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Wastewater Refining and Reuse and City-Level Water Decision Making

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ABSTRACT

Direct discharge of treated and untreated wastewater to natural surface water bodies, including rivers and lakes, is generally known to produce adverse environmental impacts that have been of concern for the past 10–15 years, particularly in the Great Lakes basin. Examples of these impacts include eutrophication and the emergence of pharmaceuticals and personal care products (PPCPs). Advancement in wastewater treatment plant operations alone may not be feasible to meet the 2016–2019 Federal Sustainable Development Strategy (FSDS), especially on phosphorus reduction targets. Diverting treated wastewater to land could present a multibarrier approach for integrated PPCPs management, and could also shift current phosphorus management from contamination to resource recovery.

Two case studies were considered: one from a developing country and the other from an industrialised country. City-level decision making in this paradigm was investigated in the context of developing countries, taking Ghana as a case study. In-depth, semi-structured interviews were conducted with senior officials of seven participating organizations from local governments, academia, and international non-governmental organizations in Ghana. The Analytical Hierarchy Process (AHP) was deployed to prioritize challenges as they were perceived by these institutions. Results show that social factors were the main barrier to wastewater reuse (31%), followed by financial (29%), institutional (24%), and technical challenges (16%). Strengths, weaknesses, opportunities, and threats (SWOT) analysis was also conducted. This was combined with a toxicological assessment of local waste stabilization ponds, whereby effluents were found to be suitable for reuse in agriculture as per the latest 2006 Food and Agriculture Organization guidelines.

Several On-site Sanitation Systems (OSSs), commonly known as septic tanks in Canada and the United States, pose a threat to groundwater and surface water, and have the potential to negatively impact human health over the long term. A new Biochar-Modified Soil Aquifer Treatment (BMSAT) system was investigated for its ability to resist or reduce toxicity and microbiological contamination as per the 2012 U.S. Environmental Protection Agency guidelines for water reuse in agriculture. The assessment included 18 chemical compounds, as well as *E. coli*. The BMSAT system showed promising results, and could be a possible alternative to existing OSSs in rural areas.

Keywords: Water reuse, Wastewater treatment, Eutrophication, Pharmaceuticals and Personal Care Products, Lakes, Impact, Socio-economic, Institutional challenges, Analytical Hierarchy Process, Decision Making, Interviews, Biochar, Soil Aquifer Treatment, Agriculture.

CO-AUTHORSHIP

This thesis has been prepared in accordance with the regulations for a Monograph thesis as stipulated by the School of Graduate and Postdoctoral Studies of the University of Western Ontario. Statements regarding the co-authorship of individual chapters are as follows:

Chapter 2: Why it is a Best Practice to Divert Treated Wastewater Away from Rivers and Lakes

All work was conducted by A. Abuhussein under close supervision of Dr. E. Yanful. Drafts of Chapter 2 were written by A. Abuhussein, and modifications were done under supervision of Dr. E. Yanful. A peer-reviewed journal article, co-authored by A. Abuhussein and E. Yanful, was submitted to the *Canadian Journal of Civil Engineering, NRC Research Press*.

Chapter 3: Local Level Governance and Management of Water Reuse in Agriculture: A case study from Ghana

All the numerical and experimental work was conducted by A. Abuhussein under close supervision of Dr. E. Yanful. Drafts of Chapter 3 were written by A. Abuhussein, and modifications were done under supervision of Dr. E. Yanful. A peer-reviewed paper, co-authored by A. Abuhussein and E. Yanful, was submitted to and published in the *2017 International Conference on Environmental Engineering, Canadian Society of Civil Engineers (CSCE)*. Another peer-reviewed paper, co-authored by A. Abuhussein and E. Yanful, was submitted to and published in the *2017 Water Technology and Environmental Control (WATEC) Conference*. A peer-reviewed journal article, co-authored by A. Abuhussein and E. Yanful, was submitted to the *Journal of Water Policy, International Water Association (IWA) Publishing*.

Chapter 4: Biochar-Modified Soil Aquifer Treatment for Water Reuse in Agriculture

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To my loving and beloved mother and father ***Khawla*** and ***Mohammed Abuhussein***

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To my caring sisters ***Alaa*** and ***Asmaa***

To my supportive brothers

To my supervisor ***Ernest Yanful***

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LIST OF ACRONYMS AND ABBREVIATIONS

3GC	Third-Generation Cephalosporin
AAS	Atomic Absorption Spectrophotometer
AC	Academia
ADP	Adenosine Diphosphate molecule
AEC	Anion Exchange Capacity
AHP	Analytical Hierarchy Process
AMR	Anti-Microbial Resistance
AOPs	Advanced Oxidation Processes
APU	Ani-Pollution Unit
ARBs	Antibiotic-Resistant Bacteria
ARGs	Antibiotic-Resistant Genes
ASP	Amnesic Shellfish Poisoning
ATP	Adenosine Triphosphate molecule
Avg	Average value
B	Boron
BC	*Conocarpus* Biochar
BCIG	A chromogenic agent: 5-bromo-4-chloro-3-indolyl-β-D-glucuronide
BMPs	Best Management Practices
BMSAT	Biochar-Modified Soil Aquifer Treatment
BOD_5	5-days test of the Biological Oxygen Demand
°C	Degrees in Celsius
ca	Circa

Ca	Calcium
CEC	Cation Exchange Capacity
CFs	Characterization Factors
CFP	Ciguatera Fish Poisoning
CFU	Colony Forming Unit
CI	Consistency Index
CIDA	Canadian International Development Agency
cm	Centimeter
CO_2	Carbon dioxide
Coliforms	Normal bacteria of the enteric tract of mammals used as an indicator of fecal pollution
CR	Consistency Ratio
DCI	Dichloroisocyanuric acid
DW	Distilled water
E	Effluent
EC	Electrical Conductivity
E. coli	A more specific fecal indicator organism; see coliforms
EDTA	Ethylenediaminetetraacetic acid
Eff	Effluent; see E
EIA	Environmental Impact Assessment
EPA	Environmental Protection Agency
FAO	Food and Agriculture Organization
FC	Fecal Coliforms

FR	Farm Residues
g, gm	Gram
GAC	Granular Activated Carbon
GDP	Gross Domestic Product
H_2O_2	Hydrogen Peroxide
H_2SO_4	Sulfuric Acid
ha	Hectare = 10,000 m^2 = 2.5 acres
HABs	Harmful Algal Blooms
HCl	Hydrogen Chloride
HKT	High-affinity Potassium Transporters
I	Influent
IC	Ion Chromatography
IC_{50}	Half Maximal Inhibitory Concentration
ICP-OES	Inductively Coupled Plasma Optical Emission Spectrometry
IISD	International Institute for Sustainable Development
INGO	International Non-Governmental Organization
ISO	International Organization for Standardization
IWM	Integrated Water Management
K	Potassium
kg	Kilogram
km	Kilometer
$KMnO_4$	Potassium permanganate
K_{ow}	Octanol/Water partition coefficient

KVA	Kilo Volt Ampere
L	Litre
LA	Local Authority
λ_{max}	The principle eigenvalue
LCA	Life Cycle Assessment
Log_{10} Removal	Removal efficiency expressed in log_{10} units; i.e. 1 log_{10} unit = 90%, 2 = 99%, 3 = 99.9%, and so on.
M	Mole per litre
m^2	Square meter
m^3	Cubic meter
meq	Milli-Equivalent
µm	Micro-meter
µM	Micromole per litre
mg	Milligram
mL	Milliliter
mmhos	Millimhos
Max	Maximum
MDR	Multi-Drug Resistant
MDL	Method Detection Limit
MFA	Ministry of Food and Agriculture
Mg	Magnesium
MHz	Megahertz
Min	Minimum

MPN	Most Probable Number
mS	Millisiemens
n	Number of Samples; Number of Criteria
N	Nitrogen
Na	Sodium
NaOH	Sodium hydroxide
ng/L	Nanogram per litre
NGO	Non-Governmental Organization
NI	No Information
NIST	National Institute of Standards and Technology, USA
NSP	Neurotoxic Shellfish Poisoning
O_3	Ozone
OACC	Organic Agriculture Centre of Canada
OCC	Opportunity Cost of Capital
OSSs	On-site Sanitation Systems
P	Phosphorus
PAC	Powdered Activated Carbon
PDF.m^2.year	Potentially Disappeared Fraction per Square Meter of land per Year
pH	A measure of acidity
PO_4^{3-}	Phosphate
ppb	Part Per Billion
PPCPs	Pharmaceuticals and Personal Care Products
ppm	Part Per Million

PSP	Paralytic Shellfish Poisoning
PVC	Polyvinyl Chloride
RI	Consistency Index of a Random-like matrix
RF	Radio Frequency
RGMM	Row Geometric Mean prioritization Method
SAR	Sodium Adsorption Ratio
SAT	Soil Aquifer Treatment
SD	Standard Deviation
SRM	Standard Reference Materials
SWOT	Strengths, Weaknesses, Opportunities, and Threats analysis
SS	Suspended Solids
SVDV	Synchronous Vertical Dual View
TDS	Total Dissolved Solids
TKN	Total Kjeldahl Nitrogen
TCLP	Toxicity Characteristic Leachate Procedure
TN	Total Nitrogen
TP	Total Phosphorus
U.K.	United Kingdom
UNDP	United Nations Development Program
UNEP	United Nations Environment Program
U.S.	United States of America
UV	Ultra-Violate
V	Volume of digestate

v/v	volume/volume percent
WHO	World Health Organization
WSP	Waste Stabilization Pond
WWTP	Wastewater Treatment Plant

RESOURCES

You are the daughter of the sea, oregano's first cousin;
swimmer, your body is pure as the water;
cook, your blood is quick as the soil;
everything you do is full of flowers, rich with the earth.

Your eyes go out toward the water, and the waves rise;
your hands go out to the earth and the seeds swell;
you know the deep essence of water and the earth;
conjoined in you like a formula for clay.

Pablo Neruda (1904 – 1973)

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Chapter 1

Introduction

1.1 Background and motivation

Municipal wastewater treatment plants are known to be point sources of a broad suite of contaminants to the Great Lakes Basin at concentrations high enough to potentially impact aquatic organisms, human health, and the economy at large. Treated wastewater discharge to natural surface water bodies contribute significantly to phenomena such as eutrophication and input of emerging contaminants including pharmaceuticals and personal care products (PPCPs) which have started to emerge at rates higher than ever. Upgrading wastewater treatment plants to include advanced processes may not be feasible to meet the 2016 – 2019 Federal Sustainable Development Strategy.

On the other hand, primary agriculture plays a vital role in the food sector which is linked to close to $100 billion per year in economic activity and approximately 1 in 7.5 jobs (CCA, 2013). Each year, Ontario generates approximately 300,000 dry tons of municipal sewage biosolids (CIELAP, 2009). Some 40% are applied to land, 40% go to landfills and 20% are incinerated. Some municipalities and industry are diverted them from landfilling and incineration due to their economic and environmental costs. More proactive measures may be required for their use in order to offset agricultural water requirements.

Unsustainable development pathways and governance failures have generated immense pressures on water resources, affecting their quality and availability and, in turn, compromising the ability of these resources to generate social and economic benefits. The United Nations World Water Development 2015 report found that Ghana is currently water vulnerable, and at current trends, it will become a water stressed country by 2025. Agriculture is the most water-consuming sector worldwide, especially with intensive agricultural activities practiced in rural areas of regions such as sub-Saharan Africa. Treated wastewater provides reliable irrigation water supplies and, as a farming practice, contributes to both the improvement of urban food supply and the livelihood of many farmers and produce traders.

The agricultural sector of Ghana accounts for about 65 per cent of the work force, about 40 per cent of the gross domestic product, and about 40 per cent of foreign currencies acquired through exports. Wastewater collection, treatment, and disposal or distribution is a persistent problem. Even if only 30 per cent of wastewater is treated and reused for agriculture, it could irrigate up to 13,800 hectares (i.e. 31 per cent of the total formally- and informally-irrigated agricultural land), while offsetting the use of fresh water sources and providing livelihood support for more than 9,000 farmers in peri-urban areas (Namara *et al*., 2011).

Urban farming can make substantial contributions beyond the provision of livelihoods and food. These include contributions to buffer zone management and flood control, thus supporting climate change adaptation strategies, land reclamation, land protection, resource recovery (from waste), urban greening, and biodiversity conservation.

Canadians are worried their bucolic backyard is fast becoming their outhouse, and are concerned septic systems may be leaking into groundwater. On-site septic treatment of sanitary wastes is proliferating throughout the Great Lakes basin – serving more than 50% of new housing in some areas, with effluent filters, required on newly installed septic systems, are not common on existing systems (GCC, 2015). Moreover, at least 40% of existing systems fail to treat wastes adequately. Leaky sewer and waterlines are of concern with 30% conveyance loss being common, and thousands of line breaks occur in the basin every year (GCC, 2015). Few jurisdictions monitor or regulate these systems in any systematic way.

One of the main motivations for this study is to investigate the suitability of a biochar-modified SAT for wastewater treatment in rural areas of Canada where on-site sanitation systems (OSSs), known as septic tanks, are used. Many rural communities rely on aging individual septic systems or drain tile networks that discharge sewage directly to surface waters, even though direct discharge of untreated sewage is illegal. In Ontario alone, there are an estimated 1.2 million septic systems potentially posing a threat to public health and the environment (AMO, 2008). Approximately 25,000 new or replacement OSSs are installed annually in Ontario with similar numbers installed in each of the Great Lakes states each year (IJC, 2010).

The Soil Aquifer Treatment (SAT) system is defined as a three-stage wastewater treatment process involving infiltration zone, vadose zone, and aquifer storage. Removal of pollutants occurs via physical, chemical and biological processes in the unsaturated and saturated zones. SAT is a relatively low-cost system, and can be an alternative to fresh water sources for: agricultural and park irrigation, sea water intrusion, and municipal uses. SAT systems were found to be capable of:

- sustaining the removal of organic carbon, nitrogen, and pathogens;
- producing water similar in structure, reactivity, and behavior to natural organic matter;
- providing anaerobic ammonia oxidation environment; and
- being resilient to pathogens and pathogen indicators.

The main advantages of SAT include: (i) SAT improves the physical, chemical and microbial quality of source water during soil passage by removing particles, microorganisms, heavy metals, nitrogen, bulk organic matter and organic micropollutants, (ii) it can be integrated with other conventional and advanced wastewater treatment systems to produce the water of desired quality for intended use, and (iii) it can serve as environmental and psychological barrier, thus increasing public acceptability of reclaimed water and promoting water recycling and reuse. However, local hydrogeological conditions may affect its permeability.

Biochar is used for soil application because of its capacity to improve soil organic content, and it was reported to have increased crop yield. Biochar is also credited for its water holding capacity, thus improving soil structure. Other applications include bio-energy to reduce dependency on fossil fuel, while offsetting carbon emissions which contributes to climate change strategic plans. However, literature lacks information on the use of biochar in commercial wastewater treatment, which could indicate lack of its assessment and or application in the removal of contaminants from wastewater, so it can be considered for application on land.

Latest wood waste survey conducted in 2004 concluded that almost 1 million tonnes are being disposed of (excluding that reused for bioenergy and recycling) yearly in Canada, with a rate of 120,000 tonnes per year for Ontario alone (Kelleher, 2007). Biochar, produced from the pyrolysis of agricultural waste and forest industry by-products as feedstock, can be used in several applications, including: (1) soil amendment, (2) carbon sequestration, (3) pollution prevention through better management of agricultural waste and run-off control, and (4) reuse potential as energy source.

1.2 Scope and objectives

The aim of this thesis is to provide better understanding of wastewater treatment, reuse, and governance. The objectives are to:

i. investigate the need for, and potential of, sustainable practices for treated wastewater management in Canada. This objective was accomplished by:

- Reviewing the rationale and justification for reusing treated wastewater;
- Understanding the impacts of current treated wastewater management actions on ecosystem, human and animal health, and their economic consequences;
- Exploring new alternatives to current treated wastewater management practices;
- Reviewing evidence of the potential of treated wastewater application on land in alleviating current issues of concern; and
- Examining the suitability of reusing treated wastewater, from financial, social, and institutional perspectives.

ii. examine the feasibility of using treated wastewater in agriculture taking Ghana as a case study of a typical developing country. This objective was accomplished by:

- Providing an updated and informative review of treated wastewater status and current characteristics of wastewater treatment plants in Ghana;
- Understanding current practices with regard to treated wastewater reuse in agriculture;
- Reviewing the latest toxicity requirements for treated effluent reuse in agriculture, and how this is reflected on cultivated crops in Ghana;
- Examining the role of waste stabilization ponds in protecting the environment and alleviating health risks;

- Outlining the potential economic benefits and institutional challenges related to applying treated wastewater to agricultural land.
- Developing a hierarchy of the main challenges to treated wastewater reuse in agriculture;
- Developing better understanding of the scale of various challenges using the analytical hierarchy process as a decision-making tool;
- Analyzing treated wastewater toxicity against international guidelines for application on land; and
- Providing informative evidence on the toxicological suitability of using treated wastewater for agriculture.

iii. investigate a Biochar-Modified SAT (BMSAT) system for the treatment of wastewater for reuse on land, as per the 2012 US Environmental Protection Agency (EPA) guidelines. This objective was accomplished by:

- critically discussing post-treatment effluent toxicity against the EPA criteria guidelines for reuse in agriculture, and
- evaluating the microbiological quality of treated effluent against these guidelines.

1.3 Innovation and thesis contribution

The thesis demonstrated a link between current wastewater management practices and eutrophication of the Great Lakes, especially Lake Erie and Lake Ontario, and highlights similar cases in the U.S. and Europe. Harmful algal blooms and hypoxia can cause morbidity and mortality of birds and marine mammals, kill fish or shellfish directly, cause loss of submerged vegetation, and affect aquaculture and biodiversity negatively. Evidence of the impact of eutrophication on biodiversity is most notable in the Golden Shoe, where treated wastewater is discharged to rivers and streams.

Shifting from current practices to more sustainable wastewater-agriculture management could prove particularly beneficial to the environment, health and economy. Currently, over 150 billion litres of untreated and undertreated wastewater are discharged into our waterways every year.

The thesis provided original evidence on various technical, social, institutional, and financial challenges to wastewater reuse in agriculture in a developing country, taking Ghana as a case study. For the first time, the thesis provided insight into the perceived ranking of these challenges from a variety of sectors. These sectors included local governments, academia, and non-governmental organizations.

The Analytical Hierarchy Process (AHP) was used to prioritize challenges as they were perceived by these institutions. Results show that social factors were the main barrier to wastewater reuse (31%), followed by financial (29%) and institutional (24%) challenges. Technical challenges were significantly at lower levels (16%). A strengths, weaknesses, opportunities, and threats (SWOT) analysis was provided in this study. Interestingly, community engagement schemes proved effective, whereas partnerships with local governments may likely have resulted in greater institutional barriers.

