding capacity and ECEC which affected the lettuce root growth, availability of water and the ability of the roots absorb or take up nutrients readily available in the soil.

Table 6-7. Effect of cultivation cycle on some soil properties after lettuce cultivation.

Treatment	pH	Bulk density (g/cm^3)	WHC (%)	N (%)	OM (%)	Avail P (mg/kg)	K (meq/100g)	Mg (meq/100g)	Ca (meq/100g)	TEB (meq/100 g)	ECEC (meq/100 g)
					After first cycle						
Soil + Co	6.2^{ab}	1.4^{bcdef}	30.0^{ab}	0.09^{a}	1.3^{a}	41.2^{d}	0.12^{a}	0.41^{a}	7.0^{e}	7.51^{cd}	7.6^{cde}
Soil + NECo	6.2^{ab}	1.4^{cdefg}	29.0^{a}	0.14^{a}	1.3^{a}	12.5^{abc}	0.53^{c}	0.77^{ab}	6.5^{de}	7.30^{bcd}	7.4^{bcde}
Soil + Mineral	5.6^{ab}	1.4^{defg}	33.0^{ab}	0.15^{a}	1.4^{a}	16.0^{abc}	0.27^{abc}	0.49^{a}	6.2^{de}	7.01^{bcd}	7.3^{bcde}
Soil only	5.9^{ab}	1.4^{defg}	28.0^{a}	0.14^{a}	1.2^{a}	2.6^{a}	0.37^{abc}	0.40^{a}	4.2^{abc}	4.98^{a}	5.0^{a}
					After second cycle						
Soil + Co	6.1^{ab}	1.2^{a}	57.7^{h}	0.13^{a}	1.4^{a}	42.1^{d}	0.25^{abc}	0.51^{a}	7.3^{e}	8.08^{d}	8.2^{e}
Soil + NECo	5.9^{ab}	1.3^{abc}	51.0^{g}	0.14^{a}	1.4^{a}	12.6^{abc}	0.42^{bc}	1.27^{abc}	5.8^{cde}	7.53^{cd}	7.9^{e}
Soil + Mineral	5.4^{a}	1.3^{ab}	47.7^{fg}	0.15^{a}	1.5^{a}	17.5^{bc}	0.29^{abc}	0.92^{abc}	6.4^{de}	7.62^{cd}	7.7^{de}
Soil only	5.8^{ab}	1.4^{bcde}	39.3^{cd}	0.15^{a}	1.3^{a}	2.9^{ab}	0.42^{bc}	0.48^{a}	4.7^{abcd}	5.66^{abc}	5.7^{abc}
					After third cycle						
Soil + Co	6.6^{b}	1.4^{bcdef}	52.3^{g}	0.14^{a}	1.4^{a}	19.1^{c}	0.14^{ab}	1.48^{bc}	4.1^{abc}	5.72^{abc}	5.9^{abcd}
Soil + NECo	6.5^{b}	1.4^{cdefg}	44.7^{ef}	0.12^{a}	0.9^{a}	19.1^{c}	0.26^{abc}	1.06^{abc}	5.3^{bcde}	6.67^{abcd}	6.8^{abcde}
Soil + Mineral	6.2^{ab}	1.4^{efg}	35.0^{bc}	0.14^{a}	1.4^{a}	19.1^{c}	0.23^{ab}	1.01^{abc}	4.9^{abcd}	6.10^{abcd}	6.2^{abcde}
Soil only	6.4^{ab}	1.5^{fg}	29.3^{a}	0.13^{a}	1.4^{a}	19.1^{c}	0.18^{ab}	1.81^{c}	3.3^{a}	5.29^{ab}	5.4^{ab}
					After fourth cycle						
Soil + Co	6.2^{ab}	1.3^{abcd}	52.0^{g}	0.15^{a}	1.5^{a}	46.5^{d}	0.22^{ab}	1.83^{c}	4.7^{abcd}	6.75^{abcd}	6.9^{abcde}
Soil + NECo	6.6^{b}	1.5^{g}	41.0^{de}	0.13^{a}	0.8^{a}	13.5^{abc}	0.21^{ab}	1.20^{abc}	5.4^{bcde}	6.90^{abcd}	7.0^{abcde}
Soil + Mineral	5.8^{ab}	1.5^{g}	30.7^{ab}	0.15^{a}	1.5^{a}	18.5^{c}	024^{abc}	1.10^{abc}	5.0^{abcd}	6.35^{abcd}	6.5^{abcde}
Soil only	5.7^{ab}	1.9^{h}	29.0^{a}	0.36^{b}	1.3^{a}	3.4^{ab}	0.22^{ab}	1.65^{bc}	3.6^{ab}	5.45^{ab}	5.6^{ab}

6.3.3 Hygiene characteristics of lettuce and soil following direct and residual effect

6.3.3.1 Hygiene of lettuce

In this study, *E. coli* presence and concentration on lettuce was not statistically significant among the treatment and cultivation cycles. *E. coli* was not detected on lettuce in the Co, NECo and mineral fertiliser treatments but was detected in the soil only treatment (control) at mean concentration of 0.2 log units/100g (Table 6-8). *E. coli* was detected during the second cultivation cycle only indicating possible one-time contamination probably introduced by a field worker or wild animals. The findings also show that, lettuce cultivated with FDF after successive cultivations was devoid of *E. coli* at the farm level. Helminth eggs were not detected in any of the treatments. Total coliforms presence and concentrations were statistically significant at $p<0.05$ for the main effects of fertiliser and cultivation cycles. Total coliforms were present on lettuce from all fertiliser treatments and in all cycles. In terms of main effect of fertiliser, the concentrations were highest (4.8 log units/100g) on lettuce from mineral amended plot (Table 6-8). The cultivation cycle effect on levels of total coliform saw a declining trend for first, second and third cycles with a high rise to 5.6-log unit/100g in the fourth cycle (Table 6-8).

Table 6-8. Pathogen contamination on lettuce after direct and residual harvest

Treatment	***E. coli*** **(log units/100g)**	**Total coliform (log units/100g)**	**Helminth eggs (ova/100g)**
Fertiliser			
Soil + Co	0.0^{a}	4.4b	$<1.0^{a}$
Soil + NECo	0.0^{a}	4.4b	$<1.0^{a}$
Soil + Mineral	0.0^{a}	4.8a	$<1.0^{a}$
Soil only	0.2^{a}	4.5b	$<1.0^{a}$
Cultivation cycle			
First cycle	0.0^{a}	4.5b	$<1.0^{a}$
Second cycle	0.2^{a}	4.2b	$<1.0^{a}$
Third cycle	0.0^{a}	3.9c	$<1.0^{a}$
Fourth cycle	0.0^{a}	5.6a	$<1.0^{a}$

The interaction effect between fertiliser and cultivation cycle treatment on total coliform levels on lettuce is shown in Figure 4. The highest levels of total coliform contamination were observed in the fourth cycle of harvest, and these were not significantly different between the fertiliser treatments. It is worth noting that total coliforms are commonly found in the environment and as such are not directly attributable to faecal contamination.

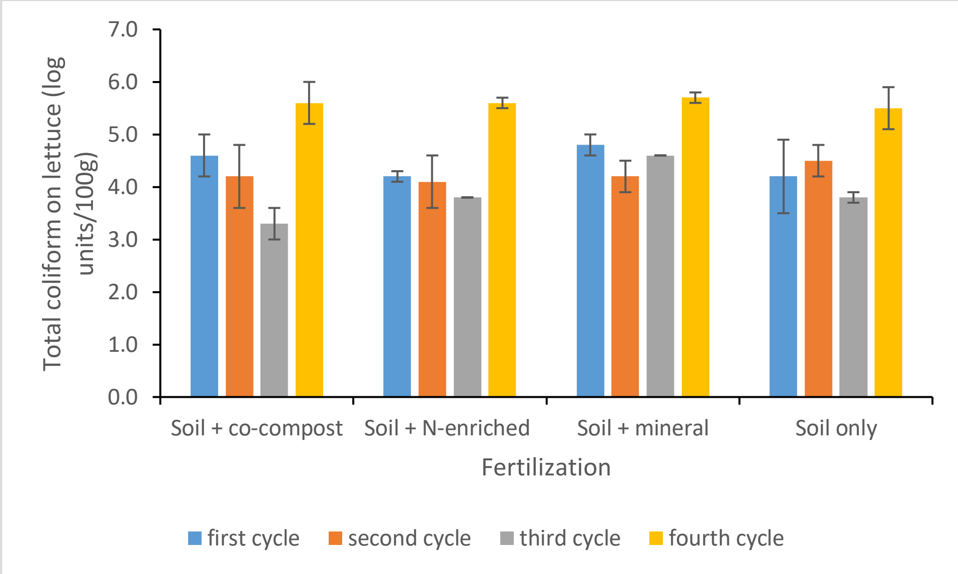

Figure 6-4. Interaction effect between fertiliser and cultivation cycle on lettuce total coliform concentration

6.3.3.2 Hygiene of treatment plots after cultivation

E. coli was present in the soils of the fertiliser treatment plots after the first and second cultivation cycles of lettuce (Figure 6-5). *E. coli* concentrations were not significantly different at $P \leq 0.05$ between the soils. In the first cycle, *E. coli* was found in the Soil + Co and Soil + mineral plots only, however after the second cycle cultivation the levels increased with the highest of 1.9 log units/g being found in the soil+ Co plots and the lowest in the soil only plots (Figure 6-5). The presence and increase *E. coli* in the FDF treated soils could be due to high water holding capacity of the media which provided suitable conditions for the *E. coli* to thrive. The results show an initial regrowth and multiplication of *E. coli* in the plots of the first and second cycle cultivations. The concentration after the second cycle was significantly higher at $p<0.05$ than in the first cycle. *E. coli* was not found after the third and fourth cultivation cycles. These may have been due to changes in the soil environment or competition for resources that may have caused a decline in colonies following the initial re-growth and multiplication.