In this thesis, an innovative wastewater treatment unit was designed to substitute current OSSs in rural areas. The new BMSAT system was examined for its toxicological and microbiological quality for the reuse of effluents in agriculture as per the 2012 U.S. EPA guidelines. The system was found to meet these criteria.

1.4 Thesis overview

The thesis is divided into five chapters. The current chapter is Chapter 1 and provides the general overview of the thesis. Chapter 2 scientifically reviews the pressing need to reuse treated wastewater on land. Chapter 2 also highlights the inherent ability of pharmaceuticals, as a unique group of emerging contaminants, to: (1) affect the ecological system, (2) develop new strains of bacteria resistant to antibiotics and other drugs, and (3) change the reproductive hormones in humans. Cases from around the world, and studies in Canada and Ontario, show the long- and short-term scales of impact of emergent contaminants presence in our water bodies. Conventional wastewater treatment plants are not typically equipped to treat pharmaceuticals due to the lack of both related regulations and established procedures, besides the high cost of high-end technologies.

This chapter proposes a multibarrier approach for integrated PPCPs management application of treated wastewater on land. The concept is based on natural, yet proven, degradation processes that have strong potential to alleviate the issue, while keeping costs at a minimum. Such processes include photodegradation, biodegradation, physical adsorption, and plant uptake. If we are to learn from the source-pathway-receptor pollution approach, the application of treated wastewater on land represents a pathway that is significantly more interrupted than direct discharge to rivers and lakes, especially for humans.

Socio-economic consequences were highlighted. The losses in Lake Erie’s value as a non-market (ecosystem) and market asset due to the algal blooms were estimated to be \$3.8 billion and \$4 billion, respectively (IISD, 2017). Recurrent costs are found in public health, housing, infrastructure, and other sectors. For example, Environment and Climate Change Canada estimated that a decline in water quality due to algal blooms would lead to up to a 6 per cent decrease in the value of all residential property within one kilometre of the Canadian shoreline of Lake Erie (MAEE, 2015). The willingness to pay for eutrophication as an index of significance was highlighted.

Treated wastewater reuse presents an opportunity to recover phosphorus, a non-renewable resource and a macro-plant nutrient, while mitigating the eutrophication of the affected lakes in the Great Lakes basin. Phosphorus was found to be the single limiting factor to eutrophication. On the other hand, many important agricultural regions in Canada, including parts of the Prairies and portions of British Columbia, are already water-stressed, and concerns about water quality exist throughout most of Canada’s agricultural lands. The chapter examines suitability of treated wastewater reuse on land as a new paradigm. Studied factors include financial affordability, social acceptability, institutional challenges and legal framework.

Chapter 3 addresses city-level water governance and management of water reuse in agriculture, taking Ghana as a case study. The agricultural sector in many developing countries is the core of the economy and is a major driver and influence to its work force and trading markets. However, the industry is the most water-consuming sector nationally, and at current trends, Ghana will become water-stressed in the near future.

Sustainable agriculture can only be achieved by integrating the wastewater and agricultural sectors. Given that the situation is worsened in urban areas due to high population density, there is a strong link between the lack of wastewater treatment and the use of polluted water (sewage streams, surface water and even contaminated underground water) in irrigated urban agriculture.

The wellbeing of plants requires soil to be *both* well-structured and nutrients-sufficient. Soil quality is deteriorating due to intensive agricultural practices over decades, resulting in much lower mineral and organic content. As in many developing countries, the disposal of domestic wastewater and sewage sludge is poorly controlled, and can lead to significant health and environmental problems. Fortunately, wastewater is rich with organic matter, minerals and macro- and micro-nutrients essential for plant growth, and it could serve as a competitive alternative to imported chemical fertilizers. It is the cheapest and most sustainable way to both improve soil characteristics and at the same time solve waste management problems.

Water governance is critical to the sustainable provision of treated wastewater for agriculture. The main constraints to this paradigm were identified, and the analytical hierarchy process was utilized to determine the scale of perceived impact that they represent to institutions in the water sector. Local authorities can play a key role in implementing the national green policy at the local government level, through either provision, partnership, promotion or self-governance. Ghana’s governance is based on decentralised power to local assemblies, and, along with a strategic vision and a better understanding of the incentives and barriers, the country has the potential to follow other successful water management experiences.

The chapter then reviewed the latest toxicity requirements for treated effluent reuse in agriculture, and how this is reflected in cultivated crops. It examined the role of waste stabilization ponds in alleviating environment and health risks, outlined the potential economic benefits and institutional challenges related to applying treated wastewater to agricultural land, It then developed a hierarchy of the main challenges to treated wastewater reuse in agriculture, developed better understanding of the scale of various challenges using the analytical hierarchy process as a decision-making tool, provided analysis of treated wastewater toxicity against international guidelines for application on land, and provide informative evidence on treated wastewater toxicological suitability for reuse in agriculture.

Chapter 4 provides a potential solution to millions of On-site Sanitation Systems (OSSs), commonly known as septic tanks in Canada and the United States which pose a threat to ground water quality and natural surface water bodies, and have the potential to affect human health over the long term. OSSs serve more than 50% of new housing in some areas of the Great Lakes area, and their waste is proliferating throughout the basin. At least 40% of existing systems fail to treat wastes adequately. Unsuitable soils result in OSSs malfunction by leaking untreated sewage up to the ground surface or a roadside ditch, or leaching to ground water. While effluent filters are now required on newly installed OSSs, they are not common on existing systems. Few jurisdictions effectively and systematically monitor or regulate these systems. If we are to learn from the Walkerton water crisis (as a result of groundwater contamination with *E. coli* in 2000), the vulnerability of ground water requires extensive soil and site assessment, strict regulatory and enforcement measures, and improved design, construction, operation and maintenance of wastewater treatment and discharge facilities, especially in rural areas.

Effluents from these septic systems are discharged to leaching beds, where Soil Aquifer Treatment (SAT) occurs. The removal of pollutants is influenced by physical, biological, and chemical processes in the infiltration, vadose, and aquifer storage zones of the SAT system. The environment favours the removal of pathogens and nitrogen. SAT can serve as environmental and psychological barrier, thus increasing public acceptability of water recycling and reuse. It is a relatively low-cost system, and effluents can be an alternative to fresh water sources for agricultural and park irrigation, salt water intrusion, and municipal uses. However, local hydrogeological conditions can affect the permeability of soil, thus the removal of not only bacteria, viruses, and helminth eggs, but also metals and ions, heavy metals, and organic carbon.

In this chapter, a new biochar-modified SAT (BMSAT) system was investigated for its toxicological and microbiological performance as per the 2012 U.S. Environmental Protection Agency guidelines for water reuse in agriculture. The assessment included 18 chemical compounds, as well as *E. coli*. A laboratory scale BMSAT system was built using a polyvinyl chloride column with an internal diameter of 5 cm and an effective length of 90 cm, and the ratio of biochar to sand was 1:1. Raw wastewater was collected from the Vauxhall Wastewater Treatment Plant (London, Ontario), which receives industrial and domestic wastewater discharges. The Upper Thames River Source Protection Area Assessment Report identified that the Plant site is located within a Highly Vulnerable Aquifer Area, and is within a Significant Groundwater Recharge Area. Since the targeted end use of effluents is reuse in agriculture, the BMSAT system was assessed for its suitability for irrigation of crops according to their tolerance to each tested element. The adsorption characteristics of biochar could improve the performance of the modified SAT system, which would serve as a practical application in rural areas where OSSs mostly exist.

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Chapter 2

Why it is a Best Practice to Divert Treated Wastewater Away from Rivers and Lakes

2.1 Introduction

Municipal Wastewater Treatment Plants (WWTPs) have been identified as point sources of a broad suite of contaminants to the Great Lakes Basin at concentrations high enough to potentially impact aquatic organisms, human health, and the economy at large. Treated wastewater discharge to natural surface water bodies contribute significantly to phenomena such as eutrophication and contamination by emerging contaminants including pharmaceuticals and personal care products (PPCPs). Upgrading wastewater treatment plants to include advanced processes may not be feasible to meet the 2016 – 2019 Federal Sustainable Development Strategy.

On the other hand, primary agriculture plays a vital role in the food sector which is linked to close to $100 billion per year in economic activity and approximately 1 in 7.5 jobs. Each year, Ontario generates approximately 300,000 dry tons of municipal sewage biosolids. Some 40% are applied to land, 40% go to landfills and 20% are incinerated. Some municipalities and industry are diverted them from landfilling and incineration due to their economic and environmental costs. More proactive measures may be required for their use in order to offset agricultural water requirements.

2.1.1 Aim and objectives

The aim of the research is to investigate the need for, and potential of, sustainable practices for treated wastewater management. The objectives are to:

- Review the rationale and justification for reusing treated wastewater;
- Understand the impacts of current treated wastewater management actions on ecosystem, human and animal health, and their economic consequences;
- Explore new alternatives to current treated wastewater management practices;
- Review evidence of the potential of treated wastewater application on land in alleviating current issues of concern; and

- Examine the suitability of reusing treated wastewater, from financial, social, and institutional perspectives.

2.2 Reasons to reuse treated wastewater

2.2.1 Reason I: Pharmaceuticals and Personal Care Products (PPCPs) in treated effluents

Pharmaceutical compounds discharged into freshwater resources are contaminating sources of drinking water in Canada (CWN, 2015). Pharmaceuticals, as a unique group of emerging contaminants, have been found to have the inherent ability to: (1) affect the ecological system, (2) develop new strains of bacteria resistant to anti-biotics and other drugs, and (3) change reproductive hormones in humans. Studies have found toxic effects of pharmaceuticals in the environment (particularly in the aquatic ecosystem; Sections 2.2.1.1 and 2.2.1.2), but more concerning is the continuous discharge of pharmaceuticals that are resistant to degradation, including biodegradation. Conventional wastewater treatment plants are not typically equipped to treat pharmaceuticals due to the lack of both related regulations and established procedures, besides the high cost of high-end technologies.

2.2.1.1 Impact on the ecological system

In contrast to humans who experience limited exposure to waterborne materials (i.e. orally), aquatic organisms could be exposed to aqueous contaminants throughout their entire lifetimes (NACWA, 2009). Studies by Sumpter and Johnson (2008), which included sampling of roach fish throughout several waterways, indicated male fish below wastewater outfalls were undergoing feminization or endocrine disruption.

There is significant evidence of feminization in aquatic species (such as wild *Elliptio complanata* mussels, fathead minnows, common carp, and walleye fish, among many others) in receiving waters (Mille-Îles River in Quebec, Lake Michigan, Lake Mead in Nevada, and Lake Nippissing, respectively) downstream of a municipal effluent outfall (Rosen *et al*., 2004; Gagné *et al*., 2011; Roy, 2014; Niemuth and Klaper, 2015). Dose-response studies indicated compounds such as estrones, estradiol, aromatic hydrocarbons, triclosan, paracetamol, ibuprofen, and carbamazepine were able to induce vitellogenin, intersex and gender shifts to females at nanogram per litre concentrations at varied durations of exposure. A study in northwestern Ontario, Canada, showed that chronic exposure of fish to estrogen concentrations as low as 5 to 6 ng/L led to feminization of males (Kidd *et al*., 2007).

In more recent studies, exposure of fish to wastewater effluent was also found to affect behaviour (Brandão *et al*., 2011; Lorenzi *et al*., 2012; Tierney *et al*., 2017). McCallum *et al*. (2017a) exposed the round goby fish to wastewater effluent and observed reduced aggressive acts, thus reducing their survival (McCallum *et al*., 2017b). The study found concentrations of PPCPs in fish tissue being highest in the brain, followed by plasma, then gonads, then liver, and muscles (McCallum *et al*., 2017). Low levels of pharmaceuticals and anti-bacterial drugs (triclosan and triclocarban) were found to alter nest protection and mating behaviour in fish (Lovett, 2010). Exposure to estrogenic chemicals also induced gene expression changes in brackish medaka (Chen *et al*., 2016).

2.2.1.2 Development of new strains of bacteria

Wastewater discharges can promote the spread of antibiotic-resistant bacteria (ARBs) and antibiotic-resistant genes (ARGs) in streams and small rivers (Yuan *et al*., 2015; Hocquet *et al*., 2016). Recent studies have shown that concentrations of antibiotics found in aquatic environments could aid selection of resistant bacteria (Henriques *et al*., 2016; Dincer and Yigittekin, 2017).

A study in Spain assessed antibiotic resistance in river biofilms (i.e. microbial layers found on rocks, plants and other surfaces in rivers). The study found that antibiotic-resistant bacteria can integrate into these biofilms, which may provide an optimal environment for the exchange of genetic material, including genes encoding resistance to antibiotics (Proia *et al*., 2015). These new genes were detected as far as 1 km downstream of the studied Wastewater Treatment Plants, which suggests resistance genes can persist in the environment even in the absence of an additional pollution source, perhaps due to the ‘drift’ of antibiotic-resistant bacteria or resistance genes in the water flow.

Li *et al*. (2010) found that multidrug-resistant (MDR) phenotypes were found in 97% on wastewater effluent samples, and in 28% in river samples, as far as 20 km downstream. Amos *et al*. (2014) found third-generation cephalosporin (3GC)-resistant *E. coli* were seven times more common downstream of a U.K. wastewater treatment plant than upstream. These bacteria spread during flooding, and rain could make it worse (Wellington, 2014).

Antibiotic-resistant bacteria were also found in conventional wastewater treatment plants that receive untreated hospital wastewaters and drug manufacturers effluents (Katouli *et al*., 2012; Marathe *et al*., 2013). Chlorination, as a typical disinfection process in these plants, was considered not effective in controlling antimicrobial resistance. About 40% of erythromycin-resistant genes and 80% of tetracycline-resistant genes could not be removed by chlorination (Yuan *et al*., 2015).

Advanced wastewater treatment by ozonation in combination with different filtering techniques has not fully eliminated ARBs, while antibiotic resistant *E. coli*, *Staphylococcus aureus*, *staphylococci*, *E. faecium*, *E. faecalis*, and *E. casseliflavus* survived (Lüddeke *et al*., 2015). Research determines that physical, chemical, and biological treatment processes result in reduction of total numbers of bacteria (Schumacher *et al*., 2003; Abdel-Raouf *et al*., 2012; Cydzik-Kwiatkowska and Zielińska, 2016). However, the treatment process plays a significant role in diversifying the bacterial communities, including transferring of genes onto non-resistant bacteria (Auerbach *et al*., 2007; Ye and Zhang, 2013; Ojer-Usoz *et al.*, 2014; Atashgahi *et al*., 2015; Guo *et al*., 2017; Yamashita *et al*., 2017).

Therefore, conditions in wastewater treatment plants are favourable for the proliferation of ARBs and ARGs (Bouki *et al*., 2013). In Canada, exposure of fathead minnow to conventional activated sludge effluent was found to result in considerable mortality, and reduced growth and egg production (Parker, 2015).

Canada is a member of the Joint Programming Initiative on Antimicrobial Resistance (JPIAMR, 2015). Part of the Initiative's mandate is to discover new preventative approaches, and find a positive impact on public health. However, prioritization of the issue and mobilized action to deliver change are needed (DOH, 2013). The risk associated with these new strains of bacteria promoted the development of the U.K.'s 5-year Antimicrobial Resistance (AMR) Strategy (2013 – 2018). The European Union has also invested heavily in this area in their Framework Programme (FP7) for Research and Technological Development, and its successor: Horizon 2020.

2.2.1.3 Potential effect on humans

Humans and wildlife may be exposed to PPCPs through fish and shellfish consumption (Baker *et al*., 2013). In addition, humans are exposed to endocrine disruptors via polluted drinking water. As a result, there is much concern about the effects of elevated concentrations of xenobiotics in the environment on human health because exposure to endocrine disruptors may lead to premature puberty in human females (Swan, 2008). In males, they inhibit the action of hormones, or alter the normal regulatory function of the endocrine system, and/or mimic natural hormones, thus they have potential hazardous effects on male reproductive axis causing infertility (Sikka and Wang, 2008; Diamanti-Kandarakis *et al*., 2009; Vandenberg *et al*., 2009; Jeng, 2014). Studies in three cities (Boston, Massachusetts; Copenhagen, Denmark; and Turku, Finland) show significant evidence that (Swan, 2008): (1) some aspects of human male reproductive health are deteriorating (at least in some parts of the world), and (2) these declines are associated with early exposure to a range of hormonally active xenobiotics.

Some substances were detected also in human body, for instance in blood, fat, and breast milk indicating also the tendency for bioaccumulation in humans (Tang *et al*., 2013). Phthalates, known for their endocrine-disruptive effects found in wastewater effluents (Clara *et al*., 2010), were detected in pooled breast milk samples from American women (Calafat *et al*., 2004), as well as a Danish-Finish cohort (Main *et al*., 2006).

Levels of phthalates in babies' urine samples showed that babies (both male and female) were exposed to these ubiquitous chemicals (Sathyanarayana *et al*., 2008). Although long-term consequences of early postnatal exposure have not been examined, recent data from Main *et al*. (2006), Braun *et al*. (2013), and others suggest that alteration of male reproductive capacity may have been caused by other reasons. Phthalate exposure has also been reported to be associated with obesity (Dong *et al*., 2017).

However, toxicological impact of micropollutants on larger organisms is rather difficult to determine, particularly the chronic toxicity of micropollutants in sub-toxical range is rather an unknown field hosting numerous questions to be answered (Mulder *et al*., 2015). Although testicular and prostate cancers, undescended testis (Virtanen and Adamsson, 2012), altered pituitary (Waye and Trudeau, 2011), chronic inflammation (Dietert, 2012), Sertoli-cell-only pattern, abnormal sexual development, hypospadias (Botta *et al*., 2014), and thyroid gland functions (Schmutzler *et al*., 2007; Boas *et al*., 2012) are also observed, the available data are insufficient to deduce worldwide conclusions. It remains that the link between endocrine disruptors and these observations is highly plausible.

2.2.1.4 The cost of addressing PPCPs by upgrading conventional wastewater treatment plants

The cost of PPCPs removal from wastewater effluents at full scale have not been studied in Canada. However, oxidative and separative tertiary treatment options (such as ozonation, electrocoagulation, membrane filtration) are typically used for this purpose, indicating high manufacturing, operating and maintenance costs (Oulton *et al*., 2010; Singh, 2012; Ensano *et al*., 2017). Some proved to be significantly inefficient such as UV radiation (Roccaro *et al*., 2013).

Mulder *et al*. (2015) provided general cost estimates for the Netherlands based on implemented full scale post-treatment of effluents of wastewater treatment plants in Germany and Switzerland. The residues of pharmaceuticals, personal care products, pesticides, biocides, plasticizers, flame retardants, and several other industrial chemicals were being targeted in these plants. The study focused on the current knowledge of the removal of emerging contaminants (or micropollutants) from effluents using: (1) ozonation followed by slow sand filtration (Figure 2.1), (2) powdered activated carbon followed by slow sand filtration (Figure 2.2), and (3) granular activated carbon (Figure 2.3). For wastewater treatment plants serving 100,000 people, average total costs were €0.18 ($0.27), €0.20 ($0.30), and €0.27 ($0.40) per m^3 of treated effluent, respectively. Another issue currently being intensively studied is the by-products arising from chemical and physical interventions (such as ozonation and the use of nanomaterials) and their toxicity.

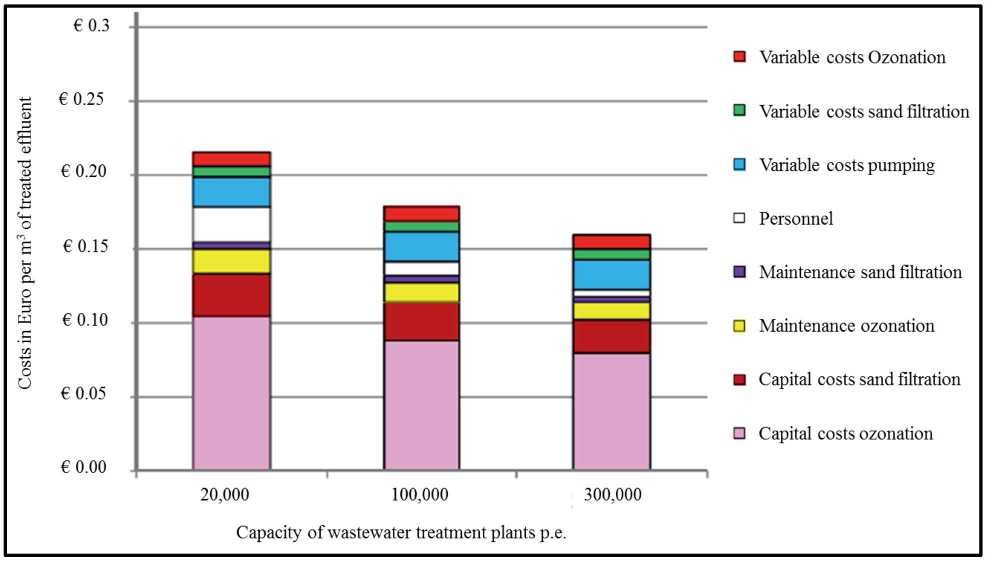

Figure 2.1 Yearly capital and operational costs to remove micropollutants through ozonation followed by sand filtration. Adopted from Mulder *et al.* (2015).

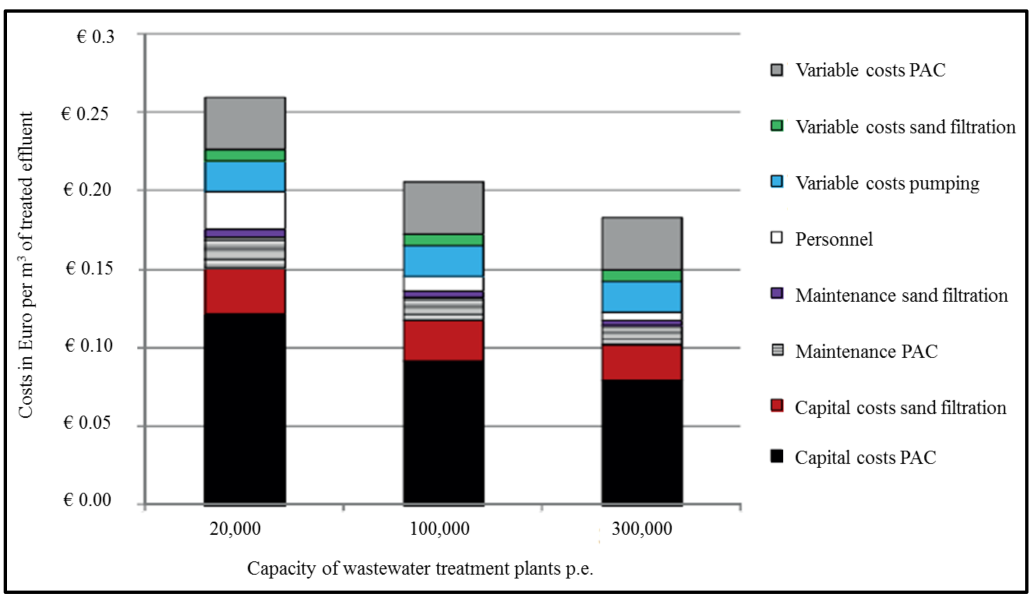

Figure 2.2 Yearly capital and operational costs to remove micropollutants through PAC followed by sand filtration. Adopted from Mulder *et al*. (2015).

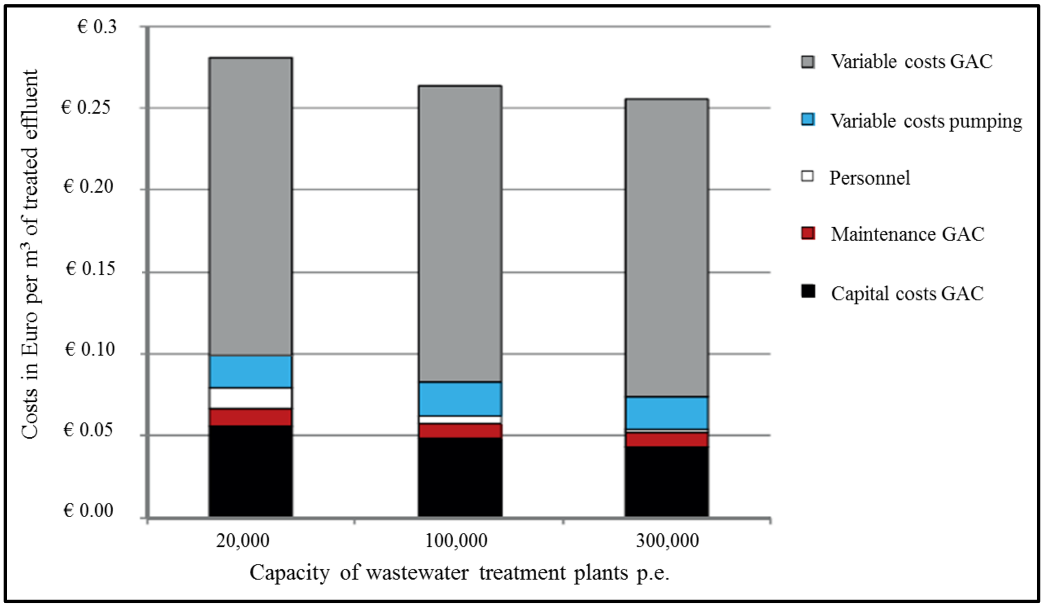

Figure 2.3 Yearly capital and operational costs to remove micropollutants through GAC. Adopted from Mulder *et al*. (2015).

Taking for example the City of London Ontario, more than 67 million m^3 of treated wastewater were discharged to natural surface water in 2014 (COL, 2014). If ozonation for instance is to be used, it may cost the City at least $17 million each year to partially address micropollutants in treated effluents.

2.2.1.5 Application of treated wastewater on land: A multibarrier approach for integrated PPCPs management

Complete removal of pharmaceuticals in wastewater treatment plants is not possible (NACWA, 2009; Ensano *et al*., 2017). Therefore, they cannot be relied upon to be the only mechanism for controlling the entry of pharmaceuticals to the aquatic environment, and eventually to humans if natural water bodies are used for drinking water. The limited capability of wastewater treatment plants warrants a new holistic approach to manage pharmaceuticals through preventive measures rather than only control (Daughton, 2003).

The multi-barrier approach has long been applied for safe drinking water strategies, with little or no emphasis on its potential for integrated PPCPs management through treated wastewater reuse, especially on land (CCME, 2004). Evidence shows that the coexistence of aerobic and anaerobic conditions in the natural systems would allow for the degradation of different kinds of PPCPs (Hijosa-Valsero *et al*., 2010; Ávila and García, 2015; Tatum, 2015). Degradation of PPCPs is driven by many processes including: photodegradation, biodegradation, and chemical degradation.

2.2.1.5.1 Photodegradation

Photodegradation of PPCPs is considered a major pathway for PPCPs degradation in the aqueous phase, but once adsorbed on soil particles, the latter becomes more important (Xing *et al*., 2011). Select pharmaceuticals (namely carbamazepine and gemfibrozil) were found to poorly adsorb to soil when experiments were conducted in the dark (Yu *et al*., 2013). Fedler *et al*. (2012) studied the effect of grass and photodegradation in the removal of the pharmaceuticals in land applied systems, using three types of soil columns: grass-covered, plastic-covered, and open columns. The study found that removal efficiencies were more than 95% for all compounds in all systems. Grassed columns were found to be 30% and 15% more capable of pharmaceuticals removal than covered and open columns (Fedler *et al*., 2012).

The degradation characteristics of 30 kinds of PPCPs commonly found in surface water were examined under UV treatment at 254 nm wavelength (Kim and Tanaka, 2009). When a UV dose of 230 mJ/cm^2 was introduced, photodegradation rates of about 3% (theophylline) to 100% (diclofenac) and about 15% (clarithromycin) to 100% (diclofenac) were observed. UV, as well as other advanced oxidation processes such as Fenton reaction and photo-catalysis, changed the polarity and functional groups of the target PPCPs (McMonagle, 2013; Papageorgiou *et al*., 2014). Therefore, these processes are deemed suitable for water reuse purposes that involve direct human contact (Hernández-Leal *et al*., 2011; Yang *et al*., 2017). Removal profiles of sixteen pharmaceutical compounds with seven advanced oxidation processes (AOPs) demonstrated four distinct patterns explained in Giri *et al*. (2010). UV-based AOPs efficiently removed clofibric acid, fenoprofen, ketoprofen, phenytoin, and triclocarban, but O_3-based AOPs were of no use (Giri *et al*., 2010).

In addition, oxygen atmosphere was found to enhance photodegradation of naproxen, gemfibrozil, and hydrophilic ibuprofen (Lin and Reinhard, 2005). The effect of combined

UV and hydrogen peroxide (H_2O_2) was studied (Pisarenko *et al*., 2012; De la Cruz *et al*., 2013; Yang *et al*., 2016). Vogna *et al*. (2004), Samarghandi *et al*. (2007), and Rosario-Ortiz *et al*. (2010) concluded that the addition of hydrogen peroxide in ozonation was generally of little use to the removal of pharmaceuticals from wastewater. Medium pressure UV lamps were found to be more efficient to maximize bench-scale degradation of the selected group of compounds (ketoprofen, naproxen, carbamazepine, ciprofloxacin, clofibric acid, and iohexol) than low pressure lamps by both UV photolysis and UV/H_2O_2 oxidation (Pereira *et al*., 2007). Photocatalysis was found to be a promising method for the oxidation of pharmaceutically active compounds such as carbamazepine and clofibric acid in water (Doll and Frimmel, 2005; Giri *et al*., 2010). In addition, dissolved organic matter in soil may act as a photosensitizer and promote photodegradation in the aqueous phase (Burrows *et al*., 2002; Andreozzi *et al*., 2003; Peña *et al*., 2011).

2.2.1.5.2 Biodegradation

Biodegradation plays a major role in the removal of PPCPs from solid matrix (Xuan *et al*., 2008; Robinson and Hellou, 2009; Majewsky *et al*., 2011). Comparative experiments conducted in sterilized soils showed that the sterilization treatment resulted in a decrease in the degradation rates of PPCPs (Yu *et al*., 2013). Microorganisms were found to reduce organic and inorganic substances in both aerobic and anaerobic conditions through enzymatic or metabolic processes (Joutey *et al*., 2013). Aerobic conditions are generally favoured for the removal of emerging contaminants (Ghattas *et al*., 2017). However, some pollutants, such as highly-halogenated aromatic compounds, have been shown to be more easily degradable under strictly anaerobic conditions (Vogel *et al*., 1987).

The biodegradation of pharmaceuticals under aerobic conditions in soil has been studied extensively (Jones *et al*., 2007; Walters *et al*., 2010; Piterina *et al*., 2012). Batch experiments and flow-through soil columns under unsaturated aerobic conditions demonstrated biodegradation for pharmaceuticals, such as ibuprofen, bezafibrate, and diclofenac (Tiehm *et al*., 2011). Long term batch experiments for a period of two and a half years on 23 pharmaceuticals and endocrine disrupting compounds were studied

(Schmidt *et al*., 2017). Fourteen out of the 23 compounds showed a degradation of more than 50% of their initial concentrations under aerobic conditions. Natural estrogens estriol, estrone and 17β-estradiol showed complete biodegradation under aerobic and nitrate-reducing conditions, with a temporary increase of estrone (Schmidt *et al*., 2017).

Mrozik and Stefańska (2014), however, found that certain antidiabetic pharmaceuticals, namely glimepiride, glibenclamide, gliclazide and metformin, were not quickly biodegradable under aerobic conditions even with half-life values of up to 120 days. Anaerobic conditions occur frequently in environmental matrices, such as soils, sediments and underground aquifers (Champ *et al*., 1979; Bethke *et al*., 2011). Certain aerobically-recalcitrant contaminants are biodegraded under strictly anaerobic conditions (Vogel *et al*., 1987; Ghattas *et al*., 2017). Highly-halogenated aromatic compounds are biodegraded by halorespiring bacteria that grow independently of inorganic electron acceptors (Holliger *et al*., 1998). Other examples of anaerobic reduction of pharmaceuticals include ciprofloxacin, norfloxacin (Golet *et al*., 2002), sulfadiazine (Li, 2014), trimethoprim, diphenhydramine, carbamazepine (Kinney *et al*., 2008), diazepam (Narumiya *et al*., 2013), and triclosan (Butler *et al*., 2011).

Five microorganisms were studied to determine their potential to biodegrade selected drugs, and to identify the metabolites arising from their biodegradation (Gauthier, 2008). The microorganism *Rhodococcus rhodochrous* showed an ability to degrade sulfamethoxazole, sulfamethizole, trimethoprim, and carbamazepine (Gauthier, 2008). Pharmaceuticals may have an impact on the diversity of microbial communities by changing their ability to metabolize different carbon sources (Gielen *et al*., 2011; Grenni *et al*., 2014).

However, Kelsic *et al*. (2015) showed, using analytical modelling, that opposing actions of antibiotic production and degradation may enable coexistence of microbial species even in well-mixed unstructured spatial environments. Interestingly, soil type, temperature, pH level, oxygen rate, and soil organic matter were all found to affect pathways, rates, and metabolites in the biotransformation process (Xu *et al*., 2009; Martín *et al*., 2012; Salgado *et al*., 2012; Zhang *et al*., 2017).

2.2.1.5.3 Physical barrier

PPCPs sorption and mobility has been related to the organic and mineral content of soils as well as properties of the contaminant such as solubility and log K_{ow} (Oppel *et al*., 2004; Williams and Adamsen 2006; Chefetz *et al*., 2008; Hofley, 2012). K_{ow} is the n-octanol-water partition coefficient that assesses chemicals partitioning in the environment. The adsorption of pharmaceuticals to soil/sediments could either promote or decease PPCPs degradation. Organic matter in soil could form complexes with PPCPs through PPCP active functional groups, and thus deactivate them (Xing *et al*., 2011). For example, dichlorophenol sorption correlated positively with particulate soil organic matter (Neumann *et al*., 2014). It was found to promote the degradation of PPCPs through oxidation processes such as ozonation and photo-oxidation (Lester *et al*., 2013; Yong and Lin, 2013).

Studies on the sorption of emerging contaminants to humic acids and dissolved organic matter have suggested that π-interactions with aromatic components may be an important sorption mechanism (Amiri *et al*., 2005; Hernandez-Ruiz *et al*., 2012; Sun *et al*., 2012; Jia *et al*., 2017).

However, photodegradation, biodegradation, and chemical degradation could be either enhanced or depressed after PPCPs sorption to soil and organic matter. For example, ciprofloxacin sorption to fine particulate organic matter was found to be rapid enough to limit the amount of the pharmaceutical available for photodegradation (Belden *et al*., 2007).

Foolad *et al*. (2016) conducted biodegradation studies of selected PPCPs by indigenous microbial community present in soil. The study found that the removal rates of acetaminophen, salicylic acid, and diethyltoluamide increased with time while no effect had been observed for other such as carbamazepine and crotamiton. The transformation of isoxaflutole to its herbicidally active diketonitrile degradate was significantly enhanced in the presence of soil and occurred more rapidly in systems containing soil with a greater soil pH (Rice *et al*., 2004).

2.2.1.5.4 Plant uptake

Plant uptake of pharmaceuticals generally occurs as a partitioning process from soil water to the plant root, and is most favorable for compounds of intermediate hydrophobicity (Wu *et al*., 2010). Carter *et al*. (2014) studied the fate and uptake of five pharmaceuticals (carbamazepine, diclofenac, fluoxetine, propranolol, sulfamethazine) and a personal care product (triclosan) in soil-plant systems using radish (*Raphanus sativus*) and ryegrass (*Lolium perenne*). The study demonstrated the ability of plant species to accumulate pharmaceuticals from soils with uptake apparently specific to both plant species and chemicals (Carter *et al*., 2014).

Although there is considerable variation between plant species, Briggs *et al*. (1982) and Topp *et al*. (1986) have shown that, for many PPCPs, uptake into plant roots from solution is inversely proportional to water solubility (or directly proportional to octanol/water partition coefficient, K_{ow}) but that transfer to shoots is more efficient for chemicals of intermediate solubility. The most readily transported chemicals are those of lower solubility, but the very soluble chemicals tend not to be appreciably sorbed. The net result is that the observed concentration of chemical in plant parts such as the stem is controlled by both kinetic and equilibrium factors with soluble chemicals being transported rapidly but partitioning weakly, and less soluble chemicals partitioning strongly thus their transport throughout the plant is delayed (Hellström, 2003). Plant uptake presents an additional barrier to PPCPs pathway to humans as opposed to the current shorter pathway through drinking water.

2.2.2 Reason II: Treated wastewater discharge to surface waters and eutrophication

The Great Lakes are threatened by anthropogenic, climate and biotic stresses, and it is increasingly difficult to manage them due to the complexity of interactions among different stressors (Cotner *et al*., 2017). Major stress factors are eutrophication, emergence of harmful algal blooms, and hypoxia (EDC, 2014; MECC, 2017; NCCOS, 2017). Studies of the Great Lakes have shown increased growth of *cyanobacteria*, known as blue-green algae, which is favored by low ratios of nitrogen to phosphorus (Schindler *et al*., 2016).