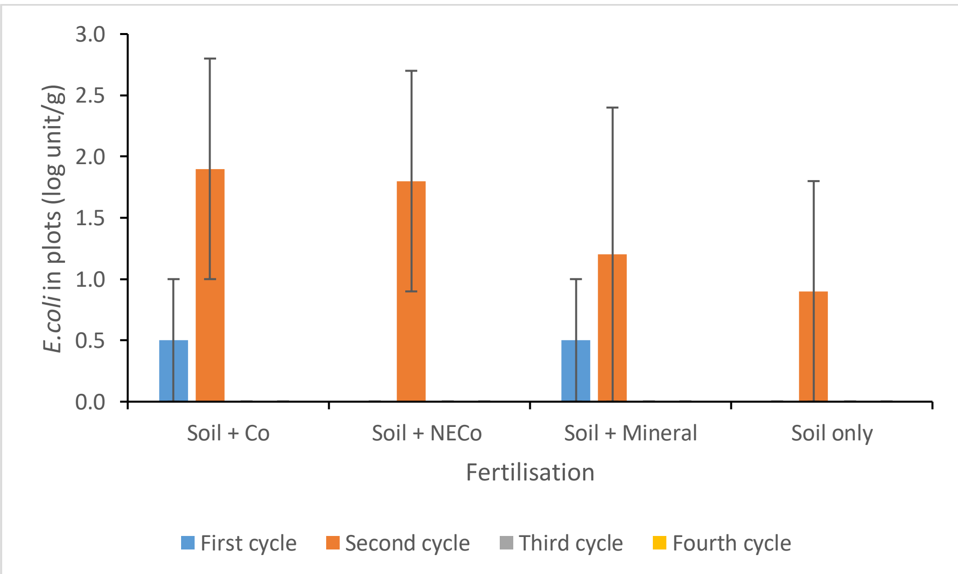

Figure 6-5. *E. coli* concentration in log units/g in experimental plots after four cycles of cultivation.

Total coliform concentrations were not significant at $p<0.05$ among the treatment plots but differences were significant between the cultivation cycles. The coliform levels were highest in the plots after the second cycle of lettuce cultivation (Figure 6-6) corresponding to highest levels recorded for *E. coli* in Figure 6-5. The total coliform levels were generally lowest after the third cycle of lettuce cultivation with the lowest figure of 2.5 log unit/g recorded in the soil + mineral plots. Contrary to the high levels of total coliform recorded on lettuce in the fourth cycle (Figure 6-5), the total coliform levels in the plots after fourth cycle were generally lower suggesting that lettuce contamination with coliform may not necessarily arise from the contaminated soils.

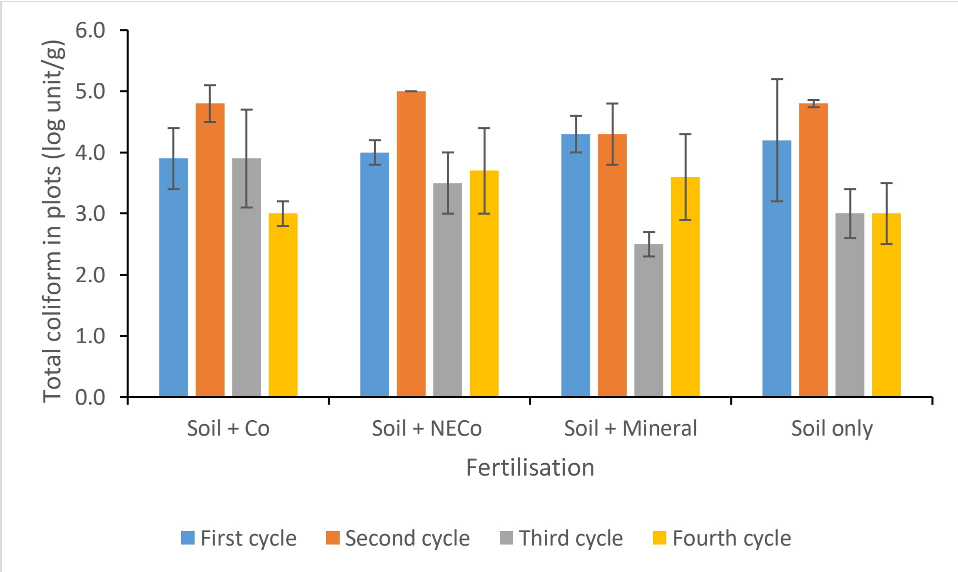

Figure 6-6. Total coliform concentration in log units/g in experimental plots after four cultivation cycles

6.4 Conclusion

Findings from this study show that FDF is a competitive alternative to mineral fertilisers. FDF Co-compost plots gave 344.4% increase in lettuce yield in the first residual effect (second cycle) compared to the control (Soil only) while the mineral fertilised plots recorded 29.3% increase in yield for the first residual effect. There was gradual decline of lettuce yield over the successive cycles. The soil bulk density, water holding capacity and ECEC of the treatment plots significantly differed after cultivations. The bulk density was relatively lower while the water holding capacity was higher in FDF fertilised plots compared to the mineral and soil only fertilised plots for the cultivation cycles. The bulk density was lowest in all plots after the second cycle whiles the water holding capacity and ECEC was highest in all plots after the same second cycle. Indicating that the optimal soil conditions occurred after the second cycle. This supported soil porosity and water availability for the roots to absorb nutrients readily available in the soil resulting in higher lettuce yields observed for FDF fertilised plots. This implies that there is the need for repetitive application of FDF (perhaps after the second cycle) to ensure yield decline does not occur to cause economic loss to smallholder farmers. These fertiliser cost savings to smallholder farmers could improve the livelihood of the household. A clear win from both environmental sustainability, poverty alleviation and circular economy.

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Chapter 7 Economic Assessment of Faecal Derived Fertilisers on lettuce Yield Following Successive Cultivations

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Abstract
This study evaluates the direct and residual effects of faecal derived fertiliser (FDF) at recommended rate on yield and profitability of lettuce for four planting cycles. Data was obtained from a field experiment that included four fertiliser treatments (FDF co-compost, N-enriched FDF co-compost, mineral, and no fertiliser) i.e., three treatments compared to a control and analysed using and cost-benefit analysis. The results show that lettuce yields were generally higher when co-compost (14.28Mt/ha) was used against N-enriched (6.89Mt/ha), mineral (5.11Mt/ha), and no fertiliser (2.95Mt/ha). Profits earned from plots cultivated with co-compost were the highest for all the cultivation cycles. Furthermore, the net present value (NPV), benefit-cost ratio (CBR), and return on investment (ROI) were highest for co-compost and N-enriched plots and increased during the residual fertilisation of the FDFs. The NPV for no fertiliser (control) plot and mineral fertiliser plot is lower compared to N-enriched and co-compost fertilisers. The study recommends that farmers should be encouraged to use FDF to increase yield and farm income.

Keywords: *Economic assessment, faecal derived fertiliser, Residual effect, Cost-Benefit Analysis, Lettuce, Ghana.*

7.1 Introduction

Globally, agriculture is faced with the challenge of meeting growing food demand while concurrently replenishing degraded soils to support biodiversity and ecosystem sustainability (Foley et al., 2011; Kirchenmann, 2010). The demand for food is expected to increase by 70% when the world's population rises from 7.4 billion to 9.1 billion by 2050 (Béné et al., 2015; FAO, 2015). To meet the rising food demand, there is a need to double crop productivity and production (Hunter et al., 2017), through the widespread use of mineral fertilisers and pesticides. However, this approach will increase agriculture's impact on water quality, climate change as well as soil quality overtime (Foley et al., 2011; West et al., 2014). Soil fertility is a fundamental determinant of agricultural productivity and one of the primary barriers to increasing food security in developing countries (Bado & Bationo, 2018;

Drechsel et al., 2004). According to Nartey et al., (2021), soil fertility depletion and low soil organic matter are causes of the declining rate of crop production in Africa. An approach to improving soil fertility management is the use of organic fertiliser sources including faecal sludge (FS) or manure (Nartey et al., 2021). Recent studies have shown that the application of mineral fertiliser alone leads to increase in yield in the short term but may lead to soil acidification and declining yields in the long term(Bado & Bationo, 2018). However, the combination of organic and mineral fertilisers is associated with increased yields and water use efficiency(Bado & Bationo, 2018; Chivenge et al., 2011; Ripoche et al., 2012).

Though agriculture has become more dependent on mineral fertilisers due to loss of soil fertility and agricultural land, stakeholders in the agricultural sector such as Ministry of Food & Agriculture; Ministry of Environment, Science, Technology and Innovation; NGOs, and other Development Partners; compost businesses, among others are developing alternative, inexpensive fertilisers that are more productive, eco-friendly, and sustainable (Kuwornu et al., 2017; Mariwah & Drangert, 2011). Organic fertilisers sourced mainly from plant, animal, and human waste have become a viable alternative due to their significant agricultural, health, and environmental benefits. Research and development activities of the International Water Management Institute (IWMI) have led to the development of a faecal derived fertiliser (FDF) – N enriched formulations which address challenges of using 'regular' compost such as bulkiness, low nutrient content, and associated health risks (Impraim et al., 2014; Nikiema et al., 2013). The FDF is safe, efficient, and cost-effective fertiliser produced from faecal sludge and other waste streams such as domestic and agricultural waste that is blended with additional input materials to enhance nutrient value (Nartey et al., 2017; Nikiema et al., 2013). FDF application to crops has the potential to address issues such as soil nutrient depletion, affordability of organic fertilisers and improving soil quality leading to increased productivity and food security (Adamtey, 2010; Ofosu Ansong, 2014; Pradhan et al., 2016). Further, it is useful in conserving ecosystem services such as improving soil structure, water infiltration, soil aeration, increasing water-holding capacity, reducing pests and diseases, and neutralizing soil toxins and heavy metals (Cofie et al., 2010; Nikiema et al., 2014).

FDF usage for crop production has been the focus of research for several decades. Previous studies in Ghana examined the performance of FDF on food crops including, cereals (Adamtey, 2010; O. O. Cofie et al., 2005; Pradhan et al., 2016), tomato, rice, maize, and pepper under different ecological zones (Nartey et al., 2021). Other studies include the usage of FDF to cultivate lettuce in urban agriculture in Ghana (Pradhan et al., 2019), beans and sweet potatoes in India (Girija et al., 2019), and cabbage in Bangledesh (Islam & Hasan, 2017). Findings from these studies have established that FDF products give higher yields as compared to mineral fertilisers in crop production. For example, Nartey et al. (2021) reported 12% higher yield for tomato, 27% for rice and maize and 30% for pepper for plots cultivated with FDF. In addition, tomato, and rice farmers from two irrigation schemes in Ghana also reported that FDF was easy to apply, cost-saving, and reduced pest and disease incidences on farms).

The application of FDF has been found to be of economic importance to farmers. For example studies in Ghana focusing on the economic/financial feasibility (Aboah et al., 2014), farmers' willingness-to-pay (WTP) for FDF in Ghana (Kusi, 2019; Kuwornu et al., 2017);, farmers perception and economic benefits of excreta use (Cofie et al., 2010), perception of FDF usage in Ghana (Appiah-Effah et al., 2015), assessing the factors influencing farmers WTP for FDF (Otoo et al., 2018) have yielded positive results. The findings from these studies revealed that FDF application increases farmer income.

Adoption of FDF in agriculture is influenced by socio-economic factors such as household head, the unit cost of fertiliser, household size, farm size, age of farmer, and household income (Appiah-Effah et al., 2015; Kusi, 2019; Kuwornu et al., 2017). Other studies also identify product attributes such as suitable credit offer, price, health risk, nutrient content, and the convenience of location as factors influencing FDF purchasing decisions in Ghana (Cofie et al., 2010; Otoo et al., 2018). Using the probit model, Otoo et al., (2018) found that farm income and off-farm experience positively determine WTP for pelletized FDF while farming experience and use of organic fertiliser reduce the probability of adopting non-pelletized FDF.

Despite the extensive research into FDF application in Ghana's crop production sector over the last two decades, there is a dearth of knowledge as to how the residual effect of the application of FDF on crops and soils affects performance of the farm following repeated crop cultivation cycles. This study assesses the direct and residual effects of FDF application using indicators such as yield, and cost-benefit ratios. Specifically, the study assesses the costs and benefits associated with investment in FDF and other types of fertilisers in repeated crop cultivation i.e., in four production cycles with one-off application of FDF and other fertilizers.

7.2 Materials and Methods

7.2.1 Study Area

The study was carried out at field scale at the Jekora Ventures Limited (JVL) – Yilo Krobo Municipal Assembly (YKMA) Recycling Plant located at Akorley, Somanya in the YKMA of Ghana (Figure 7-1). The annual rainfall of the area ranges from 750 to 1,600 mm and it spans from May to October (bimodal). Average temperatures range between 24 and 30°C while relative humidity ranges between 60 and 90% (Sadiq, 2016). The major soil type is Savanna Ochrosol (Eastern Regional Co-ord Council, 2016). It has low nutrient reserves, with the topsoil consisting of dark greyish brown humus sandy or clay loams (Eastern Regional Co-ord Council, 2016).

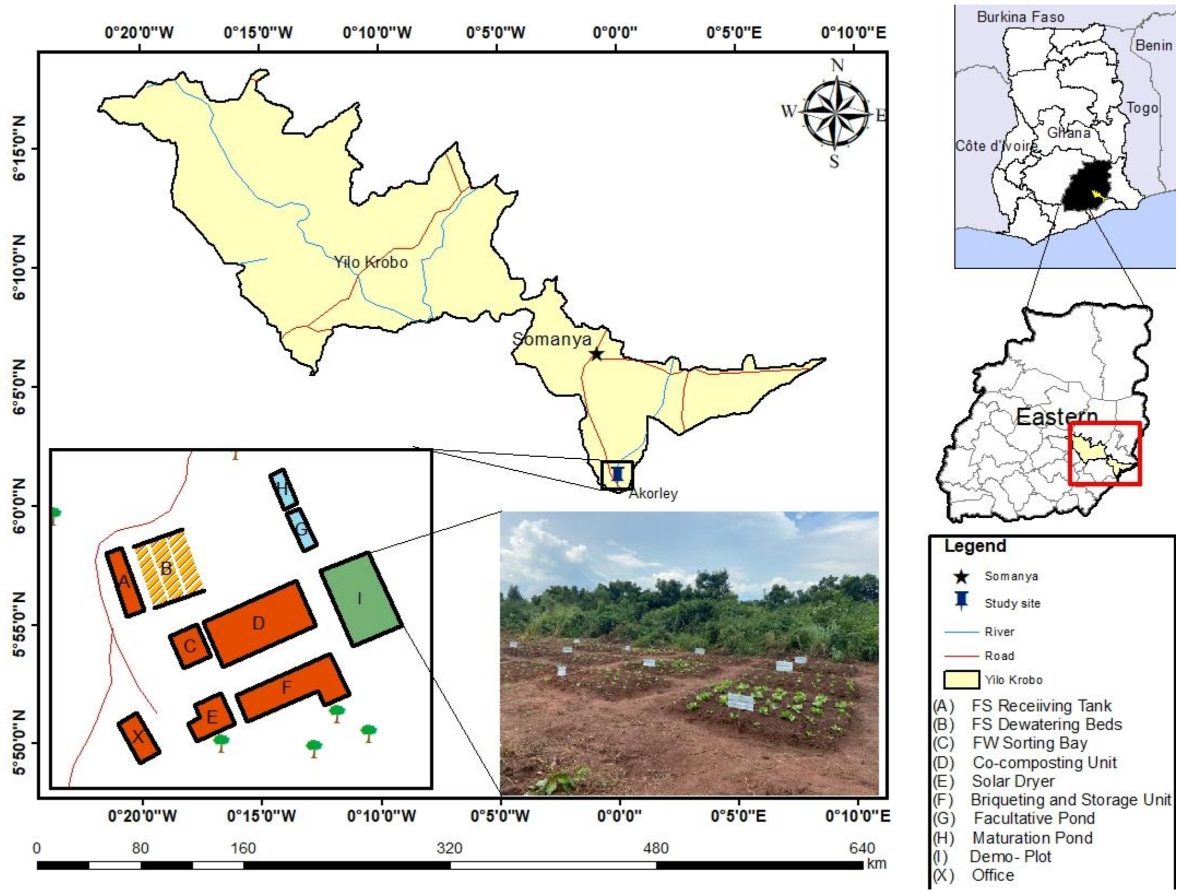


Figure 7-1. Map of study site

7.2.2 Data

The unit of analysis is the smallholder farmer based on a per hectare (ha) land production. The price for the mineral fertiliser inputs is a product of current national policy (Planting for Food and Jobs) on subsidising fertiliser cost for smallholder farmers rather than inherent cost.