A key factor contributing to the growth of blue-green algae is the amount of available nutrients such as phosphorus and nitrogen (Horachek *et al*., 2015). A lake with extremely low concentrations of dissolved inorganic carbon but with increasing inputs of phosphorus and nitrogen would cause algal blooms, no matter how limiting carbon is (Schindler *et al*., 1972). Although algae are about 50% carbon, phosphorus is the limiting nutrient despite the fact that algal cells are less than 1% phosphorus (Wetzel, 1983).

2.2.2.1 Current treated wastewater management and eutrophication

Nutrient loading is caused by household products containing phosphorous compounds, such as detergents, as well as organic and chemical fertilizers used in agricultural practices, urbanization and human waste (Roelofs, 2015). Municipal wastewater treatment plants and industrial wastewater discharges are the major sources of nutrients, particularly phosphorus (WSI, 2009). Blue-green algal blooms can be also be caused by stormwater runoff as well as leaching from septic tank systems (MECC, 2017a). Treated wastewater is loaded with inorganic nitrogen and phosphorus, and its discharge to natural water bodies causes eutrophication and more long-term problems. In Europe, sewage effluents may well provide a greater risk of river eutrophication than diffuse sources from agricultural land, even in rural areas with high agricultural phosphorus losses (Jarvie *et al*., 2006).

Evidence of sewage-driven eutrophication and harmful algal blooms in Florida's Indian River Lagoon was reported in Lapointe *et al*. (2015). Qin *et al*. (2011) conducted water exchange experiments using treated wastewater in an artificial landscape pond. A eutrophication model was calibrated and applied to evaluate the effects of water exchange on algae growth in the pond. The study found the joint dilution process and nutrient supply process of water exchange initially cause algae level to rise and then rapidly decline as the hydraulic retention time decreases, with phosphorus input being the limiting factor (Qin *et al*., 2011).

For Lake Fure in Denmark, it was decided in the 1970s to expand the treatment of the wastewater to include nutrient removal (98% removal of phosphorus). By these measures, phosphorus loading was reduced from 33 to 2.5 tonnes per year, with the remaining phosphorus loading coming from stormwater overflow, treated wastewater, and diffuse sources (UNEP, 2001). However, the full effect of the measures taken have not been seen although several lake retention times (each of 20 years) have passed. This is because the internal loading, i.e. the loading from the sediment, is still about 12 tonnes per year, and further phosphorus reduction (at 99% removal) was recommended (UNEP, 2001).

Struijs *et al*. (2010) performed a life cycle impact assessment of inland water eutrophication in Europe. Normalization factors based on the emission of total phosphorus resulted in 60.1 disappeared fraction of species.m^3/person with a relative contribution of 16% by manure application, 18% by fertilizer application, and 66% by sewage treatment plant emissions (Struijs *et al*., 2010). This is also confirmed in the Baltic Marine Environment Protection Commission's 2004 report (Figure 2.4).

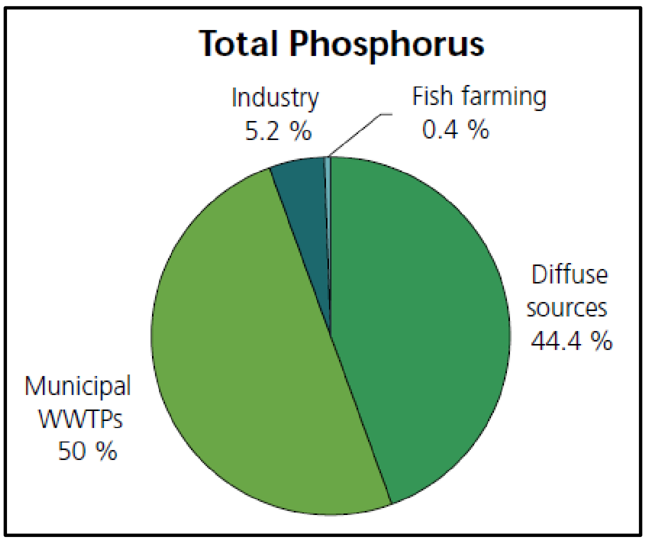

Figure 2.4 Proportion of the inputs of total phosphorus by source into surface waters within the catchment area of the Baltic Sea in 2000 (BMEPC, 2004).

Extensive wastewater treatment involving nutrient removal has been introduced for many lakes in northern Europe, but, as the results for Lake Fure show, a long time will elapse before the full effect of this treatment can be observed (UNEP, 2001). In many cases, non-point diffuse pollution will be needed to be reduced considerably, which is clearly a much more difficult task than reducing point source pollution.

The European Union passed a bill in 2011 to restrict phosphates in dishwasher detergents to less than 0.3 gram per hard water dosage starting from 2017 (Chemical Watch, 2011). A year earlier, sixteen U.S. states limited dishwasher detergent phosphate to less than 0.5 percent phosphorus by weight from 8 to 9 percent in traditional detergents (AP, 2010; Hochanadel, 2010). In 1970, eutrophication was identified by the Council on Environmental Quality and the Federal Water Quality Administration as perhaps the single most difficult water pollution in the U.S. Consequently, several municipalities in the U.S. enacted laws limiting the phosphorus content of detergent to 8.7% (ReVelle and ReVelle, 1988).

In 1989, Canada enacted the Concentration of Phosphorus in Certain Cleaning Products Regulations, and called for a reduction to 0.5% by weight expressed as elemental phosphorus (or 1.1% by weight expressed as phosphorus pentoxide) in dishwashing detergents, and a reduction to 2.2% as elemental phosphorus (or 5% by weight as phosphorus pentoxide) in laundry detergents (ECCC, 2017). More than 10 years later, the reduction in phytoplankton concentrations in the open water of Lakes Erie and Ontario were only one-third (Dolan and McGunagle, 2002).

Limiting phosphorus inputs to the Great Lakes was recognised as key to controlling excessive algal growth. However, the nutrient management approaches used in the 1970s are no longer adequate (Figure 2.5; Figure 2.6; Bunch, 2016).

Figure 2.5 Image from space showing visible coastal phytoplankton blooms in Lakes Erie, Michigan, Huron, and Ontario, stretching over thousands of kilometers (Earth Observatory, 2010).

Figure 2.6 An algal bloom outbreak in Sodus Bay of Lake Ontario in 2010 severely impacted the local recreational economy (Bunch, 2016).

Lake Washington, in Seattle, was the classic example of sewage diversion and demonstrated the potential impact of reducing phosphorus inputs to lakes. From the mid-

1960s, when diversion began, to the mid-1970s, total phosphorus levels in the lake declined by two-thirds, and Secchi disk clarity tripled (Schussler *et al*., 2007). To date, most efforts to reduce non-point source pollution are similar to those used to deal with point source pollution in that they remove the pollutant after it is produced by using end-of-pipe Best Management Practices (BMPs). These include septic systems, stormwater detention ponds, infiltration basins, constructed wetlands, and buffer strips. These practices, however, often overlook phosphorus accumulation in the watershed.

2.2.2.2 The impact of eutrophication

Lake Erie supplies drinking water to more than 11 million consumers, processes millions of gallons of treated wastewater, provides important species habitat, supports substantial industrial development, and generates more than $50 billion in annual income from tourism, recreational boating, shipping, fisheries, and other industries (Watson *et al*., 2016). The full scale of the effect of eutrophication has not been quantified yet, and studies continue to identify threats to ecosystem, human and animal health, and the economy.

2.2.2.2.1 Ecosystems Implications

Harmful algal blooms (HABs) and hypoxia, a direct result of algae decomposition which consumes dissolved oxygen and releases carbon dioxide, can cause morbidity and mortality of birds and marine mammals, kill fish or shellfish directly, cause loss of submerged vegetation, and affect aquaculture and biodiversity (Figure 2.7; Zingone and Enevoldsen, 2000; Matsuyama and Shumway, 2009; Grattan *et al*., 2016).

In 2011, a research group revealed that birds were victims of poisoning by domoic acid, a potent neurotoxin produced by planktonic diatom of the genus *Pseudo-nitzschia* and red algae *Chondria armata*, inducing severe seizures and killing of wildlife (Marić *et al*., 2011). As domoic acid binds to brain receptors in birds, symptoms such as confusion, disorientation, scratching, seizures, coma, and even death occur (Lefebvre *et al*., 2001). More recent studies confirmed their findings (Table 2.1; Lefebvre *et al*., 2012; Turner, 2014; Saeed *et al*., 2017).

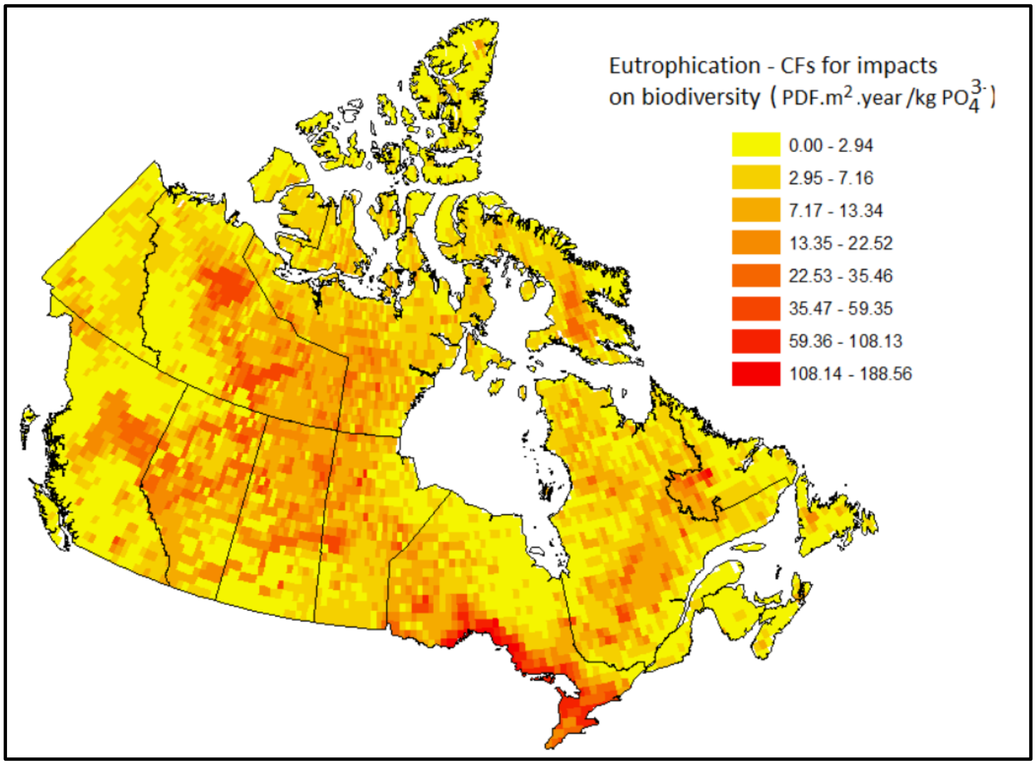

Figure 2.7 Freshwater eutrophication impact assessment on biodiversity (Helmes *et al.*, 2012).

Note: PDF.m².year is Potentially Disappeared Fraction per Square Meter of land per Year; CFs: Characterization Factors.

Table 2.1 Documented wild bird mortality caused by algal toxins (Friend and Franson, 2001).

Toxin	Algal species	Toxin type(s)	Migratory bird species affected	Route of exposure
Cyanobacterial	*Microcystis* sp. *Anabaena* sp. *Anabaena* sp. *Nodularia* sp. *Oscillatoria* sp.	Hepatotoxins (microcystins and nodularin) Neurotoxin (anatoxin-a and anatoxin-a(s))	Unidentified ducks, geese, and songbirds, Franklin's gull, American coot, mallard,	Oral (water)

			American wigeon	
Domoic acid	*Pseudonitzschia* sp.	Neurotoxin	Brown pelican, Brandt’s cormorant	Oral (food items)
Saxitoxin	*Alexandrium* sp.	Neurotoxin	Shag, northern fulmar, great cormorant, herring gull, common tern, common murre, Pacific loon, and sooty shearwater	Oral (food items)
Brevetoxin	*Gymnodinium* sp.	Neurotoxin	Lesser scaup	Oral (food items)

Planktivorous fish accumulate high levels of domoic acid during toxic *Pseudo-nitzschia* blooms (Lefebvre *et al*., 2002). Fish then serve as vectors of the toxin to seabirds and marine mammals causing mass mortality events in populations of these piscivorous predators (Sierra-Beltran *et al*., 1997; Lefebvre *et al*., 1999; Scholin *et al*., 2000). Shaw *et al*. (1997) studied the effects of dissolved domoic acid on the copepod *Tigriopus californicus* grazing on the non-toxic diatom *Thalassiosira pseudonana*. It was found to be toxic to the copepods, and caused mortality at half maximal inhibitory concentration (IC_{50}) as low as 8.62 µM (Shaw *et al*., 1997).

Cyanobacteria blocks light required for photosynthesis from reaching the bottom of lakes, affecting aquatic vegetation, and in turn, aquatic species that rely on the vegetation for food and nurseries (Anderson, 2005). Hypoxia is the most severe symptom of eutrophication and is becoming more prevalent (OECD, 2012). It was found to cause degradation of submerged aquatic vegetation beds and other valuable shallow water habitats (Lotze *et al*., 2006; Bricker *et al*., 2007).

Minimum dissolved oxygen requirements vary by species, but effects start to appear at concentrations lower than 3 mg/L (Figure 2.8; Ritter and Montagna, 1999; Rabalais *et al*., 2001; Small *et al*., 2014). In anaerobic conditions, *Clostridium botulinum* thrive and produce the toxin avian botulism, which was found in invasive Round Goby species, resulting in the death of thousands of sea and migratory birds along Lake Michigan shores (Corden, 2016).

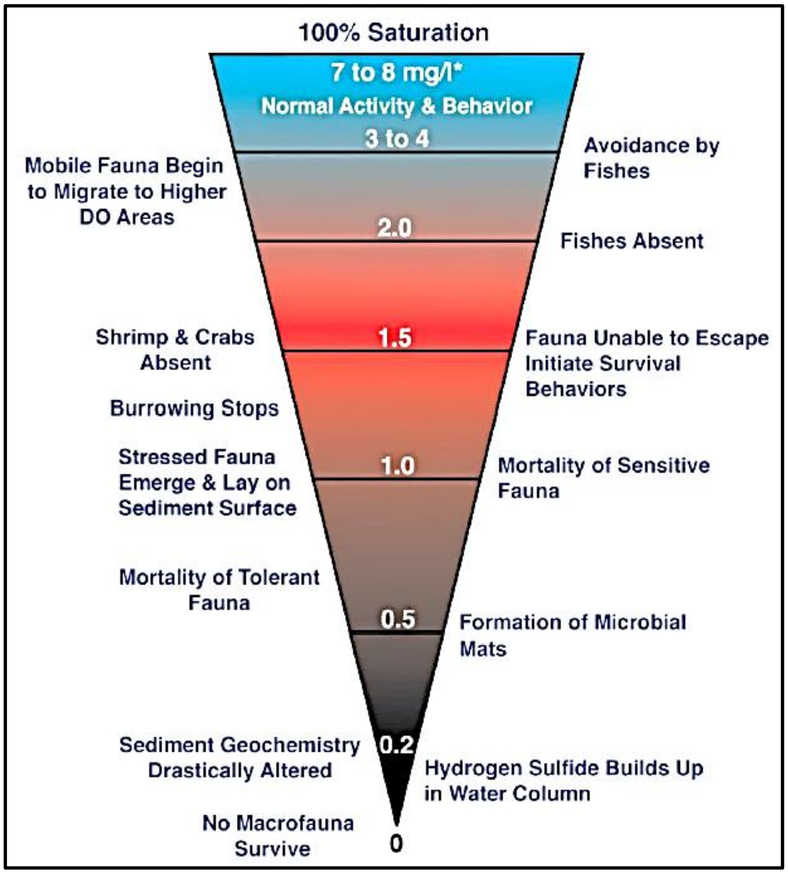

Figure 2.8 Response of fauna to declining oxygen concentration (Díaz and Rosenberg, 1995; Rabalais *et al*., 2001).

Eutrophication and associated hypoxia were linked to point industrial and municipal discharges to tidal rivers and freshwater bodies. The perceived effects of nutrient enrichment and hypoxia depend on the spatial scales of interest, which range from local, sometimes severely affected, water masses to entire fisheries ecosystems (Breitburg *et al*.,2009). Hypoxia was found to impact aquaculture communities' habitat and function (OECD, 2012).

2.2.2.2.2 Human and Animal Health Impact

The first domoic acid poisoning incident in Canada was reported in 1987 following consumption of contaminated blue mussels (*Mytilus edulis*) (Hynie *et al*., 1990). Since then, domoic acid producing algal blooms are accelerating frequently worldwide (Figure 2.9). Clinical examination of the affected people showed gastrointestinal, neurologic, and cardiovascular disorders and permanent short-term memory loss (Pulido, 2008).

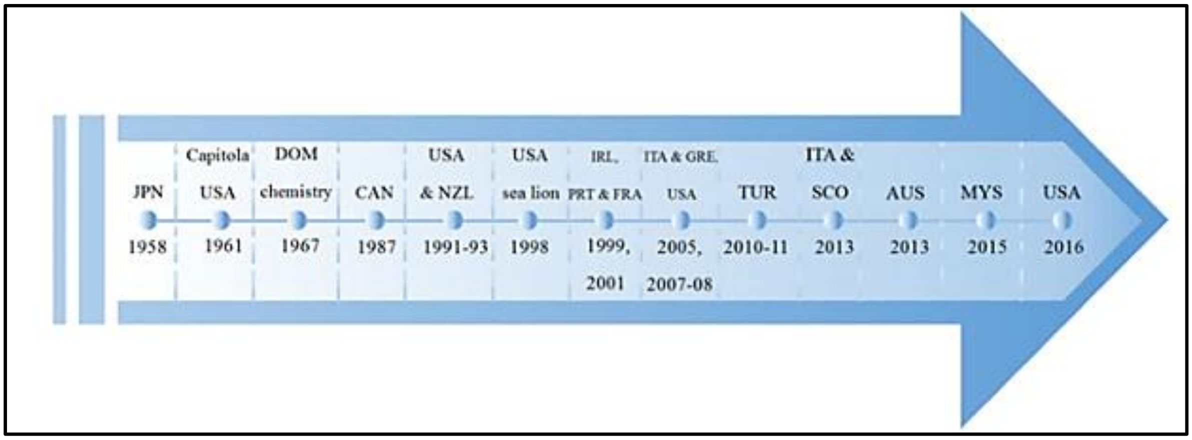

Figure 2.9 Domoic acid reported incidents since the 1987 Canada event (Saeed *et al*., 2017).

Note: JPN: Japan, USA: United States, CAN: Canada, NZL: New Zealand, IRL: Ireland, FRA: France, PRT: Portugal, TUR: Turkey, ITA: Italy, SCO: Scotland, AUS: Australia, and MYS: Malaysia.