The data used for the analysis was obtained from a lettuce field experiment using a randomized complete block design (RCBD) (Nartey et al., upcoming). Lettuce seeds *(Lactuca sativa L. var. Great lakes)* were obtained from the local agro-chemical shop and sown in a nursery. Healthy and vigorous lettuce seedlings at true 4-leaf stage were transplanted after four weeks onto the experimental plots on raised beds at a spacing of 30cm × 30cm (Pradhan et al., 2019; Torgbo et al., 2018). The planting population was 48 plants/m^2 which amounted to 111,111 plants/ha. The treatments, FDFs that is co-compost produced from food waste and dewatered FS and ammonium sulphate N-enriched FDF were obtained from the JVL - YKMA Recycling Plant in Somanya. The FDF co-compost and N-enriched FDF co-compost were produced according to the methods and procedure described by Nartey et al., (2022) and Adamtey et al., (2009). The mineral fertilisers, ammonium sulphate and $NPK_{23\text{-}10\text{-}5}$ were purchased from the local agro-chemical shops.

The total number of treatments were four (4); FDF co-compost, FDF N-enriched co-compost, mineral (NPK and Sulphate of Ammonia), and no fertiliser (control) with three replicates making a total of 12 experimental plots. Ghana's Ministry of Food and Agriculture (MoFA) recommended fertiliser rate of 150 KgN/ha for lettuce were applied only one time at the start of experiment to the plots with an area of 4.32 m^2 (2.4m x1.8 m) and at the start of the experiment. Data was collected on the yield of lettuce (kg), cost of fertiliser (GHS), market price of lettuce (GHS), labor (man-days), cost of farm activities (ploughing, watering, fertiliser application, sowing, weeding, etc). This information was used to analyse the profitability of the treatments. For the purposes of this analysis, the following assumptions in Box 7-1 were made with regards to cost inputs, and market prices of produce among others.

Box 7-1. Input assumptions

Assumptions

i. Time spent in nursery bed preparation is 2 hours; sowing of seed lasted for an hour. Further, watering of nursery plants lasted 15 minutes per day and it continued for 32 days. Therefore, there is a total of 8 hours of watering which is equal to 1 man-day. Irrigation of transplanted seedlings lasted for 1.5 hours each day for 35 days, 52.5 hours of irrigation which equals 6.6 man-days.

ii. Input cost; Seed – USD 13.04/200 grams; Co-compost[1] – USD 5.7/50 kg; NPK[2] – USD 7.82/25 kg; Sulphate of Ammonia (SA)[2] – USD 18.75/50 kg; N-enriched[3] – USD 6.52/50kg (i.e., 4.74 kg of SA + 50 kg of Co-compost); Cost of water (nursery) – USD 32.6/680 lit (10 liters applied each day for 32 days); Cost of water (irrigation) – USD 32.6/681 lit (10 liters applied each day for 35 days).

iii. The per unit price of lettuce is USD 0.90/kg.

iv. A discount rate of 23% is assumed. This is the average cost of capital (i.e., interest on capital borrowed) prevailing in Ghana.

NB:

[1] Retail price of a 50Kg bag of faecal derived fertiliser co-compost at the treatment plant.

[2] Government subsidised prices of NPK and Sulphate of Ammonia at the retail shop.

[3] The cost of N-enriched co-compost made of retail price of 50Kg bag and the cost of sulphate of Ammonia to enrich to 3% total N.

7.2.3 Cost-Benefit Analysis (CBA)

The performance of FDF application was evaluated using a CBA methodology which is commonly used to assess the profitability of alternative investments in both the public and private sectors (Brent, 1996). The relative profitability of alternative fertiliser regimes, comparing their different flows of cost and benefits over the four cultivation cycles were assessed. The costs and benefits of different fertiliser treatments (co-compost, N-enriched, mineral (NPK and S/A), and no fertiliser) were performed using the Benefit-Cost Ratio (BCR) and Net Present Value (NPV) indicators. The steps included (i) specifying the underlying assumptions, (ii) generating a cash flow projection for each scenario, (iii) applying discounted measures of investment worthiness under each scenario and (iv) conducting a sensitivity analysis for each scenario, to check whether the investments would remain viable if revenues were reduced and costs increased.

Farm Income is estimated for each fertiliser treatment using budgetary analysis. All cost of production was deducted from the total output value. The total cost of production includes the cost of labor, seeds, fertiliser, herbicides, and pesticides, rent on land, and depreciation on farm assets. The straight-line method was used to estimate the depreciation of farm assets to obtain a fixed cost of production.

7.3 Results and Discussion

7.3.1 Effect of different fertiliser types on yield performance

The yield of lettuce under co-compost was higher than the other fertiliser types. The average yields were 4.86 Mt/ha (no fertiliser), 11.11 Mt/ha (Mineral), 7.95 Mt/ha (N enriched) and 27.93 Mt/ha (co-compost) for production cycle 1 (Table 7-1). Co-compost fertiliser plots had the lowest yield variability for the first two production cycles ranging from 2% to 9% (Table 1), compared to other fertiliser types. The yield of lettuce fluctuates between 9% and 21% for the no fertiliser plots, 12% and 34% for mineral fertiliser, and 5% and 33% for N-enriched fertiliser for the first two production cycles (Table 7-1). In the third production cycle, mineral fertiliser had the highest yield variability (73%), followed by no fertiliser (68%), co-compost fertiliser (64%), and N-enriched fertiliser (59%). However, variability in yield decreased to about 5% and 8% for enriched and co-compost respectively.

Table 7-1. Yield of Lettuce (Mt/ha) in after four cultivation cycles.

TREATMENTS	**Cycle 1**	**Cycle 2**		**Cycle 3**		**Cycle 4**	
	Yield (Mt/ha)	Yield Mt/Ha)	%Δ in Yield	Yield (Mt/ha)	%Δ in Yield	Yield (Mt/Ha)	%Δ in Yield
No fertiliser	4.86 (0.21)	4.15 (0.09)	-14.62	1.08 (0.68)	-73.94	1.71 (0.41)	59.06
Mineral	11.11 (0.12)	5.36 (0.34)	-51.71	2.71 (0.73)	-49.32	1.24 (0.16)	-54.38
N-enriched	7.94 (0.33)	10.93 (0.05)	37.64	5.73 (0.58)	-47.61	2.92 (0.05)	-48.88
Co-compost	27.93 (0.09)	18.44 (0.02)	-33.97	7.40 (0.64)	-59.82	3.35 (0.08)	-54.78

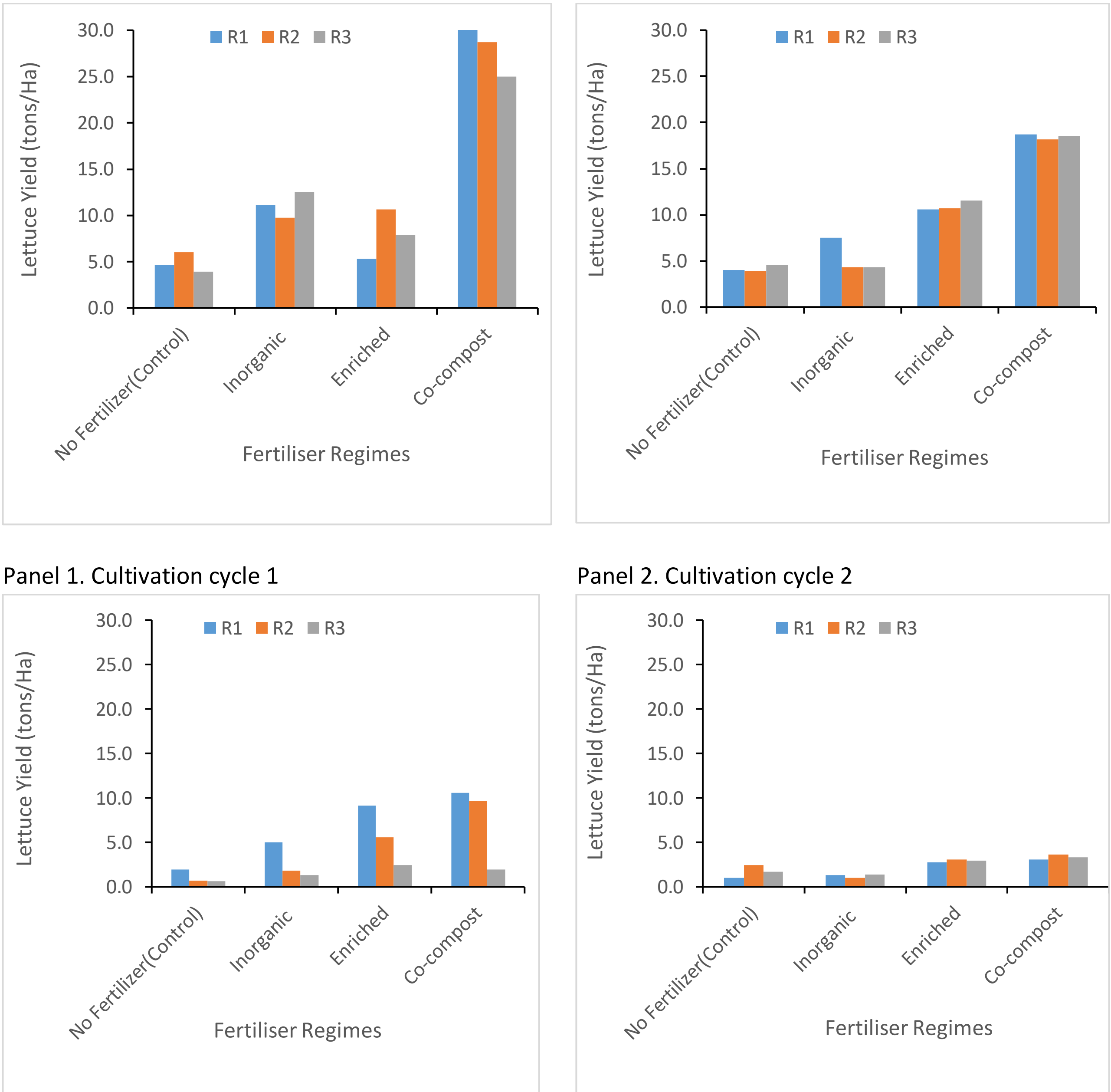
Figures in parenthesis are coefficient of variation (CV) which measures variability in yield within the treatment. Source: Field Data, 2021

Results for cultivation cycle 2 show that the average yield of lettuce under co-compost is relatively higher compared to other treatments. Average yield ranges between 4.15 Mt/ha (No fertiliser), 5.36 Mt/ha (mineral), 10.93 Mt/ha (N-enriched) and 18.44 Mt/ha (co-compost). Further, it decreases across all treatments except for the N-enriched which had about 37.64% increase in lettuce yield. Though yield variability is comparatively low among other treatment regimes, 9% (No fertiliser), 5% (N-enriched), 2% for co-compost, the results revealed very high yield variability for plots applied with mineral fertiliser (34%). This high yield variability implies higher uncertainty in yield levels and by extension variable profits with mineral fertiliser. Also, yield decrease is highest for mineral fertiliser compared to others (Figure 7-2). In addition to high yield variability in cycle 2, plots planted with mineral observed yield decreased by about 51.7% compared to the control field (14.6%) and co-compost (33.9%)[1]. However, yield increased under N-enriched from 7.94 Mt/ha to 10.94 Mt/ha (37.34%). The observed decrease in lettuce yield under mineral fertiliser may be attributable to the loss of nutrients due to leaching (Bationo et al., 2018).

Results from the study generally showed better performance of lettuce under co-compost for all the cultivation cycles. These results agreed with Kutu et al., (2011), who reported an increase in the yield of spinach after the application of FDF compost. Similarly, Cofie and Adamtey (2009), also established a two to threefold increase in yields of crops (maize, sorghum, and cabbage) grown on FDF treated

[1] $\left(\frac{Y_{C1}-Y_{C2}}{Y_{C1}}\right) * 100\%$

soils as compared to untreated soils. The relatively higher performance of the co-compost and N-enriched plots after the one-time application is attributable to the gradual release of nutrients (Gutser et al., 2005). Several authors find that the residual effect of compost increases yield after the first application. For instance, Eghball and Power (1999) also observed that about 60% of manure N and 80% of compost N became usable to plants in the subsequent years and improved crop yield. Similarly, Ginting et al., (2003)observed that residual effects of compost on soil properties improve the soil quality for several growing seasons without additional application and support crop growth and yield. Similar results were obtained by Djawu (2017), in cowpea production in coastal Ghana.



Panel 1. Cultivation cycle 1

Panel 2. Cultivation cycle 2

Panel 3. Production cycle 3

Panel 4. Production cycle 4

Figure 7-2. Yield (kg) of lettuce under different fertiliser regimes. NB: R1, R2, R3 represent replications under each treatment for cycles 1, 2, 3, and 4.

7.3.2 Economic Analysis of Lettuce under Different Fertiliser Regimes

7.3.2.1 Profitability of Lettuce

Table 7-2 shows the profitability of different fertiliser regimes. The profit from lettuce production increased from USD3,233.00/ha for No fertiliser to USD8,342.17/ha for mineral, USD 5,186.34/ha for N enriched, and USD 22,272.24/ha for co-compost (Cycle 1) (Table 7-2). The results also revealed that fields applied with co-compost make 6 times more profit per hectare than plots with no fertiliser. Similarly, profits per hectare from co-compost (USD 22,272.24) are about three times higher than mineral fertilizer (USD8,342.17), four times higher than the N-enriched compost (USD5,186.34). The observed highest profit from the plot of co-compost is partly due to the high yield.

Profit levels decreased for all fertiliser treatments except the N enriched plots which increased (from USD5,186.34/ha to USD8,670.75/ha) during the second cycle (Table 7-2). This is partly due to the release of nutrients from the organic and inorganic components of the N-enriched, resulting in higher yield (from 7.9Mt to 10.94Mt) as well as the relatively low total cost in cycle two compared to cycle one. Specifically, profit levels were USD2,559.75/ha, USD3,657.75/ha, USD8,670.75/ha, and USD15,420.75/ha for plots with no fertiliser application, mineral, N-enriched, and co-compost respectively. Furthermore, yields from plots applied with co-compost from the first cultivation cycle observed a higher profit level in the subsequent production season. Profit levels from plots applied with co-compost were 6, 4, and 2 times higher than plots with no fertiliser, mineral, and N-enriched compost, respectively. Though change in yield decreased during the second cycle (Cycle 2) for co-compost, this treatment observed higher yield relative to the other treatments, which led to higher gross margin and higher profits compared to the other fertiliser treatments. Except for N-enriched, profits earned from lettuce decreased by 20.84%, 56.15%, and 30.37% under no fertiliser, mineral, and co-compost fertiliser, respectively (Table 7-2).

Table 7-2. Gross Margin and Profits of Different Fertiliser Regimes for Four cultivation Cycles.

Parameters	No Fertiliser	Mineral	N-enriched	Co-compost
			Cycle 1	
Gross Return (USD/ha)	4,410.00	9,990.00	7,110.00	25,110.00
Total Variable Cost (USD/ha)	1,168.37	1,639.93	1,915.76	2,829.86
TOTAL COST (USD)	1,176.26	1,647.82	1,923.65	2,837.75
Gross Margin (USD/ha)	3,233.73	8,342.17	5,186.34	22,272.24
			Cycle 2	
Gross Return (USD/ha)	3,735	4,833.00	9,846.00	16,596.00
Total Variable Cost (USD/ha)	1,168.4	1,168.4	1,168.4	1,168.4
TOTAL COST (USD)	1,175.2	1,175.20	1,175.20	1,175.20
Gross Margin (USD/ha)	2,559.75	3,657.75	8,670.75	15,420.75
			Cycle 3	
Gross Return (USD/ha)	973.23	2,447.71	5,157.4	6,668.6
Total Variable Cost (USD/ha)	1,168.4	1,168.4	1,168.4	1,168.4
TOTAL COST (USD)	1,175.2	1,175.2	1,175.2	1,175.2
Gross Margin (USD/ha)	-202.01	1,272.47	3,982.16	5,493.36
			Cycle 4	
Gross Return (USD/ha)	153.9	111.6	2,642.86	3,025.05

Total Variable Cost (USD/ha)	1,168.4	1,168.4	1,168.4	1,168.4
TOTAL COST (USD)	1,175.2	1,175.2	1,175.2	1,175.2
Gross Margin (USD/ha)	-1,021.3	-1,063.6	1,467.66	1,849.85

Source: Field Data, 2021

Table 7-3. Change in profit under different treatments for four cultivation cycles.