Symptoms of harmful algal blooms include paralytic shellfish poisoning (PSP), ciguatera fish poisoning (CFP), diarrheal shellfish poisoning, neurotoxic shellfish poisoning (NSP), amnesic shellfish poisoning (ASP). ASP results in neuronal degeneration and hippocampal necrosis. The symptoms range from gastro intestinal disturbance to neurotoxic effects, such as hallucinations, tissue necrosis, memory loss, and physical perplexity.

PSP is caused by clams and mussels feeding on phytoplankton producing high levels of dinoflagellate. The dominant species of dinoflagellate associated with PSP in Canada are in the *Alexandrium* family, previously called *Gonyaulax* (FOC, 2015). Symptoms include numbness, tingling, loss of muscular coordination, terminating in paralysis, and inability to breathe. The U.S. National Oceanic and Atmospheric Administration has recorded algal blooms that produced significant toxin saxitoxin, which causes PSP among other 24 other toxins, that one or two small contaminated mussels could kill a healthy adult human (Powers, 2015).

CFP, caused by ciguatoxin, is the most common non-bacterial food toxin related to seafood ingestion in the United States, Canada, and more recently, Europe (Lanska, 2017). Although rarely lethal, the clinical symptoms are serious and may include more than 80 physiological disorders, and prevention is the primary clinical and technical objective (Erdner *et al*., 2008). NSP is caused by brevetoxins in which nerve cell paralysis results in gastroenteritis, muscle cramps, seizures, paralysis, and other neurological symptoms after the consumption of toxic shellfish (Van Dolah *et al*., 2001; Kirkpatrick *et al*., 2004)

Domoic acid poisoning was reported in sea lions, common dolphins, southern sea otters, and whales (Scholin *et al*., 2000; Lefbevre *et al*., 2002; Bejarano *et al*., 2008). It was first diagnosed in California sea lions (*Zalophus californianus*) in 1998 with mass stranding of animals on the U.S. west coast (Gulland, 2000). Studies on three harmful algal blooms between 1998 and 2000 reported over 100 adult female sea lions (*Zalophus californianus*) and one Northern fur seal (*Callorhinus ursinus*) intoxicated by domoic acid (Silvagni *et al*., 2005). The detection of domoic acid showed behavioral and neural disorders. Clinical signs were ataxia, head weaving, tumbling, coma, and seizures up to one week leading to death (Saeed *et al*., 2017).

The overall health status of dolphins is an indicator of the wellbeing of entire aquatic systems, and possible environmental trends (PBDP, 2016). Dolphins are a sentinel species for understanding future health threats to humans (Preston, 2016). In 2008, a mass mortality of bottlenose dolphins *Tursiops truncatus* and Florida manatees *Trichechus manatus latirostris* co-occurred with a severe bloom of the toxic algal species *Karenia brevis* along the eastern coast of Florida (Fire *et al*., 2015). Other damages can be short-term (including changing the color of the skin or shedding of dead skin), or long-term impacts (including skin cancer), and are well documented (Speekmann *et al*., 2006; Schaefer *et al*., 2009; Berdalet *et al*., 2015; Lapointe *et al*., 2015; Grattan *et al*., 2016).

2.2.2.2.3 Socio-economic Consequences

The existence of dense algal growth areas can inhibit or prevent access to waterways, affecting water sports (Knockaert, 2014). It reduces the transparency and navigation in the water which lessens the recreational values and opportunities of the lakes, especially for boating (Earth Eclipse, 2017). Anoxic bottom-water conditions produce hydrogen sulphide, which negatively impacts recreational use of the water (Knud-Hansen, 1994). Discoloration of water and the accumulation of dead fish brings losses to the fishing industry, reduction in the amusement and recreational experiences of visitors, and drops attendance in hotels and restaurants (Vojáček, 2010).

The economic losses in the recreational industry as a direct result of eutrophication of fresh water was estimated at $1.16 billion yearly in the U.S. (Dodds *et al*., 2009). In 2017, the International Institute for Sustainable Development (IISD) put forward a study on the cost of pollution in Canada (IISD, 2017). The losses in Lake Erie's value as a non-market (ecosystem) and market asset due to the algal blooms were estimated to be $3.8 billion and $4 billion, respectively. These figures relate to Lake Erie alone, and no basis exists to conclude losses in asset values for all degraded natural water bodies in Canada. Such losses are likely much larger (IISD, 2017).

Recurrent costs are found in public health, housing, infrastructure, and other sectors. Literature lack sufficient information on the cost of blue-green algae on health in Canada. Anderson *et al*. (2000) estimated public health costs at 45% of the total annual economic impact of harmful algal blooms in the United States. A tentative estimate of the cost of tap water-borne pathogens in 2015 is $895 million based on Canadian expenditures on bottled water and water filtration devices (IISD, 2017).

Excessive algal growth, especially in lakes serving as a source for drinking water, makes treatment (e.g. disinfection and filtration) more expensive, clogs intakes, increases corrosion of pipes, and can often cause odour and taste problems (Vollenweider, 1968; ReVelle and ReVelle, 1988). Environment and Climate Change Canada assumed that a decline in water quality due to algal blooms would lead to up to 6 per cent decrease in the value of all residential property within 1 kilometre of the Canadian shoreline of Lake Erie, depending on the severity of the bloom (MAEE, 2015). This amounts to $712 million, compared to $242 million loss in housing value at the U.S. shoreline of the lake (Bingham *et al*., 2015).

The amount of welfare loss is expressed in people's willingness to pay (e.g. in dollar/household year) to reduce eutrophication and its impacts (Leo De Nocker and Wattage, 2014). It indicates the extent to which people have preference for a situation without or with less eutrophication and related impacts, and to which extent they are prepared to financially contribute to guarantee this reduction. The average willingness to pay varies considerably across countries.

Ahtiainen *et al*. (2012) studied people in nine countries, and the average contribution ranged between 4 and 110 euros per person per year, totalling Euro 4 billion yearly. The Canadian willingness to pay index to control eutrophication has not been determined. However, Barbier *et al*. (2016) studied how this index for eutrophication varies with income, and found that income elasticity was 0.1 to 0.2 for low-income respondents, and 0.6 to 0.7 for high-income respondents.

2.3 Recovering nutrients for reuse in Canada

For decades, wastewater treatment has been practiced with little attention paid to recovering nutrients and or energy. Resource recovery has gained momentum in recent years, especially nutrients essential for plant growth such as nitrogen and phosphorus. However, excess application of macro- or micro-nutrients could lead to adverse effects on crop quantity as well as human and plant health.

Irrigation, on the other hand, is the world's largest consumer of blue water (70 to 80 per cent of global water consumption). In 2010, 838 million m^3 of fresh water were used for irrigation (Statistics Canada, 2013). It has been estimated that all agricultural sectors in Ontario used a total of 296 million m^3 of fresh water per year (de Loë *et al*., 2001). Sectors included livestock, field crop, fruit crop, vegetable crop, greenhouse/sod/nursery, golf courses and aquaculture. When golf courses and aquaculture estimates are removed, the total agricultural water use is 168 million m^3 of water per year.

Only 9.5% of the farms (115 out of 1,200) in the Province of Ontario use off-farm water sources to irrigate crops ranging from tap water to treated wastewater. It is commonly believed that Canada has an abundance of water (Sprague, 2007). In reality, most of this water is located in regions where agriculture does not take place (Kreutzwiser and de Loë, 2010). Many important agricultural regions in Canada, including parts of the Prairies and portions of British Columbia, are already water-stressed, and concerns about water quality exist throughout most of Canada's agricultural lands (AAFC, 2007; Stewart *et al*., 2011).

In Ontario, the combined capacity of more than 450 municipal sewage treatment plants is 6.7 million cubic metres per day (totalling 2,445 million m^3 per year) (ECO, 2003). Almost all of the treated effluents are being discharged to water courses, including Lake Ontario, Ottawa River, Thames River, the Grand River, and other water ways connected to the Great Lakes. Throughout the Great Lakes, wastewater treatment plants whose effluents often violate provincial water quality guidelines have rendered hundreds of miles of near-shore areas uninhabitable for aquatic life. WWTPs have been major contributors to the environmental degradation experienced in the 43 Great Lakes sites deemed "Areas of Concern" by the International Joint Commission (Kapitain, 1995).

However, balanced nutrition is an important aspect (Laegreid *et al*., 1999). Shifting from current practices to more sustainable wastewater-agriculture management could prove particularly beneficial to the environment, health and economy. Currently in Canada over 150 billion litres of untreated and undertreated wastewater (sewage) are dumped into waterways every year. Using wastewater for agriculture can reduce the burden on the near-capacity wastewater treatment plants. The Food and Agriculture (FAO) guidelines on treated effluent reuse in agriculture are less stringent than guidelines for discharge into water courses. This is of particular interest since more than 24% of the wastewater treatment plants do not meet health and environmental guidelines for discharge in Ontario (Kapitain, 1995). Up to date information is not available in the literature.

It has been estimated that Ontario municipal sewage treatment plants release 18 tonnes of organic compounds, and 1,100 tonnes of heavy metals into Ontario waterways each year, principally because of industrial releases to municipal sewage systems. Industrial discharges to sewers remain unregulated by the province. Source-treatment of industrial wastewater could free up effluents of toxic heavy metals, pharmaceuticals and endocrine disruptors.

2.3.1 Chemical fertilizers *vs*. organic fertilizers

The well-being of plants require soil to be both well-structured and nutrients-rich. While chemical fertilizers make nutrients available to plants immediately, they fail to sustain the soil structure. In fact, chemical fertilizers do not replace many trace elements that are gradually depleted by repeated crop plantings, resulting in long-term damage to the soil. Because the nutrients are readily available, there is a danger of over fertilization. This not only can kill plants but upset the entire ecosystem (Table 2.2). One third of fertilizers consumed by Canadian farmers is imported (AAFC, 2015).

Table 2.2 Constraints imposed by excess nutrients application levels.

Excess nutrient	*Agronomic consequences*
Nitrogen	Delay in ripening
	Higher sensitivity to disease
	Tendency to lodging of grains
	Burning of seedlings
	Leaching losses and groundwater pollution
Phosphorous	Poor rate of growth
Potassium	Fixation and adsorbing complexity
	Increases of soil salinity level
	Unbalance with magnesium
Magnesium	Toxicity

Particularly important, chemical fertilizers have the following long term-effects: (1) Repeated applications may result in a toxic buildup of chemicals such as arsenic, cadmium, and uranium in the soil. These toxic chemicals can eventually make their way into cultivated crops, and (2) Long-term use of chemical fertilizer can change the soil pH, upset beneficial microbial ecosystems, increase pests, and even contribute to the release of greenhouse gases (Massri and Labban, 2014; Han *et al*., 2016).

The price of chemical fertilizers is largely dependent on fluctuating petroleum product prices, leaving farmers vulnerable to unstable supply market. The source-pathway-recipient pollution control implies that direct discharge of treated wastewater to water courses is a shorter path to humans than application on land. In addition to reduced levels of contaminants through adsorption to soil particles, agricultural application of treated effluent would allow for a diverse microbial community to further degrade these contaminants, and eventually reduce exposure to humans.

2.3.2 Suitability of treated wastewater reuse

2.3.2.1 Financial affordability

Primary production and profitability in the agricultural industry are highly dependent upon fuels and fertilizers. Fuel and fertilizer costs accounted for 18% of total Canadian farm expenses, or $7.7 billion in 2014. For every one cent per litre increase in fuel prices, Canadian farmers' annual machinery fuel bill goes up by approximately $27 million. For fertilizer, every one cent per kilogram increase in price adds about $61 million to Canadian farmers' annual fertilizer bill (AAFC, 2012).

Fertilizer prices in Canada rose steadily starting in 2003, but increased sharply to reach a historical high in 2008. These increases abruptly halted in 2009 as a result of falling commodity prices, restricted availability of credit, and a sudden fall in energy prices. However, fertilizer prices resumed their climb in 2011 and continued to increase in 2012 in response to high energy prices and strong worldwide fertilizer demand driven by rising crop prices.

When the monetary value of all the benefits associated with a water-reuse project is calculated, traditional engineering cost-benefit analysis can serve to compare the project to its alternatives. When all the benefits are assigned to a single agency, such analysis can also be used to determine the economic feasibility of the project. Unfortunately, neither of these situations normally prevails. In the first place, the benefits of water reuse, in agriculture for example, include watershed protection, local economic development, improvement of public health, energy conservation, environmental protection, and other factors, which are not readily quantified by traditional cost-benefit techniques. Second, the benefits are usually distributed among a number of agencies, and the general public is not readily assigned (Lazarova and Bahri, 2004).

Life Cycle Assessment (LCA) is a way of addressing environmental issues and opportunities from a system or holistic perspective and evaluating a product or service system with the goal of reducing potential environmental impacts over the entire life cycle (Blumenfeld *et al*., 2003). LCA provides an adequate instrument for environmental decision support, while assessing the resource cost and environmental implications of different patterns of human behaviour. However, LCA requires significant sets of data and quantified measurements that may limit its use, as life cycle inventories of the water industry is not normally directly accessible and or properly compiled.

2.3.2.2 Social acceptability

Sustainable implementation of a wastewater treatment and reuse project must address technical as well as social, cultural and economic factors. Cultures are rarely homogeneous and frequently shaped by a complex variety of subcultures with widely different orientations. Important socio-cultural factors may affect the feasibility and acceptability schemes for treated wastewater reuse. Sustainable water management in the cultural context is greatly affected by religion which can act as a promoter or inhibitor of new ideas (Kley and Reijerkerk, 2009).

High levels of social and political acceptability provide greater legitimacy of an adaptation measure and influence the ease of implementation. It is important that a response to treated wastewater reuse be consistent with the social, economic and environmental goals and objectives of a community.

Market feasibility may refer to the ability to sell treated wastewater to food producers, or it can refer to the marketability of products grown with wastewater. In 2006, the Organic Agriculture Centre of Canada (OACC) oversaw a study on sales of certified organic products through the traditional mainstream supermarkets. It found that total sales of certified organic food had grown 28% overall from 2005 to 2006, with sales of pre-packaged certified organic goods up 31% while fresh products were up 22% (Statistics Canada, 2009). On the other hand, farmers' perception of using treated wastewater has not been investigated in Canada or Ontario. However, one study indicates that only 1.3% of Canadian farmers are organic producers, farming just over 390,000 hectares (Forge, 2004).

The main reasons farmers gave for their willingness to farm organically are their concerns for the environment and about working with agricultural chemicals in conventional farming systems. There is also an issue with the amount of energy used in agriculture, since many farm chemicals require energy intensive manufacturing processes that rely heavily on fossil fuels. Organic farmers find their method of farming to be profitable and rewarding (OMAFRA, 2009).

Environmental damage frequently occurs due to two main types of market failure:

- Financial damage costs that are not transparent or paid directly, i.e. picked up by society; and
- Environment being treated as a free good with no price attached.

Externality is defined as the costs and benefits which arise when the social or economic activities of one group of people have an impact on another, and when the first group fails to fully account for their impact (European Commission, 2016). In theory, economic instruments can be used to internalise environmental externalities by attaching a price to using the "environment". Practically, it is very difficult to place monetary value on the environmental costs, but such instruments can still be used in preference to command and control to meet agreed goals in a flexible and effective manner.

Feed-in tariff schemes have been traditionally used in the energy sector for replacing fossil fuel sources with renewable or low-carbon sources, such as anaerobic digesters, solar panels, combined heat and power… etc. A best practice would be to reflect on the indirect energy savings made by both reducing the use of chemical fertilizers and avoiding discharge into surface water, or place a similar subsidy programme along with other technical cost breakthroughs which would generate and sustain installations.

2.3.2.3 Legal framework and institutional challenges

The 2016 – 2019 Federal Sustainable Development Strategy set two medium term targets in relation to the strategic pristine lakes and rivers goal (ECCC, 2016a):

1. Reduce phosphorus loading into Lake Erie by 40% to achieve the binational (Canada-US) phosphorus targets from a 2008 baseline, by 2025.
2. Reduce additional estimated 2000 kilograms of phosphorus per year to Lake Simcoe in support of Ontario's target to reduce phosphorus inputs into Lake Simcoe to 44,000 kilograms of phosphorus per year by 2045.

Efficient phosphorus removal can only be achieved in tertiary treatment (such as biological nutrient removal units). However, less than 9% of Canada's wastewater treatment plants were equipped with tertiary treatment systems as of 2009, the most recent year for which data are available (Figure 2.10). Achieving these targets through investments in advanced wastewater treatment alone may not be feasible. In addition, not all sewage wastes are treated in such facilities. According to Environment and Climate Change Canada (2016b), 3 per cent of Canadian homes connected to municipal sewer systems in 2009 saw their wastes sent directly into the environment without treatment. Another 16 per cent received only primary treatment (which does not remove phosphorus) before release, and a further 13 per cent of households managed their own sewage using private septic systems, where the quality of treatment is difficult to judge (IISD, 2017). Diversion of treated wastewater from natural surface water bodies (including rivers and lakes) should be considered.

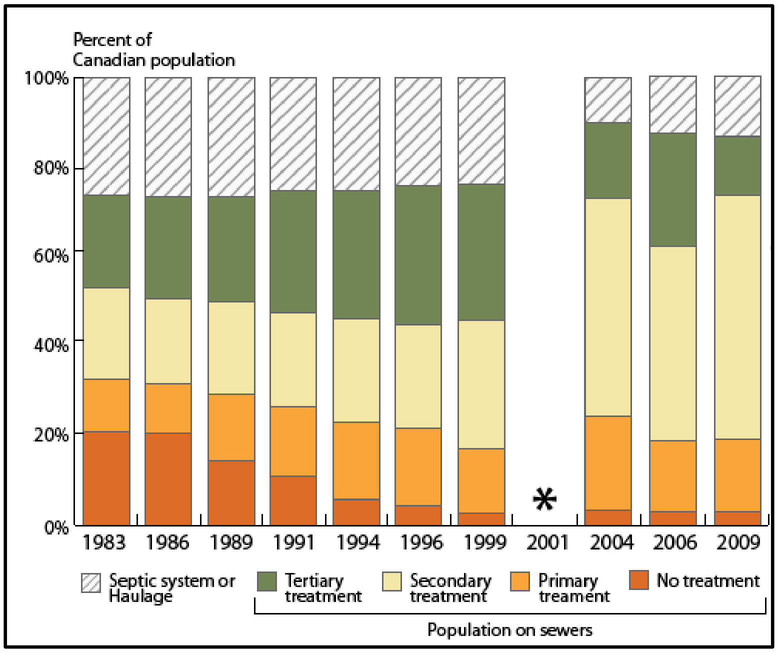

Figure 2.10 Municipal wastewater treatment indicator (ECCC, 2016b).