Treatments	**Cycle 1**	**Cycle 2**		**Cycle 3**		**Cycle 4**	
	Mean Profit (USD)	**Mean Profit (USD)**	**%Δ in Profit**	**Mean Profit (USD)**	**%Δ in Profit**	**Mean Profit (USD)**	**%Δ in Profit**
No Fertilizer	3,233.73	2,559.75	-20.84	-202.01	-107.89	-1021.3	-405.57
Mineral	8,342.17	3,657.75	-56.15	1272.47	-65.21	-1063.6	-183.59
N-Enriched	5,186.34	8,670.75	67.18	3982.16	-54.07	1461.8	-63.29
Co-compost	22,272.24	15,420.75	-30.76	5493.36	-64.38	1839.8	-66.51

Profit levels under N-enriched and co-compost were higher than mineral and no fertiliser treatments. The higher profits obtained from the co-compost field in the third cycle of production compared to other fields is due to higher yield. Additionally, profits decreased further in the third production cycle by 107.89%, 65.22%, 54.07%, and 64.38% for no fertiliser, mineral fertiliser, N-enriched and co-compost fertiliser, respectively (Table 7-3**).** However, in the fourth cultivation cycle (Cycle 4), nearly 90% of the lettuce yield from no fertiliser and mineral fertiliser plots were not marketable (leaves were very small). Thus, those plots recorded significant losses (Table 7-2). Plots with N-enriched and co-compost still made profits of about USD1,467.66 and USD 1,849.85, respectively for cycle 4. The results from the field study revealed that it was not profitable to cultivate lettuce on the field with no fertiliser beyond the second cultivation cycle (cycle 2). In addition, lettuce cultivation on plots with mineral fertiliser was also not profitable beyond the third cycle (cycle 3). However, plots with N-enriched and co-compost fertilisers observed profit levels for all the four cultivation cycles, though profits decreased significantly from successive cycles. Indicating that, the residual effect of the FDF could be profitable to smallholder farmers beyond four successive cultivation cycles after one-off application.

7.3.2.2 Cost-Benefit Analysis

Using a discount rate of 23%, a positive NPV was found for all fertilizer treatments of lettuce cultivation. This m*eans that* the discounted present value of all future cash flows associated with lettuce cultivation is attractive. However, as compared to no fertiliser and mineral fertiliser applications, N-enriched, and co-compost applications provide a higher NPV of lettuce cultivation. When co-compost is used on the farm, the NPV of lettuce output is greatest (USD 15,322.4), followed by N-enriched application (USD 5,633.0), and inorganic fertiliser application (USD 4,878.0). When no fertiliser is applied to the farm, however, lettuce production generates a low NPV (USD 2,355.1). Furthermore, the NPV of N-enriched and co-compost increases after the first cycle of cultivation (cycle 1) when no extra fertiliser is used. However, in the second cycle of cultivation, when no fertiliser is applied, the NPV for the no fertiliser plot and mineral fertiliser plots decrease. This suggests that N-enriched and co-compost plots retain enough nutrients and moisture to sustain lettuce development in the following cultivation, allowing producers to reap greater benefits, even in the medium to long term.

Lettuce cultivated using co-compost and N-enriched had a BCR of 14.5 and 8.4 and ROI of 385.7 and 281.1, respectively while lettuce cultivated using no fertiliser and mineral fertiliser had a BCR of 3.2 and 4.1 and ROI of 109.5% and 130.2%, respectively. An ROI of 385.7 indicates that for every dollar that is invested in the cultivation of lettuce using a co-compost, about USD 385.7 is generated. This implies that investment in co-compost for lettuce production is profitable and higher than investment in N-enriched which has an ROI of 281.1. On average, lettuce cultivated with co-compost and N-enriched generates a BCR of 7.8 and 4.7 and ROI of 309.2% and 175.7%, respectively, while cultivating lettuce using no fertiliser and mineral fertiliser generates a BCR of 2.3 and 3.3 and ROI of 67.8% and 116.9%, respectively. However, BCR and ROI for plots with no fertiliser and mineral fertiliser was 0.8 and -8.6% and 0.9 and -5.0% in the third and fourth cultivation cycles respectively. This implied that lettuce production was not profitable in these fields. This is attributable to a significant reduction in yields.

These findings established that lettuce farms produce more benefits when co-compost and N-enriched fertilisers are applied to the soil than when no fertiliser and mineral fertiliser are applied. The benefits of using co-compost and N-enriched are enormous since they produce higher yields, compared to using mineral fertiliser. Compost-based fertilisers such as co-compost and N-enriched do not only increase soil nitrogen content but also enhance the storage of soil organic carbon and influence the pH and soil bulk density.

Table 7-4. Cost-Benefit Analysis of Fertiliser Regimes

	No fertiliser					Mineral				
Cash Flow (USD)	**Cycle 1**	**Cycle 2**	**Cycle 3**	**Cycle 4**	**Pooled**	**Cycle 1**	**Cycle 2**	**Cycle 3**	**Cycle 4**	**Pooled**
Cash Inflow	4,410	3,735	973.2	153.9	2318.0	9990	4833	2447.7	111.6	4345.6
Cash Outflow	1,176.3	1,175.2	1,175.2	1175.2	1175.5	1647.8	1175.2	1175.2	1175.2	1293.4
Net Cash Flow	3233.7	2559.8	-202	-1021.3	1142.6	8342.2	3657.8	1272.5	-1063.6	3052.2
Discount Factor @ 23%	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9
Discounted Cash Inflow	4187.7	3367.7	877.5	138.5	2145.6	9486.5	4357.8	2207	100.6	4038.0
Discounted Cash Outflow	1117	1059.7	1059.7	1057.7	1088.1	1,564.8	1059.7	1059.7	1059.7	1185.9
Discounted Net Cash Flow	3070.8	2308.1	-182.1	(919.2)	1057.6	7,921.7	3298.1	1147.3	(959.1)	2852.1
Net Present Value @23%	2629.1	2081.1	-164.2	(830.4)	928.9	6,782.3	2973.8	1034.5	(864.7)	2481.5
Cost-Benefit Ratio	3.7	3.2	0.8	0.1	2.0	6.1	4.1	2.1	0.1	3.1
ROI	138.4	109.5	-8.6	-86.9	38.1	297.1	130.2	45.3	-37.9	108.7
	N-Enriched					**Co-compost**				
Cash Flow (USD)	**Cycle 1**	**Cycle 2**	**Cycle 3**	**Cycle 4**	**Pooled**	**Cycle 1**	**Cycle 2**	**Cycle 3**	**Cycle 4**	**Pooled**
Cash Inflow	7110	9846	5157.4	2637.0	6187.6	25110	16596	6668.6	3015.0	12847.4
Cash Outflow	1923.7	1175.2	1175.2	1175.2	1362.3	2837.8	1175.2	1175.2	1175.2	1590.9
Net Cash Flow	5186.3	8670.8	3982.2	1461.8	4825.3	22272.2	15420.8	5493.4	1839.8	11256.5
Discount Factor @ 23%	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9
Discounted Cash Inflow	6751.7	8877.8	4650.3	2377.7	5664.4	23844.5	14964.1	6012.9	2718.5	11885.0
Discounted Cash Outflow	1826.7	1059.7	1059.7	1059.7	1251.4	2694.7	1059.7	1059.7	1059.7	1468.4
Discounted Net Cash Flow	4925	7818.2	3590.6	1318.0	4413.0	21149.7	13904.4	4953.2	1658.9	10416.5
Net Present Value @23%	4216.5	7049.4	3237.5	1188.4	3,923.0	18107.5	12537.2	4466.1	1495.7	9,151.6
Cost-Benefit Ratio	3.7	8.4	4.4	2.2	4.7	8.8	14.1	5.7	2.6	7.8
ROI	168.2	281.1	129.1	124.4	175.7	557.1	385.7	137.4	156.5	309.2

Source: Field Data, 2021

7.4 Conclusions and Policy Implications

Significant yield differences were observed between all the treatment types. The yield on plots with FDF co-compost were highest compared to the rest of fertiliser treatments. Yield ranged between 4.8 Mt/ha for plots with no fertiliser to 27.91 Mt/ha for plots with FDF co-compost. There was a significant reduction in yield of 51.71% on the plot with mineral fertiliser after the initial fertiliser application. In all, yield variability was very low among plots that were applied with FDF co-compost, ranging between 1.5 % to 9.0 % for the four production cycles. The highest variability in yield (34%) was observed on plots with mineral fertiliser after no additional fertilisation in successive cultivations. Following the unit of economic analysis being at the smallholder farmer level, profit was highest under plots cultivated with co-compost. Co-compost made about 6 times more profit per hectare than plots with no fertilizer, about 2 times more than mineral fertiliser, and about four times more than the N-enriched compost. The high profit from the co-compost was due to the high yield obtained from these plots and the low total cost of all treatment options. In addition, whiles profits were highest in co-compost treated fields, they only increased in the N-enriched plots in cycle 2 and decrease in all other cycles. Furthermore, the study found that the NPV, CBR, and ROI for co-compost and N-enriched increases after cycle 1 (i.e., cycle 2 to 4). However, in cycle 2 of cultivation, the NPV for the no fertilizer (control) plot and mineral fertiliser plot decreased significantly.

The study revealed that FDF provides relatively higher crop yield and residual yield effect during the entire lettuce cultivation compared to mineral fertilisers. However, similar economic analysis would have to be conducted on other segments of the sanitation value chain for example the treatment and production of FDF before drawing conclusions about the financial viability of the approach overall. This requires sensitisation and education of smallholder farmers about the short and long-term benefits of these FDF and their impact on the soil ecosystem. It is also recommended further studies be carried out using farmer plots involving different crops such as cereals, legumes, and other vegetables. An assessment of farmers willingness to use and pay for FDF following this new information on residual effects needs to be carried out to know the acceptance of these fertilisers. Farmers' acceptance and willingness to use co-compost and N-enriched may also depend on the market potential of food products produced using the recommended fertilisers. Therefore, a consumer market survey involving a choice experiment is imperative for the sustainable recommendation and adoption of FDF co-compost and N-enriched fertilisers by smallholder farmers.

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Chapter 8 Final Discussions

8.1 Introduction

This chapter presents an implication of the study and overall conclusions and recommendations. The overall aim of this research was to generate new knowledge and understanding on the recovery of nutrients and *E. coli* inactivation during treatment and use of fertiliser produced from faecal sludge (FS) and solid waste. This aim was realised through the execution of specific experiments and their findings discussed below.

8.2 Overall conclusions

This study set out to investigate the recovery efficiency of nutrients and *E. coli* inactivation efficiency in decentralised FS and food waste (FW) treatment plants hence it was hypothesised that through optimisation of key variables, the co-treatment of FS and FW has the potential to recover nutrients and organic matter at International Standard safety levels while remaining economically viable for smallholder farmers. As such faecal derived fertilisers (FDFs) could be considered an alternative to mineral fertilisers and a potential solution in the advancement of circular economy and food security. The hypothesis was tested through the various objectives that were achieved and their conclusions discussed in the following sub-sections:

8.2.1 Recovery of nutrients and *E. coli* inactivation in decentralised faecal sludge and food waste treatment systems (Objectives 1 and 2)

These studies successfully characterised the total N, P and K recoveries and *E. coli* inactivation dynamics in a decentralised FS and FW treatment plant. It was found that in an end-to-end assessment of the plant, significant losses of nutrients (total N, P and K) occurred at the dewatering stages and co-composting stages of treatment. For example, between 50-70% of total N from FS was lost at the dewatering stage of treatment while more than 50% of the total N was lost during co-composting. The estimated total N recovery efficiency was 20-47%. These findings show that while these decentralised co-treatment plants aim to recover nutrients, the empirical recovery efficiency is actually low. This low efficiency was due to nutrient losses in the system through various process and pathways which could be optimised to minimize losses. This finding therefore supports elements of the hypothesis that through optimisation of key variables, the co-treatment of FS and FW has the potential to recover nutrients and organic matter.

Very few end-to-end studies have been carried out on decentralised FS treatment systems in literature (Levira et al., 2023; Mrimi et al., 2020). However, these few studies only focused on treatment of FS and the performance efficiency of those treatment systems against national discharge guidelines and not on nutrient recovery potentials. In terms of pathogens, the end-to-end assessment of the decentralised treatment plant found an inactivation efficiency of 0-14% for *E. coli* removal in the percolate after dewatering. The retention period in the maturation pond resulted in 99-100% inactivation efficiency of *E. coli*. As such, the *E. coli* levels observed in most of the cycles were below the maximum limits of 10 CFU/ 100ml set by the Ghana EPA and EU (Ghana EPA, 2000; Truchado et al., 2021). The final FDF (co-compost) characteristics described after treatment also revealed *E. coli* levels to be below detection limit thus meeting the quality guidelines set by the USEPA (USEPA, 1993) and EU (Leifert, 2017). These findings also support the elements of the hypothesis that with optimisation of the treatment process, the recovery of nutrients and organic matter can result in safe FDF meeting local and international standards.

8.2.2 Production and storage of faecal derived fertilisers from decentralised treatment systems (Objective 3)

This study answered two fundamental questions: (a) is the co-composting process and product consistent? and (b) does the quality of the product alter during storage? within the broader hypothesis framing. Findings showed that the FDF product was not consistent in its characteristics. There was significant differences between batches for some nutrients (N, NH_4, NO_3, K, avail. K, Ca, Mn, and K) characteristics. A closer look at the consistency of measured parameters in replicated piles within batches showed coefficient of variations (CVs) ranging between 0-125% and 3-111% for heavy metals and nutrients, respectively. The pathogen inactivation observed in this study was consistent with reported literature on pathogen inaction during co-composting (Cofie et al., 2009; Manga et al., 2023; Nartey et al., 2017). An important conclusion from the study was that, while characteristics of FDF was not consistent between batches, the degree of consistency may be of less importance to the FDF producer, user, and regulator if complete inactivation of *E. coli* and helminths occur meeting the international standards.

During storage, Cd and Cr levels in both FDF co-compost (Co) and enriched co-compost (NEco) did not meet the ECN-QAS standard. Storage temperature and duration of storage did not affect indigenous *E. coli* levels in any of the FDFs in this study indicating that re-growth of indigenous *E. coli* that had previously been inactivated below detection limits was unlikely barring extraneous contamination. In terms of nutrient characteristics of stored FDF, The study showed that total N was not affected by the changing temperature or storage. Longer storage of NECo under lower temperatures resulted in decrease in NH_4-N concentrations. These findings further support elements of the hypothesis that FDF could be considered as an alternative to mineral fertilisers.

8.2.3 Residual effect of applied faecal derived fertilisers on crop yield and economic profitability to smallholder farmers (Objectives 4 and 6).

The objectives of this study to evaluate the direct and residual effect of one-off application of FDF on lettuce yield, quality, and soil properties and economic profitability to smallholder farmers were achieved. The results showed higher lettuce yields observed for the FDF Co-compost during the first cultivation cycle (direct effect) than for the other treatments. It was also observed that applying FDF Co-compost at the recommended rate for lettuce increased lettuce yield by 344% in the first residual effect (second cycle) compared to no fertiliser plot (Soil only). While the mineral fertilised plots recorded 29.3% increase in yield. In addition to the yields, *E. coli* was absent on lettuce after successive cultivations. Nitrogen uptake was highest in FDF co-compost during the second cultivation cycle. This confirms empirical evidence of the residual fertilisation of FDF and supports the hypothesis that FDF could be an alternative to mineral fertiliser from a food security point of view.

How profitable will this FDF use be to farmers? It was found that plots amended with FDF Co-compost made 6 times more profit over Soil only, about 3 times more over mineral fertiliser, 4 times more over FDF enriched co-compost. Profit levels decreased for all treatments during the residual effects except the FDF enriched plots. The FDF amended plots observed higher profit levels. The findings of this study provide new knowledge and information on FDF profitability to small holder farmers beyond one time cultivation to include residual effect which hitherto was not available in literature. FDF Co-compost and enriched co-compost also had a benefit cost ratio (BCR) of 14.5 and 8.4 as well as return on investment (ROI) of 385.7 and 281.1, respectively. This meant that smallholder farmers can benefit financially (profits) from residual fertilisation of FDF making it an economically viable option for smallholder farmers.

It can therefore be concluded that, based on the overall findings from this research that, there are compelling empirical evidence to support the acceptance the overall hypothesis that through optimisation of key variables, the co-treatment of FS and FW has the potential to recover nutrients and organic matter at International Standard safety levels while remaining economically viable for smallholder farmers. As such faecal derived fertilisers could be considered an alternative to mineral fertilisers and a potential solution in the advancement of circular economy and food security.

8.3 Implications of the research

Conclusions from this research implied that full recovery of nutrients is possible and achievable in decentralised treatment systems for FS and FW when avenues for nutrient losses are blocked as much as possible from the system. This research quantified significant losses of total N in the treatment and recommended the need for strategies, approaches and technologies that are low-cost, simple, and environmentally friendly to reduce N losses during dewatering and the co-composting stages. It is believed once these strategies, approaches and optimisations are put in place, it will ensure at least about 90% nutrients are recovered as much as possible from FS and FW to contribute to alternative fertiliser to offset sole reliance on mineral fertilisers. In Ghana, for example, an additional 168,600MT of nutrients per year are required to meet the growth targets for all major crops identified in the country's Agriculture Sector Investment Plan (MoFA, 2021). Meanwhile, the amount of nutrients lost in FS from the Greater Accra Region alone is estimated at 18,240, 2,200 and 4,920 MT of N, P and K, respectively, per year which once extrapolated to cover the rest of the 15 regions of the entire country can satisfy about 82% of the annual nutrient needs of crops. Promoting full nutrient recovery from this system not only promotes circularity but also ensures that safely managed sanitation is enhanced in Ghana and beyond. As decentralised plants treat the FS it also leads to reduced public health risks from pathogens. The processes assessed in the decentralised treatment system showed that the treatment interventions render the FDF product safe with less likelihood of indigenous *E. coli* regrowth during storage.