The success of water re-use projects however, does not just depend on the effectiveness and suitability of the technology, but also on the presence of an institutional framework that ensures that the treated water can be distributed and used safely and efficiently (Lawrence *et al*., 2003). Differences in legal regimes, institutional settings, and socio-economic contexts across the country mean that there is no single framework that will be effective in all jurisdictions (CCA, 2013). There are principles and promising practices that have been shown to be effective in supporting sustainable management of water resources. These include (CCA, 2013):

- Ensuring governance operates at the appropriate scale, which can help facilitate coordination of management efforts across relevant jurisdictions and stakeholders;
- Integrating land-use planning with water management decisions, which can assist in incorporating the needs of multiple users, while ensuring sustainable water management in the long run; and
- Incorporating knowledge into the decision-making process (including scientific, traditional, and local knowledge), which can lead to more robust solutions that account for the complex and interconnected nature of current water management and governance challenges. Transdisciplinary research, where researchers and

partners from the farm community, industry, and government jointly define problems and research programs, is an important way to facilitate knowledge co-production.

It is also urgent that Canada acts quickly on policies, programs, and infrastructure towards a sustainable framework for water management in agriculture. Effective water governance is critical to the success of agriculture, and can take the form of a producer-responsibility policy (for the industry) and product stewardship programme (for provincial and municipal governments). For example, unclear rules regarding who can use water and inefficient decision-making systems create risks and uncertainties for farmers (CCA, 2013).

Reuse and zero discharge are the ultimate forms of prevention of point-source pollution of surface water. Combined with stricter regulations on non-point-source pollution, such as watershed runoff, this should maintain drinkable, swimmable, fishable, and optimum recreational conditions in our rivers and streams.

2.4 Conclusions

1. Treated wastewater discharge to natural surface water bodies contributes significantly to environmental phenomena such as the emergence of PPCPs and eutrophication. The exact extent to which treated effluent contributes to such phenomena may be difficult to quantify.
2. PPCPs have adverse effect on the ecological systems. Aquatic organisms could be exposed to such contaminants throughout their entire lifetimes even at low concentrations, potentially leading to feminization, and altered behaviour, which may affect their survival.
3. Treated effluents, containing PPCPs, contribute to the development of anti-microbial resistant bacteria and antibiotic-resistant genes, which were found as far as 20 km downstream of point discharges.
4. There is no significant epidemiological evidence on the impact of PPCPs on humans. However, PPCPs and endocrine disruptors could inhibit the action of hormones, or alter the normal regulatory functions of the endocrine system.

5. The running costs of addressing PPCPs through only advanced wastewater treatment plants may not be affordable.
6. For PPCPs, the application of treated wastewater on land could present a new multi-barrier approach, and could likely allow natural processes (such as biodegradation, photodegradation, physical adsorption to soil and sediments, and plant uptake) to more integrally alleviate the issue. However, the combined effect of these processes requires further investigation.
7. Sewage effluents may well provide a greater risk of river eutrophication than diffuse sources from agricultural land.
8. Phosphorus is the limiting factor for the growth of algae in lakes.
9. Hypoxia and harmful algal blooms can cause morbidity and mortality of birds and marine mammals, kill fish or shellfish directly, cause loss of submerged vegetation, and affect aquaculture and biodiversity.
10. Harmful algal blooms poisoning, including paralytic shellfish poisoning, ciguatera fish poisoning, diarrheal shellfish poisoning, neurotoxic shellfish poisoning, amnesic shellfish poisoning, has been reported in humans and animals.
11. The socio-economic consequences of sewage-driven eutrophication can be difficult to quantify. However, the cost of eutrophication on drinking water treatment infrastructure, fishing and recreational industries, and public health are likely prohibitive.
12. The willingness to pay for eutrophication generally exists. However, further studies are needed to determine how much the public in Canada is willing to pay.
13. The source-pathway-recipient pollution control implies that direct discharge of treated wastewater to water courses is a shorter path to human than application on land.
14. Recent federal strategies have set ambitious targets. Measures, such as producer-responsibility and feed-in tariffs, could not only contribute to achieving these targets, but also promote resource recovery.

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Chapter 3

Local Level Governance and Management of Water Reuse in Agriculture: A case study from Ghana

3.1 Introduction

Unsustainable development pathways and governance failures have generated immense pressures on water resources, affecting their quality and availability and, in turn, compromising the ability of these resources to generate social and economic benefits (UNWWD, 2015). The United Nations World Water Development report (2015) found that Ghana is currently water vulnerable, and at current trends, it will become a water stressed country by 2025 (Figure 3.1). Agriculture is the most water-consuming sector worldwide, especially with intensive agricultural activities practiced in rural areas of regions such as sub-Saharan Africa (Clay, 2013; EC, 2015). Treated wastewater provides reliable irrigation water supplies and, as a farming practice, contributes to both the improvement of urban food supply and the livelihood of many farmers and produce traders.

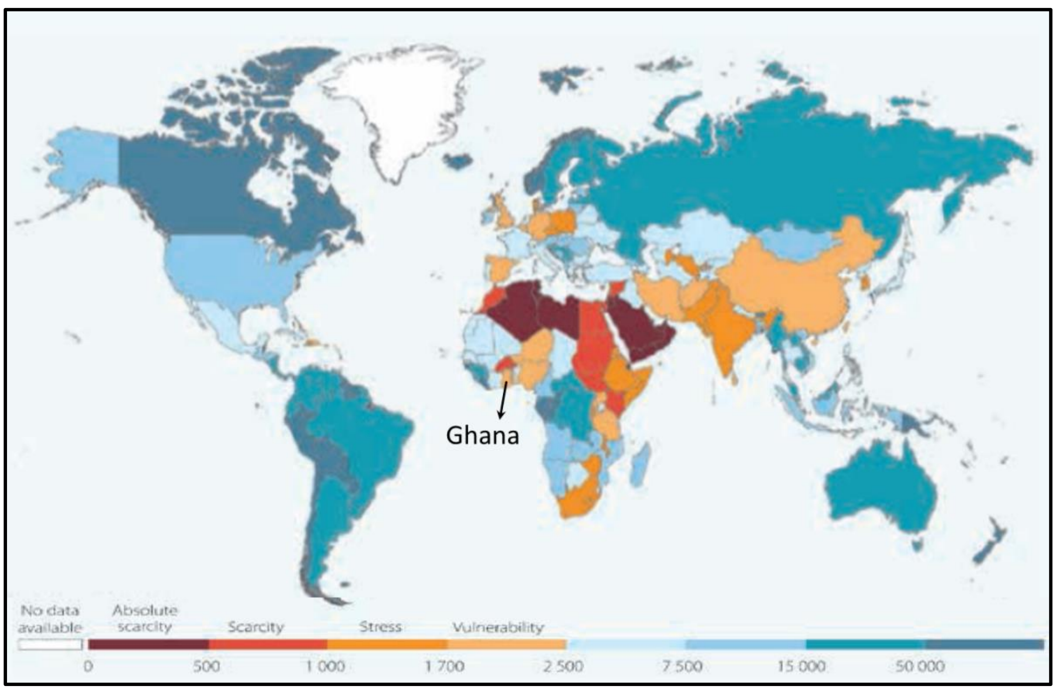

Figure 3.1 Total renewable water resources per capita in 2013 (Adopted from UNWWD, 2015).

The agricultural sector of Ghana accounts for about 65 per cent of the work force, about 40 per cent of the gross domestic product, and about 40 per cent of foreign currencies acquired through exports (Namara *et al*., 2011). Wastewater collection, treatment, and disposal or distribution is a persistent problem. Even if only 30 per cent of wastewater is treated and reused for agriculture, it could irrigate up to 13,800 hectares (i.e. 31 per cent of the total formally- and informally-irrigated agricultural land), while offsetting the use of fresh water sources and providing livelihood support for more than 9,000 farmers in peri-urban areas (Agodzo *et al*., 2003; Amoah, 2008; MFA, 2011; Namara *et al*., 2011).

Urban farming can make substantial contributions beyond the provision of livelihoods and food. These include contributions to buffer zone management and flood control, thus supporting climate change adaptation strategies, land reclamation, land protection, resource recovery (from waste), urban greening, and biodiversity conservation.

3.1.1 Aim and objectives

The aim of the study is to examine the feasibility of using treated wastewater in agriculture taking Ghana as a case study of a typical developing country. The objectives are to:

- Provide an updated and informative review of treated wastewater status and current characteristics of wastewater treatment plants in Ghana;
- Understand current practices with regard to treated wastewater reuse in agriculture;
- Review the latest toxicity requirements for treated effluent reuse in agriculture, and how this is reflected on cultivated crops in Ghana;
- Examine the role of waste stabilization ponds in protecting the environment and alleviating health risks;
- Outline the potential economic benefits and institutional challenges related to applying treated wastewater to agricultural land.
- Develop a hierarchy of the main challenges to treated wastewater reuse in agriculture;

- Develop better understanding of the scale of various challenges using the analytical hierarchy process as a decision-making tool;
- Analyze treated wastewater toxicity against international guidelines for application on land; and
- Provide informative evidence on the toxicological suitability of using treated wastewater for agriculture.

3.2 Potential of fertigation for nutrients recovery

Fertigation is the means by which valuable plant nutrients are transported into land via treated wastewater irrigation. This saves up to 40 per cent of needed fertilizers, while reducing significantly the risk of fertilizers runoff and leaching (Harrison, 2009; Hines *et al.*, 2010; Kafkafi and Tarchitzky, 2011; Agropedia, 2014). Other opportunities of fertigation are that it facilitates convenient disposal of waste products, and improves crop yield and quality due to additional macro- and micro-nutrients, as well as organic matter, added to the soil via treated effluent (Hussain *et al*., 2002).

However, wastewater, especially when used in irrigation without adequate treatment, has constituents that can cause soil and crop contamination, hence posing public health and environmental risks (Dion, 2010). This is commonly seen in most developing countries which lack resources for effective wastewater treatment facilities; and so, large volumes of wastewater generated, especially in urban areas, remain untreated.

3.2.1 Wastewater treatment plants characterization

In 2013, the International Water Management Institute (IWMI) conducted a survey, which excluded some decommissioned facilities, and counted 66 wastewater treatment plants and 11 faecal sludge treatment plants in Ghana (Amerasinghe *et al*., 2013). Of the 66 plants, only three could be considered centralized wastewater treatment facilities, while the remaining vast majority served as decentralized sewer systems.

Only one faecal sludge and 11 wastewater treatment plants, representing about 16 per cent of all treatment plants, were found to be fully functional, while the majority of the plants, more than 57 per cent, were found to be non-functional (Figure 3.2) (Drechsel and

Keraita, 2014). Most of these functional and semi-functional plants were small capacity plants owned by hotels and private companies (Murray and Drechsel, 2011). Innovative changes are therefore necessary for conventional wastewater treatment to function as a resource-recovery option in developing countries. In recent years, some of these changes have included research towards re-engineering conventional wastewater treatment systems to make them more appropriate for irrigation, by optimizing the water and nutrient contents in treated wastewater effluents (Drechsel and Keraita, 2014).

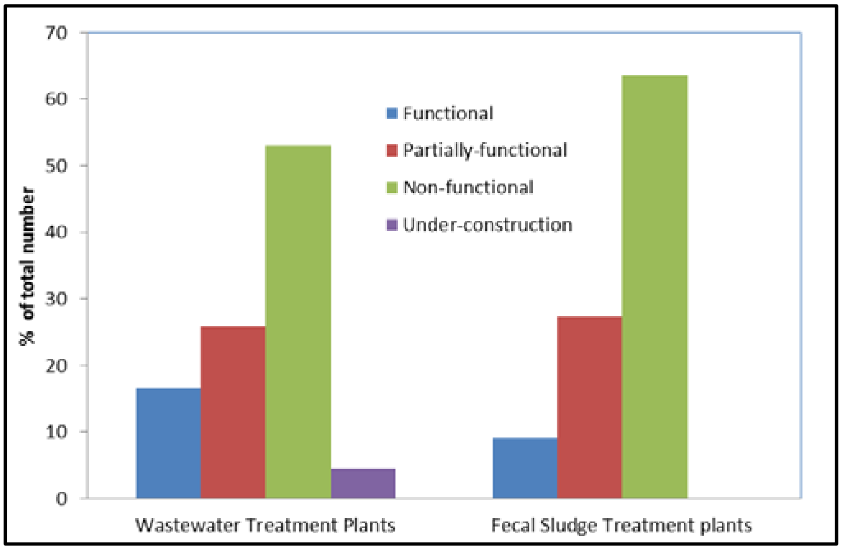

Figure 3.2 Status of Ghana wastewater and faecal treatment plants (Drechsel and Keraita, 2014).

More than 280 million cubic meters of wastewater were generated from urban Ghana in 2008 (Amoah, 2008). Wastewater treatment is very limited to less than 8 per cent of the mostly-domestic wastewater currently generated (UNWAIS, 2011). Given that the situation is worse in urban areas due to high population density, there is a strong link between the lack of wastewater treatment and the use of polluted water (sewage streams, surface water and even contaminated underground water) in irrigated urban agriculture. Some of the common routes by which wastewater arrives at farms include (FAO, 2012):

i. Wastewater ➤ Stream ➤ Vegetable farm;
ii. Wastewater ➤ Drain/gutter ➤ Farm pond ➤ Vegetable farm;
iii. Wastewater ➤ Stream ➤ Farm pond ➤ Vegetable farm;
iv. Wastewater ➤ Shallow well ➤ Vegetable farm;

v. Wastewater ➤ Wastewater treatment plant ➤ Vegetable farm.

The way in which wastewater arrives at farms varies depending on the season, the availability of other sources of water, and the location of the farm (FAO, 2012). Scenarios *i–iii* are common in drier cities like Accra and Tamale, while *iv* and *v* are common in wetter cities, such as Kumasi. A multiple-barrier approach and safe reuse practices can be developed through action research involving a number of stakeholders at different levels along the food chain (Figure 3.3).

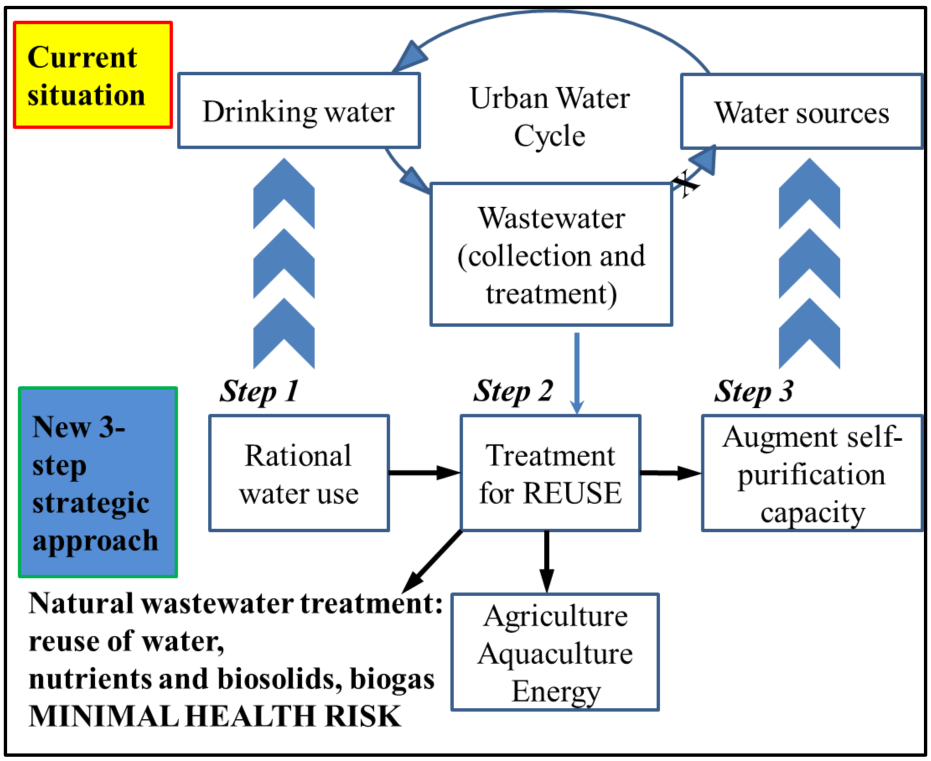

Figure 3.3 A new paradigm for urban and peri-urban water and wastewater management in developing countries.

3.2.2 Waste stabilization ponds in, and for, Ghana

Energy deficiency and very low efficiency of conventional wastewater treatment plants in developing countries triggered interest in natural treatment systems (Laary, 2015). In 2012, more than 85 per cent of Ghana's population did not have access to adequate sanitation (UNWWD, 2015). Water carriage sewerage systems are not always the most appropriate sanitation solution for the disposal of liquid domestic wastes in developing countries (Arthur, 1983). However, where water carriage systems are proposed, the first treatment option that should always be considered is the use of Waste Stabilization Ponds (WSPs) (Arthur, 1983; Crites *et al*., 2015). WSPs are natural wastewater treatment systems that require no electro-mechanical input and are simple to operate and maintain (Gloyna, 1971; Drechsel *et al*., 2009). They are recognized for their (Mara, 2004):

- Simplicity: WSPs are constructed by basic earth moving and minimal civil works.
- Low cost: Compared to other technologies such as aerated lagoons, oxidation ditches, and conventional systems, WSPs are the cheapest. This is also dependent on land price and the Opportunity Cost of Capital (OCC).
- High efficiency: WSPs are very effective in removing pathogens: up to 6 $\log_{10}$ unit reductions of excreted bacteria, up to 4 $\log_{10}$ unit reductions of excreted viruses, 100 per cent removal of helminth eggs, and more than 90 per cent of protozoan cysts.
- Robustness: WSPs are also resistant to heavy metals and shock hydraulic and organic loads.

Given the efficiency of WSPs in removing pathogens, effluents have been successfully used for both restricted and unrestricted irrigation in many developing countries in the Middle East and Africa, especially in those with hot sunny climates (Section 3.3) (Scott *et al*., 2004). WSPs are the most extensively used systems in Ghana, with almost all faecal sludge and large-capacity sewage treatment plants using the system (Amoatey and Bani, 2011). Recorded trickling filters and activated sludge plants are of low capacity and belong to private enterprises such as larger hotels. Less than a quarter of all treatment plants are operational (Section 3.2.1), and therefore, that calls WSPs' performance in Ghana into question.

Appropriate pond treatment for faecal sludge and wastewater requires the development of specific design and operational guidelines. Simply using WSP design criteria for faecal-sludge-only ponds will lead to uneconomical designs and inadequate plant performance. Other related aspects could be:

- Lack of appreciation by many designers to the complexity of physical, biological and chemical processes within WSPs;
- Lack of consistency in design, construction and operation aimed at optimal performance;
- Lack of appropriate design tools and methodologies suitable for local conditions; and
- Changing nature of the rapidly-developing technology.

It is important to implement and operate WSPs for a range of applications and design objectives, while following a certain set of standards that can be adopted nationally to accommodate technology development and changes in local conditions and information concepts and ideas with time.

3.3 Economic benefits

A traditional cost-benefit analysis for current treated-wastewater discharge schemes often leads to many important costs being overlooked as the setting of the system boundaries is difficult. An example of the magnitude of such costs can be seen by considering the discharge of wastewater to the sea. In addition to the capital and recurrent costs, important consideration should be given to “downstream” costs including drinking-water treatment, degradation of the coastal environment, damage to fishing industries, pollution of recreational water, and lost tourism revenues. Each one of these external costs may in turn incur further costs.

Ghana is the fourth largest fertilizer-consuming country in West Africa region, with the bulk of it being imported (Fuentes *et al*., 2012). Farmers are becoming more vulnerable to fluctuating international markets, and this explains the higher prices farmers have paid in recent years for fertilizers. In 2009, some 218,000 metric tons were consumed, and the average price of all used fertilizers was US 31.5 dollars for a 50-kg bag (IFPRI, 2011). Treated wastewater (excluding treated sludge) contains 20 to 40 per cent fertilizer value, so its reuse has the potential to save US 27.5 to 54.9 million dollars each year.

The rate of application of fertilizers is currently at 7.3 kilograms per hectare, most of which is a variation of chemical constituents. This is considered low in light of a strong subsidy program implemented by the Government of Ghana (Fuentes *et al*., 2012). This may be, at least in part, attributed to a loss of trust in the ability of chemical fertilizers to maintain soil fertility, among other reasons. Treated wastewater and produced sludge are natural soil amendments that contain more than just nitrogen, phosphorus and potassium. Soils in Ghana are generally deficient in major nutrients and organic matter (UNEP, 2011). Adding organic matter to mineral soils can improve both their physical properties (infiltration, water holding capacity, structure, etc.) and their chemical properties (fertility, cation exchange capacity, etc.).

Through agricultural utilization of treated wastewater, producers can benefit, and possibly derive marketing potential, from materials that would otherwise be dumped into landfills or present environmental pollution problems. Increased crop yield has been attributed to treated wastewater, and, at a large scale, this could contribute to food security as well as increased public health. For example, a yield increase of 15 tons of cabbage per hectare with drip irrigation, combined with savings of US 216 dollars per hectare per year in water pumping and fertilizer costs when using organic sludge rather than chemical nitrogen fertilizer, provide a much-needed additional income to Ghana's peri-urban farmers and enable a sustainable production of fresh vegetables for the urban population even during the dry season (FAO, 2013). In addition, better water management practices have many other socio-economic benefits. Nauges and Strand (2013) reported a 60 per cent increase in girls' school attendance in Ghana as a result of only 15 minutes reduction in water collection time.

3.4 Governance and the Golden Rule

The Canadian International Development Agency (CIDA) contributed through numerous projects to development activities in Ghana, most notably the provision of potable water through boreholes in the Upper West and Upper East Regions, relatively poor and food-insecure areas of Ghana due to several historical, climatic, agricultural, and institutional factors. Attempts by governmental agencies such as the Ministry of Food and Agriculture (MFA) and several local, national and international organizations, including CIDA, has resulted in only modest improvements (Dittoh, 2006; UNEP, 2011).

The water debate in relation to poverty alleviation has a dimension that is quite often ignored: its relationship with public governance. The key question is: can local authorities best use their influence, experience, support and interest to increase the threshold of treated wastewater reuse in agriculture? To help answer this question, the role of metropolitan, municipal and district assemblies in Ghana in implementing strategic national green policies becomes of interest. Bulkeley and Kern (2006) identify a new governance framework that is of particular interest in this research. This framework consists of the following four modes: governance by provision, governance by authority, governance through enabling and self-governing.

Governance by provision is defined as the shaping of local council practice through the delivery of particular forms of service and resource (Bulkeley and Kern, 2006). In essence, through the provision of direct services, local governments are able both to control the nature of infrastructure development and to influence practices of public consumption. Governing by authority refers to situations in which national governments intervene directly in local politics through mandates or other mandatory means (Jollands *et al*., 2009).

Governing through enabling refers to situations where national governments stimulate local action by providing the enabling conditions of such action. It is the form of governance when local governments facilitate, co-ordinate, and encourage action through partnerships, private voluntary-sector agencies, and various forms of community engagement. Self-governing is the capacity of local government to govern its own activities and is accomplished through better self-management (i.e. within its own infrastructure). Although this option has not been seen on large scale, it may be used as pilot projects for green policies.

There are broadly two approaches to increase or maximize the impact of local governments in the context of environmental conservation, and particularly in terms of water reuse (Kelly and Pollitt, 2011). The first approach is to increase the size of local council’s playing field through decentralizing central government power, and thus increasing assemblies’ responsibility. Water is traditionally being thought to be outside the remit of local authorities and hence local governments can take control over water consumption, treatment, and reuse within their districts.

The second approach is to effectively utilize the playing field of the assemblies as much as possible. This involves better management of existing resources and processes. As we learn from Singapore, which had large proportion of its wastewater discharged to the environment untreated at all in 1970s, Singaporean Parliament passed the "Environmental Public Health Bill" and the "Local Government Integration Ordinance" that required municipalities to develop their own plans and enforcement measures supported by a Presidential Ani-Pollution Unit (APU) (Tortajada *et al*., 2013).

The purpose of the plan was two-fold: to balance environmental considerations and economic imperatives, thus avoiding the sort of environmental destruction many developing countries are experiencing as a result of policies aimed at attracting "investments" at all costs, and to adopt a spirit of enforcement through persuasion and advice along with strong fining mechanisms rather than coercion (Cleary, 1970; Hernandez, 1993).

In order to secure widespread demand, local assemblies should adopt a Golden Rule for treated wastewater providers. The Golden Rule can be defined as the criterion by which expected savings on fertilisers, crop yield, and water bills exceed all associated repayments. The new strategy aims both to revolutionize the efficiency of wastewater treatment and rate of reuse at farms and businesses, and to help farmers insulate against fluctuating fertilizers prices, thus creating more secure and independent agricultural sector that is cheaper to run. A life cycle assessment that follows local conditions and prices can determine whether such policy satisfies the Golden Rule.

3.5 Technical appropriateness of treated wastewater used in irrigation

Irrigated agriculture will play a noticeable role in sustainable crop production to feed the rapidly increasing population of Ghana (Amoah, 2008). However, the use of treated wastewater for irrigation is subject to meeting the environmental and hygienic requirements recommended by the Food and Agriculture Organization (Pescod, 1992).

Risks to plant health are reduced if there is little or no industrial effluent in the wastewater, but in all cases, five parameters should be monitored during the irrigation season: electrical conductivity, the sodium adsorption ratio, boron, total nitrogen, and pH (The World Bank, 2010).

***i*. Electrical conductivity**: As a measure of the "salinity hazard", the electrical conductivity should be less than 70 to 300 milli Siemens per metre (mS/m) at 20 degrees Celsius (Rhoades *et al*., 1992). Irrigation with water that is too saline causes interference with the capacity of a plant's roots to absorb water and nutrients, and therefore reduces crop yield (Mara, 2004). However, if the salinity of the applied water exceeds 300 mS/m, the water might still be usable but its use may need to be restricted to more permeable soils and more salt-tolerant crops, where high leaching fractions are more easily achieved (Ayers and Westcot, 1994).

Ghana's urban vegetable production has been impacted by: (1) general scarcity of non-saline surface- or ground-waters including perennial streams, (2) the saline nature of the soils, and (3) unsuitable hilly topography with flood-prone flatlands. Significant agricultural areas are cultivated with crops that are tolerant to salinity (Tables 3.1 and 3.2).

Secondly, groundwater resources in Ghana are facing severe deterioration due to water intrusion, poor overall management, and declining groundwater reserves, which in turn contributes to elevated salinities and electrical conductivities starting from 281 mS/m (Darnault, 2008; Banoeng-Yakubo *et al*., 2009; Namara *et al*., 2010; Yidana *et al*., 2011; Nnadi *et al*., 2015). It is believed that more water is pumped or discharged from aquifers than is replenished naturally. Fertigation will significantly reduce dependency on groundwater as resource for agriculture, which will in turn help fight back sea water intrusion.

Table 3.1 Cultivated land area planted for selected food crops (in thousand hectares).

Crop	*2009*	*2010*	**Crop**	*2009*	*2010*	**Crop**	*2009*	*2010*
Maize	954	992	**Plantain**	325	328	**Sorghum**	267	253
Millet	187	177	**Yam**	379	385	**Cocoyam**	225	205

Rice	162	181	**Cocoa**	NI[a]	1,600	**Oil palm**	NI	350

[a] NI: No Information. Source: MFA (2011).

Table 3.2 Crops and salinity thresholds.

Crop	*Salinity Threshold (mS/m)*	**Crop**	*Salinity Threshold (mS/m)*
Maize	250	**Cotton**	770
Date palm	680	**Sorghum**	740
Olive	500	**Wheat**	740
Millet, channel	T[a,b]	**Lucerne**	340
Sudan grass	510	**Rice, paddy**	300[b]
Lemon	200	**Barley**	1000
Valencia	240	**Sunflower**	480

[a] T: Tolerant but salinity tolerance level is not determined.
[b] Sources: Pescod (1992), Tanji and Kielen (2002).

***ii*. Sodium adsorption ratio (SAR)**: Excess sodium in relation to calcium and magnesium concentrations in soils: (1) causes adverse physicochemical changes in the soil destroying its structure, (2) affects its internal drainage reducing permeability of the soil to water and air, and (3) causes difficulty for crops to absorb nutrients and water (Pescod, 1992; MPDB, 2000). For safe irrigation, the SAR should be less than 18 (Mara, 2004).

***iii*. Boron**: Irrigation water contaminated with boron is one of the main causes of boron toxicity in plants, and it is the continued use and concentration of boron in soil, especially in hot regions with high evapotranspiration rates and increased salinity levels, that leads to toxicity problems (Gupta *et al*., 1985; Butterwick *et al*., 1989). Literature lacks sufficient data on boron occurrence and concentration in raw wastewater and treated effluents in Ghana.

***iv*. Total nitrogen concentration**: Nitrogen in the irrigation water has much the same effect as soil-applied nitrogen fertilizer and an excess will cause problems, just as too much fertilizer would. Too much nitrogen can reduce crop yields or cause crop damage (Mara, 2004). A study on activated sludge plants in Ghana found effluent total nitrogen values between 16 and 54 milli-grams total nitrogen per litre (mg TN/L) for the wet and dry seasons, respectively (Adonadaga, 2014). Most commonly grown crops can tolerate 30 mg TN/L, but the Environmental Protection Agency of Ghana has set the total nitrogen limit at 50 mg TN/L (Ayers and Westcot, 1994; Adonadaga, 2014).

***v.* pH**: The pH range for irrigation water is 6.5 to 8.4, which does not present any problems for Ghana's treated effluent. For example, activated sludge effluents had a mean pH of 8.1, and WSPs effluents generally have lower pH levels (Sarkodie *et al*., 2014).

Not all trace elements are toxic and in small quantities many, such as iron, manganese, molybdenum, and zinc, are essential for plant growth. However, excessive quantities will cause undesirable accumulations in plant tissue and reduce growth. A comprehensive study of trace elements present in Ghana's treated wastewater effluents has not been conducted yet, but it is thought that concentrations are below the maximum recommended Food and Agriculture Organization (FAO) limits knowing that generated wastewater in arid and semi-arid developing countries is predominantly domestic.

3.6 Part I: Water decision-making

3.6.1 Research question

Headline Question: What are the main categories of constraints to treated wastewater reuse in agriculture in Ghana? What scale of perceived impact do they represent to institutions in the water sector?

3.6.2 Methodology and scope of work

The research has been approved by the ethics review panel in accordance with the University of Western Ontario ethics protocols and the Canadian Tri-Council Policy Statement on ethical conduct of research involving humans. All interviews were conducted in English. In-depth semi-structured interviews were conducted with policy-makers engaged in governance or management in the urban water sector. The author sought to interview those with expertise in water management and policy, specifically individuals who had significant experience with science-policy interfaces using a snowball sampling strategy. Interviews took approximately one and a half hour to complete. With respondent written consent, interviews were audio-taped and transcriptions were prepared or reviewed by the interviewer.

A stakeholder analysis was performed in order to involve the "right stakeholders" according to a matrix of their authority, support, influence, and interest or need. The seven participating organizations represented local authorities, academia, and not-for-profit organizations. The majority held senior or executive level positions within their organizations, and were able to speak to decision-making processes in urban water policy or management and the use of academic and scientific research within those processes.

The interview guide (including questions and probes), email script, consent, and letter of information were prepared and also approved by the ethics review panel of the University. Questions in interviews' guide included (but are not limited to):

- Is the organization currently involved in or planning any projects or initiatives on wastewater treatment?
  If so:
    - What are the objective(s) of the project/initiative?
    - What stage are you at?
    - Who is involved internally and externally? Do you plan to coordinate with others, e.g. national, international and not-for-profit organizations?
    - What is the project timeline?
    - What do you foresee as the benefits of the project?
    - Can the project goals be adjusted according to changing conditions? Are you willing to consult others? Who?
- In what ways do you think the organization could engage with the water sector?
- How do you think farmers would benefit from your wastewater treatment project/policy?
- Are they engaged or involved in any organization-sponsored awareness/educational programs?

    - How far do you think sponsored training programs would affect their acceptance to reuse treated wastewater in their farms?
    - Are there any examples in your area of jurisdiction/operations?
    - Does such a strategy have more impact on a certain gender than another? Why?
    - Does such a strategy have more impact on a certain age group than others? Why?
    - Do you think there are religious or traditional influences on reusing treated wastewater for agriculture?
- Which organizations would you work with to encourage treated wastewater reuse in agriculture? Why?
- Can your experience of similar programs or initiatives inform you of the best approach to follow in the future?
  If so:
    - Who are/are likely to be your key partners? Are there any arrangements currently in place?
    - How do you think the authority of local assemblies can be best utilized for local implementation of a water reuse policy?
    - What do you think should be done to educate people about the beneficial resources present in wastewater? Who do you think can best disseminate information (mainly among farmers)? How?
- What do you see as the opportunities for the organization objectives and strategy?
- What do you think the barriers are, or would be, to the implementation of your project/policy? Categorize them (financial, technical, social, institutional… etc.).
    - Have you had any similar challenges to your previous projects/initiatives? How did you tackle them?
    - Do you think there are any ways to overcome these barriers?
    - Do you think there is a role for central Government through which it can help overcome these barriers?

- What particular aspects in the national policy need to be strengthened in your opinion to increase water reuse in agriculture?

All interviews were conducted in person at the participating organization. Transcripts were standardized so to facilitate distinction between the interviewer and respondent. Hard returns were omitted between the question and response so that both the question and the response would be captured as a single unit during analysis. After transcripts were standardized, they were imported in Microsoft Word into NVivo and coded at the University of Western Ontario. To develop expertise in NVivo, the researcher completed a self-study tutorial in approximately 40 hours. Consistent keywords and ideas were grouped together in this process to identify consistent themes throughout the interviews.

Themes were ranked by the frequency in which they emerged. However, this ranking process was not simply quantitative; as the interviewees themselves often identified many relevant factors but usually placed particular emphasis on just a handful. The researcher ensured internal validity through follow up discussions of thematic analysis (including the interpretation of complex examples) ranking the most salient themes; which were based not just on frequency but also the importance that interviewees themselves placed upon them.

These weights were incorporated in the Analytical Hierarchy Process (AHP). The AHP is based upon the premise that to make a decision: "…we need to know the problem, the need and purpose of the decision, the criteria of the decision, their sub-criteria, stakeholders and groups affected and the alternative actions to take" (Saaty, 2008). The successful development of an AHP survey rests upon four main stages (Wattage and Mardle, 2007):

1. The development of a hierarchy of criteria;
2. A pairwise comparison survey to elicit the preferences of respondents;
3. Analysis of respondents' results;
4. Aggregation of data to establish the relative importance attributed to variables.

While the relative simplicity of the AHP means that it is not necessary for surveys to be conducted using face-to-face interviewing, this approach can be advantageous as it allows the researcher to explain and demonstrate the technique prior to completion. Given that respondents were unlikely to have encountered the AHP before, a decision was taken to use face-to-face interviewing instead of a mail survey. Fieldwork was conducted between May and November 2016.

One of the advantages of the AHP methodology is that it does not require probability sampling, nor necessarily large samples, in order to generate valid and reliable results. However, it is necessary to ensure that respondents are selected on the basis of being representative of the experience under study.

3.6.3 Analysis and results

3.6.3.1 Developing the AHP hierarchy

Development of the AHP hierarchy involves the researcher establishing a clear goal and then identifying the criteria and sub-criteria that may realistically influence this goal. It is therefore essential that the hierarchy is representative of the system under study, and that the criteria are clear and convey the same meaning to all respondents. Having established the goal as 'identifying constraints to water reuse in agriculture', a hierarchy was developed using multiple data sources. The outcome of this process was a hierarchy consisting of four first-tier criteria and twelve second-tier sub-criteria (Table 3.3).

Table 3.3 Hierarchy of challenges to water reuse in agriculture.

Technical	**Institutional**	**Financial**	**Social**
Knowledge of wastewater treatment	Environmental regulations	Profitability	Attitudes and preferences
Knowledge of engineering design	Provision of water services	Financial risk	Knowledge and skills
Knowledge of toxicity requirements	Quality assurance	Lack of capital	Formal qualifications
Knowledge of microbiological issues	Accessibility	Fertilizer market	Perceptions

Consideration was given to the number of criteria and resultant pairwise comparisons presented to respondents to avoid fatigue and inconsistency. The number of pairwise comparisons can be derived using the formula: [n(n−1)/2], where n represents the number of individual criteria. It is possible to reduce the number of pairwise comparisons by comparing all of the first-tier criteria, but only comparing sub-criteria within their respective criteria groups only – rather than both within and between groups (Whitmarsh and Palmieri, 2009). The application of this technique for the hierarchy in Table 3.3 resulted in six first-tier pairwise comparisons, and a further twenty-four second-tier comparisons. However, a matrix of twenty-four pairwise comparisons was deemed to be excessive; resulting in a decision to focus upon the first-tier of criteria only (four pairwise comparisons).

3.6.3.2 Pairwise comparisons, AHP calculations, and results

3.6.3.2.1 Local Authority I

Table 3.4 Developed scale of impact of categorized constraints to water reuse in agriculture for Local Authority I.