The recommendation put forth in Section 8.4 by this study is to ensure that the co-composting stage and maturation pond treatment stages should never be avoided in a FS and food waste (FW) treatment system. While processing FS using this system has many economic and environmental benefits for Ghana's struggling sectors like the sanitation, waste management and agriculture sectors, the financial sustainability and financing mechanisms remain a challenge. Generally, treatment plants co-treating FS with other organic wastes for agriculture production have not reached full financial viability yet in developing countries (Mallory et al., 2020). Systemic challenges, for example collection cost of FS and organic waste and the high level of manual operations prevent the cost effectiveness of the treatment systems. Other challenges include unrealistic product pricing mechanisms and a lack of established markets and/or regulations. In the decentralised system assessed in Ghana, the capital expenditure (Capex) for the construction of the facility was donor funded while operational expenditure (Opex) is borne by the operators of facility with breakeven estimated to be between 3-5 years based on Opex alone. Hence, collection, transportation, and treatment stages without external incentives or subventions are generally not profitable. However, at the end use/reuse stage (especially for fertiliser) results from this research show that direct and residual benefit of one-time application of the FDF is profitable for small holder farmers. It most cases, treatment as a business is often seen as a separate entity from reuse and thus evaluated separately.

Assessing the end-to-end treatment and reuse value chain in this study has revealed that the reuse components of the entire chain that is FDF and its residual effect could be profitable with lettuce. Therefore, it should be considered in its entirety as a promising potential for attaining some financial viability. This would mean that there must be a new approach where treatment operators can include greenhouse vegetable production or crop farming as part of treatment to realise the full potential of the FS resource even beyond in Ghana and beyond. The entire system should be viewed on an economic viability basis, and not on financial viability merits only. This study recommends that these approaches be replicated in Ghana at scale led by the private sector with active support from Government on enabling environment and policy. Some of the enabling environment support could be tax waiver of truck/equipment for treatment as well as some subsidies products etc. Additionally, there could be new policies enacted to support circular economy treatment systems and certification. As promising as this is based on the findings of this study, it is also important to note that my study site was originally designed to have some demonstration farms within the treatment facility making it logistically convenient to have both treatment units and farming in one-stop shop. This may not necessarily be reflective of other existing FS treatment systems in Ghana and beyond. However, future treatment systems could consider this new approach in setting up.

Ghana, just like the rest of the sub-Saharan African countries is experiencing rapid population increases and increasing big cities (urban centers). This comes with the need to have adequate treatment infrastructure to meet the growing demand. Sustainable consumption and production as captured in SDGs entrusts us to employ circular economic treatment modules other than linear modules. Ghana is poised to harness the full potential of circular economy of waste due to increasing awareness on climate change issues and increasing knowledge and technological know-how. Decentralised treatment FS such as the one studied should be scaled rapidly. The overall contributions from the study largely confirm the thesis hypothesis which stated that "through optimisation of key variables, the co-treatment of FS and solid waste has the potential to recover nutrients and organic matter at International Standard safety levels whilst remaining economically viable for smallholder farmers. As such FDF could be considered an alternative to mineral fertilisers and a potential solution in the advancement of circular economy and food security."

8.4 **Overall Recommendations**

Following the overall conclusions drawn from this study, it is recommended that there is the need for strategies, approaches and technologies that are low-cost, simple, and environmentally friendly to reduce N losses during dewatering and composting stages. The co-composting stage and maturation pond treatment stages should never be avoided in a FS and solid waste treatment system. There is the need to explore causes of the within and between FS and FW composting batch variations as well as an investigation into ways to ensure consistency. The study recommends that NECo FDF should not be kept for more than four months in storage. There is the need for FDF application on a repetitive cycle (after the second cycle) to ensure yield decline does not occur to cause economic loss to farmers. The study recommends that smallholder farmers should be encouraged to use FDF Co to increase yield and farm income.

8.5 Contribution to knowledge

The contribution to the body of knowledge as outcome of this research work is presented in Table 8-1.

Table 8-1. Contributions to knowledge presented by the research work.

Domains of contribution	Extent of contribution		
	What has been confirmed	What has been developed	What has advanced knowledge
Theoretical knowledge		- The mass balance of nutrients (NPK) and *E. coli* in a decentralised faecal sludge and food waste treatment plant is established (Chapter 3 and 4). - The consistency levels of faecal sludge and food waste composting in batches has been identified (Chapter 5a) - There is residual effect of faecal derived fertilisers on crops making it a suitable alternative to mineral fertilisers	- The nutrient recovery rate and losses prevailing in a decentralised faecal treatment system have been quantified for the first time (Chapter 3).
Empirical evidence	- The characteristics of faecal sludge and food waste co-compost (faecal derived fertiliser) are not consistent between and within successive batches of production (Chapter 5a). - Indigenous *E. coli* regrowth is less likely (absent) in uninoculated faecal derived fertiliser in storage (Chapter 5b). - There is residual effect of applied faecal derived fertiliser which can support lettuce growth for up to 4 cultivation cycles without additional fertilisation (Chapter 6). - Economic benefit (profits) from plots amended with FDF Co-compost made 6 times more profit over control, about 3 times	- The nutrient recovery rates and loses per treatment cycle in a faecal sludge and food waste treatment system has been developed (Chapter 3). - Relatively higher NPK availability and conservation were achieved at lower storage temperatures (<25°C) and with longer storage duration (Chapter 5b). - The residual effect of one-time application of faecal derived fertilisers on lettuce has been established for up to 4 cultivation cycles (Chapter 6).	The N recovery efficiency of decentralised faecal sludge and food waste treatment plant is 20-47%. Between 50-70% of N from raw FS is lost at the dewatering stage. (Chapter 3). - At field scale, the co-composting of faecal sludge and food waste is not consistent in characteristics between and within successive batches of production. Replicate co-compost piles within batches exhibited coefficient of variation (CV) of measured parameters ranging between 0 – 125%. (Chapter 5a).

	more over mineral fertiliser from direct lettuce yields (Chapter 7).		- There is no indigenous regrowth of *E. coli* in stored faecal derived fertilisers under different storage temperatures and durations (Chapter 5b). - Residual effect of one-time application of faecal derived fertiliser can support lettuce yield for two (2) successive cultivation cycles before reapplying (Chapter 6).
Methodology		- Design of a robust and adaptable end-to-end assessment methods for faecal sludge and food waste treatment plant (Chapter 3 and 4).	
Knowledge of practice		- Co-composting of faecal sludge and food waste in successive batches is not consistent. One of factors contributing to this inconsistency is the variable nature of feedstocks. Producers who are looking to produce consistent quality of faecal derived fertiliser that meets international organic fertiliser requirements can tweak the feedstock mixing ratio to achieve quality (Chapter 5a). - The cost of faecal derived fertiliser is cheaper than mineral fertilisers and farmers can use it with greater confidence residual effects and increased profits (Chapter 6 and 7).	

8.6 Limitations of thesis

8.6.1 Sample collection and methodology

The design of sample collection and methods for the end-to-end assessment of the faecal sludge treatment plant followed real time treatment operations in Chapters 3 and 4. This feasible method did not allow for replication of sampling /experiments but rather for the sampling on repeated basis (repetitions). Due to the nature of operations, multiple sampling procedures were employed including grab and composite sampling at different points of sampling culminating into uneven samples numbers and sizes for analyses. It also paved way for uneven sampling frequencies between treatment cycles.

8.6.2 Time constraints

Due to time constraints and the timeframe for the PhD study, the execution of various experiments had to be adjusted not to follow the exact consecutive order of process flow. Ideally, the storage studies on faecal derived fertilisers in Chapter 5b would have preceded the lettuce field study in Chapter 6. However, these separate experiments had to be carried concurrently as each took more than 6 months to complete.

8.7 Future work

The future work will leverage on the data obtained in this study on the NPK and *E. coli* flows in the decentralised treatment system to build predictive models on nutrient recoveries and losses as well pathogen inactivation in decentralised FS and solid waste treatment systems in sub-Saharan Africa. Also, future research will be carried out on the consistency of FS and FW co-composting to determine a threshold value supported by an exploration of causes of within and between batch variations. This will supported by FS co-compost producers, users, regulators, and academia to create acceptable consistency levels.

Further research will also be carried out using farmer plots involving different crops such as cereals, legumes, and other vegetables to access the residual effects of FDF. However, an assessment of farmers willingness to use and pay for FDF following this new information on residual effects needs to be carried out to know the acceptance of these fertilisers. Farmers' acceptance and willingness to use FDF may also depend on the market potential of food products produced using the recommended fertilisers. Therefore, a consumer market survey involving a choice experiment is imperative for the sustainable adoption of FDFs.

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