Criteria		More important	Scale
A	**B**	**A or B**	**(0 to 9)**
Technical	Institutional	B	4
	Social	B	9
	Financial	B	2
Institutional	Social	B	7
	Financial	A	3
Social	Financial	A	9

Therefore, the developed AHP matrix is:

$$\text{Matrix A: } \begin{bmatrix} 1 & 0.25 & 0.11 & 0.5 \\ 4 & 1 & 1/7 & 3 \\ 9 & 7 & 1 & 9 \\ 2 & 0.33 & 1/9 & 1 \end{bmatrix}$$

The row geometric mean prioritization method (RGMM) is one of the most extended AHP prioritization procedures (Dong *et al.*, 2010). To calculate RGMM:

$$\text{Matrix B: } \ln \begin{bmatrix} 1 & 0.25 & 0.11 & 0.5 \\ 4 & 1 & 1/7 & 3 \\ 9 & 7 & 1 & 9 \\ 2 & 0.33 & 1/9 & 1 \end{bmatrix} = \begin{bmatrix} - & -1.39 & -2.2 & -0.69 \\ 1 & - & -1.95 & 1.1 \\ 2 & 1.95 & - & 2.2 \\ 1 & -1.1 & -2.2 & - \end{bmatrix}$$

For the first row: $\dfrac{\sum Row}{Number\ of\ criteria} = \dfrac{-1.39-2.2-0.69}{4} = -1.1$

The exponential is: $e^{-1.1} = 0.32$

Results for all rows: $\begin{bmatrix} 0.32 \\ 1.14 \\ 4.88 \\ 0.52 \end{bmatrix}$

The sum is: 6.88.

RGMM = $\dfrac{0.32}{6.88} = 5\%$

This process is repeated for all rows.

The resultant RGMM matrix: $\begin{bmatrix} 5\% \\ 17\% \\ 71\% \\ 8\% \end{bmatrix} \begin{Bmatrix} Technical\ Challenges \\ Institutional\ Challenges \\ Social\ Challenges \\ Financial\ Challenges \end{Bmatrix}$

The consistency ratio (CR) compares the consistency index (CI) of the matrix in question (the one with our judgments) with the consistency index of a random-like matrix (RI). A random matrix is one where the judgments have been entered randomly and therefore it is expected to be highly inconsistent. More specifically, RI is the average CI of 500 randomly filled-in matrices, and is equal to 0.9 for a matrix of 4 criteria. To calculate the consistency ratio:

Prioritization results (the resultant RGMM matrix): $\begin{bmatrix} 0.05 \\ 0.17 \\ 0.71 \\ 0.08 \end{bmatrix}$

Calculation of weighed columns: $\begin{bmatrix} 0.05 & 0.0425 & 0.0781 & 0.04 \\ 0.2 & 0.17 & 0.101 & 0.24 \\ 0.45 & 1.19 & 0.71 & 0.72 \\ 0.1 & 0.0561 & 0.079 & 0.08 \end{bmatrix}$

Calculation of weighed columns by summing rows: $\begin{bmatrix} 0.2106 \\ 0.711 \\ 3.07 \\ 0.3141 \end{bmatrix}$

$$\frac{Weighed\ sum}{Priority} = \begin{bmatrix} 4.21 \\ 4.18 \\ 4.32 \\ 3.93 \end{bmatrix},\ \sum \ = 16.6$$

The principle eigenvalue $\lambda_{\max} = \dfrac{16.6}{4} = 4.157$

To calculate the consistency index (CI):

$$CI = \frac{\lambda_{\max} - n}{n-1} = \frac{4.157-4}{4-1} = 0.052$$

To calculate the consistency ratio (CR):

$$CR = \frac{CI}{RI} = \frac{0.052}{0.9} = 5.8\%$$

Saaty (2012) has shown that judgement matrices consistency ratio (CR) of 0.10 or less is acceptable. If the consistency ratio is greater than 0.10, it is necessary to revise the judgments to locate the cause of the inconsistency and correct it.

This process is repeated 6 more times for Local Authorities 2, 3, and 4, Academia 1, and International non-governmental organizations 1, and 2. See Appendix A.

3.6.3.2.2 Local Authority II

Table 3.5 Developed scale of impact of categorized constraints to water reuse in agriculture for Local Authority II.

Criteria		More important	Scale
A	**B**	**A or B**	**(0 to 9)**
Technical	Institutional	A	1
	Social	B	1
	Financial	B	3
Institutional	Social	A	2
	Financial	B	5
Social	Financial	B	2

Therefore, the developed AHP matrix is:

$$\text{Matrix A: } \begin{bmatrix} 1 & 1 & 1 & 0.3 \\ 1 & 1 & 2 & 0.2 \\ 1 & 0.5 & 1 & 0.5 \\ 3 & 5 & 2 & 1 \end{bmatrix}$$

To calculate RGMM:

$$\text{Matrix B: ln } \begin{bmatrix} 1 & 1 & 1 & 0.3 \\ 1 & 1 & 2 & 0.2 \\ 1 & 0.5 & 1 & 0.5 \\ 3 & 5 & 2 & 1 \end{bmatrix} = \begin{bmatrix} - & - & - & -1 \\ - & - & 0.69 & -2 \\ - & -0.69 & - & -1 \\ 1 & 1.61 & 0.69 & - \end{bmatrix}$$

For the first row: $\dfrac{\sum Row}{Number\ of\ criteria} = \dfrac{-1}{4} = -0.25$

The exponential is: $e^{-0.25} = 0.75$

Results for all rows: $\begin{bmatrix} 0.75 \\ 0.72 \\ 0.655 \\ 2.281 \end{bmatrix}$

The sum is: 4.406.

RGMM = $\frac{0.75}{4.406} = 17\%$

This process is repeated for all rows.

The resultant RGMM matrix: $\begin{bmatrix} 17\% \\ 17\% \\ 15\% \\ 51\% \end{bmatrix} \begin{Bmatrix} \textit{Technical Challenges} \\ \textit{Institutional Challenges} \\ \textit{Social Challenges} \\ \textit{Financial Challenges} \end{Bmatrix}$

To calculate the consistency ratio:

Prioritization results (the resultant RGMM matrix): $\begin{bmatrix} 0.17 \\ 0.17 \\ 0.15 \\ 0.51 \end{bmatrix}$

Calculation of weighed columns: $\begin{bmatrix} 0.17 & 0.17 & 0.15 & 0.153 \\ 0.17 & 0.17 & 0.3 & 0.102 \\ 0.17 & 0.085 & 0.15 & 0.255 \\ 0.51 & 0.85 & 0.3 & 0.51 \end{bmatrix}$

Calculation of weighed columns by summing rows: $\begin{bmatrix} 0.643 \\ 0.742 \\ 0.66 \\ 2.17 \end{bmatrix}$

$$\frac{Weighed\ sum}{Priority} = \begin{bmatrix} 3.78 \\ 4.36 \\ 4.4 \\ 4.25 \end{bmatrix}, \sum = 16.79$$

The principle eigenvalue $\lambda_{\max} = \frac{16.79}{4} = 4.198$

To calculate the consistency index (CI):

$$CI = \frac{\lambda_{\max} - n}{n - 1} = \frac{4.198 - 4}{4 - 1} = 0.066$$

To calculate the consistency ratio (CR):

$$CR = \frac{CI}{RI} = \frac{0.066}{0.9} = 7.3\%$$

3.6.3.2.3 Local Authority III

Table 3.6 Developed scale of impact of categorized constraints to water reuse in agriculture for Local Authority III.

Criteria		**More important**	**Scale**
A	**B**	**A or B**	**(0 to 9)**
Technical	Institutional	A	2
	Social	B	1
	Financial	A	3
Institutional	Social	B	6
	Financial	B	1
Social	Financial	A	7

Therefore, the developed AHP matrix is:

Matrix A: $\begin{bmatrix} 1 & 2 & 1 & 3 \\ 1 & 1 & 0.17 & 1 \\ 1 & 6 & 1 & 7 \\ 0.3 & 1 & 0.14 & 1 \end{bmatrix}$

To calculate RGMM:

Matrix B: $\ln \begin{bmatrix} 1 & 2 & 1 & 3 \\ 1 & 1 & 0.17 & 1 \\ 1 & 6 & 1 & 7 \\ 0.3 & 1 & 0.14 & 1 \end{bmatrix} = \begin{bmatrix} - & 0.69 & - & 1.1 \\ - & - & -1.77 & - \\ - & 1.79 & - & 1.95 \\ -1.2 & - & -1.97 & - \end{bmatrix}$

For the first row: $\dfrac{\sum Row}{Number\ of\ criteria} = \dfrac{1.79}{4} = 0.45$

The exponential is: $e^{0.45} = 1.56$

Results for all rows: $\begin{bmatrix} 1.56 \\ 0.17 \\ 2.55 \\ 0.45 \end{bmatrix}$

The sum is: 4.73.

RGMM = $\dfrac{1.56}{4.73} = 33\%$

This process is repeated for all rows.

The resultant RGMM matrix: $\begin{bmatrix} 33\% \\ 4\% \\ 53\% \\ 10\% \end{bmatrix} \begin{Bmatrix} Technical\ Challenges \\ Institutional\ Challenges \\ Social\ Challenges \\ Financial\ Challenges \end{Bmatrix}$

To calculate the consistency ratio:

Prioritization results (the resultant RGMM matrix): $\begin{bmatrix} 0.33 \\ 0.04 \\ 0.54 \\ 0.1 \end{bmatrix}$

Calculation of weighed columns: $\begin{bmatrix} 0.33 & 0.08 & 0.53 & 0.3 \\ 0.33 & 0.04 & 0.09 & 0.1 \\ 0.33 & 3.18 & 0.53 & 0.7 \\ 0.1 & 0.04 & 0.07 & 0.1 \end{bmatrix}$

Calculation of weighed columns by summing rows: $\begin{bmatrix} 1.24 \\ 0.53 \\ 4.74 \\ 0.31 \end{bmatrix}$

$$\frac{Weighed\ sum}{Priority} = \begin{bmatrix} 3.76 \\ 13.25 \\ 8.94 \\ 3.1 \end{bmatrix}, \sum = 29.05$$

The principle eigenvalue $\lambda_{\max} = \frac{29.05}{4} = 7.26$

To calculate the consistency index (CI):

$$CI = \frac{\lambda_{\max} - n}{n-1} = \frac{7.26-4}{4-1} = 0.0108$$

To calculate the consistency ratio (CR):

$$CR = \frac{CI}{RI} = \frac{0.0108}{0.9} = 12\%$$

3.6.3.2.4 Local Authority IV

Table 3.7 Developed scale of impact of categorized constraints to water reuse in agriculture for Local Authority IV.

Criteria		More important	Scale
A	**B**	**A or B**	**(0 to 9)**
Technical	Institutional	A	2
	Social	A	5
	Financial	B	3
Institutional	Social	A	3
	Financial	B	8
Social	Financial	B	5

Therefore, the developed AHP matrix is:

$$\text{Matrix A:}\begin{bmatrix} 1 & 2 & 5 & 0.3 \\ 1 & 1 & 3 & 0.1 \\ 0 & 0.3 & 1 & 0.2 \\ 3 & 8 & 5 & 1 \end{bmatrix}$$

To calculate RGMM:

$$\text{Matrix B: ln}\begin{bmatrix} 1 & 2 & 5 & 0.3 \\ 1 & 1 & 3 & 0.1 \\ 0 & 0.3 & 1 & 0.2 \\ 3 & 8 & 5 & 1 \end{bmatrix} = \begin{bmatrix} - & 0.69 & 1.61 & -1.2 \\ - & - & 1.1 & -2.3 \\ 1 & -1.2 & - & -1.61 \\ 1.1 & 2.08 & 1.61 & - \end{bmatrix}$$

For the first row: $\dfrac{\sum Row}{Number\ of\ criteria} = \dfrac{1.1}{4} = 0.275$

The exponential is: $e^{0.275} = 1.32$

$$\text{Results for all rows:}\begin{bmatrix} 1.32 \\ 0.74 \\ 0.63 \\ 3.3 \end{bmatrix}$$

The sum is: 5.99.

RGMM = $\frac{1.32}{5.99} = 22\%$

This process is repeated for all rows.

The resultant RGMM matrix: $\begin{bmatrix} 22\% \\ 12\% \\ 11\% \\ 55\% \end{bmatrix} \begin{Bmatrix} \textit{Technical Challenges} \\ \textit{Institutional Challenges} \\ \textit{Social Challenges} \\ \textit{Financial Challenges} \end{Bmatrix}$

To calculate the consistency ratio:

Prioritization results (the resultant RGMM matrix): $\begin{bmatrix} 0.22 \\ 0.12 \\ 0.11 \\ 0.55 \end{bmatrix}$

Calculation of weighed columns: $\begin{bmatrix} 0.22 & 0.24 & 0.55 & 0.165 \\ 0.22 & 0.12 & 0.33 & 0.055 \\ 0 & 0.036 & 0.11 & 0.11 \\ 0.11 & 0.96 & 0.12 & 0.55 \end{bmatrix}$

Calculation of weighed columns by summing rows: $\begin{bmatrix} 1.175 \\ 0.725 \\ 0.256 \\ 1.74 \end{bmatrix}$

$$\frac{\textit{Weighed sum}}{\textit{Priority}} = \begin{bmatrix} 5.34 \\ 6.04 \\ 2.33 \\ 3.16 \end{bmatrix}, \sum = 16.87$$

The principle eigenvalue $\lambda_{\max} = \frac{16.87}{4} = 4.22$

To calculate the consistency index (CI):

$$CI = \frac{\lambda_{\max} - n}{n-1} = \frac{4.22-4}{4-1} = 0.072$$

To calculate the consistency ratio (CR):

$$CR = \frac{CI}{RI} = \frac{0.072}{0.9} = 8\%$$

3.6.3.2.5 Academia I

Table 3.8 Developed scale of impact of categorized constraints to water reuse in agriculture for Academia I.

Criteria		More important	Scale
A	**B**	**A or B**	**(0 to 9)**
Technical	Institutional	B	9
	Social	B	6
	Financial	A	2
Institutional	Social	B	1
	Financial	A	8
Social	Financial	A	7

Therefore, the developed AHP matrix is:

$$\text{Matrix A:} \begin{bmatrix} 1 & 0.11 & 0.17 & 2 \\ 9 & 1 & 1 & 8 \\ 6 & 1 & 1 & 7 \\ 1 & 0.13 & 0.14 & 1 \end{bmatrix}$$

To calculate RGMM:

$$\text{Matrix B: ln} \begin{bmatrix} 1 & 0.11 & 0.17 & 2 \\ 9 & 1 & 1 & 8 \\ 6 & 1 & 1 & 7 \\ 1 & 0.13 & 0.14 & 1 \end{bmatrix} = \begin{bmatrix} - & -2.2 & -1.77 & 0.69 \\ 2.2 & - & - & 2.08 \\ 1.8 & - & - & 1.95 \\ - & -2.4 & -1.97 & - \end{bmatrix}$$

For the first row: $\dfrac{\sum Row}{Number\ of\ criteria} = \dfrac{-3.28}{4} = -0.82$

The exponential is: $e^{-0.82} = 0.44$

Results for all rows: $\begin{bmatrix} 0.44 \\ 2.9 \\ 2.56 \\ 0.34 \end{bmatrix}$

The sum is: 6.24.

RGMM = $\dfrac{0.44}{6.24} = 7\%$

This process is repeated for all rows.

The resultant RGMM matrix: $\begin{bmatrix} 7\% \\ 46\% \\ 41\% \\ 6\% \end{bmatrix} \begin{Bmatrix} Technical\ Challenges \\ Institutional\ Challenges \\ Social\ Challenges \\ Financial\ Challenges \end{Bmatrix}$

To calculate the consistency ratio:

Prioritization results (the resultant RGMM matrix): $\begin{bmatrix} 0.07 \\ 0.46 \\ 0.41 \\ 0.06 \end{bmatrix}$

Calculation of weighed columns: $\begin{bmatrix} 0.07 & 0.05 & 0.07 & 0.12 \\ 0.63 & 0.46 & 0.41 & 0.48 \\ 0.42 & 0.46 & 0.41 & 0.42 \\ 0.07 & 0.057 & 0.057 & 0.06 \end{bmatrix}$

Calculation of weighed columns by summing rows: $\begin{bmatrix} 0.31 \\ 1.98 \\ 1.71 \\ 0.244 \end{bmatrix}$

$$\frac{Weighed\ sum}{Priority} = \begin{bmatrix} 4.43 \\ 4.3 \\ 4.17 \\ 4.07 \end{bmatrix}, \sum = 16.67$$

The principle eigenvalue $\lambda_{\max} = \frac{16.67}{4} = 4.17$

To calculate the consistency index (CI):

$$CI = \frac{\lambda_{\max} - n}{n-1} = \frac{4.17-4}{4-1} = 0.056$$

To calculate the consistency ratio (CR):

$$CR = \frac{CI}{RI} = \frac{0.056}{0.9} = 6.2\%$$

3.6.3.2.6 International Non-Governmental Organization I

Table 3.9 Developed scale of impact of categorized constraints to water reuse in agriculture for International Non-Governmental Organization I.

Criteria		More important	Scale
A	**B**	**A or B**	**(0 to 9)**
Technical	Institutional	A	2
	Social	B	5
	Financial	B	8
Institutional	Social	B	4
	Financial	B	7
Social	Financial	B	2

Therefore, the developed AHP matrix is:

$$\text{Matrix A: } \begin{bmatrix} 1 & 2 & 0.2 & 0.1 \\ 1 & 1 & 0.25 & 0.1 \\ 5 & 4 & 1 & 0.5 \\ 8 & 7 & 2 & 1 \end{bmatrix}$$

To calculate RGMM:

$$\text{Matrix B: } \ln \begin{bmatrix} 1 & 2 & 0.2 & 0.1 \\ 1 & 1 & 0.25 & 0.1 \\ 5 & 4 & 1 & 0.5 \\ 8 & 7 & 2 & 1 \end{bmatrix} = \begin{bmatrix} - & 0.69 & -1.61 & -2.3 \\ - & - & -1.39 & -2.3 \\ 1.61 & 1.39 & - & -0.693 \\ 2.08 & 1.95 & 0.69 & - \end{bmatrix}$$

For the first row: $\dfrac{\sum Row}{Number\ of\ criteria} = \dfrac{-3.22}{4} = -0.805$

The exponential is: $e^{-0.805} = 0.45$

Results for all rows: $\begin{bmatrix} 0.45 \\ 0.4 \\ 1.78 \\ 3.25 \end{bmatrix}$

The sum is: 5.88.

RGMM = $\dfrac{0.45}{5.88} = 7\%$

This process is repeated for all rows.

The resultant RGMM matrix: $\begin{bmatrix} 7\% \\ 7\% \\ 30\% \\ 54\% \end{bmatrix} \begin{Bmatrix} Technical\ Challenges \\ Institutional\ Challenges \\ Social\ Challenges \\ Financial\ Challenges \end{Bmatrix}$

To calculate the consistency ratio:

Prioritization results (the resultant RGMM matrix): $\begin{bmatrix} 0.07 \\ 0.07 \\ 0.3 \\ 0.54 \end{bmatrix}$

Calculation of weighed columns: $\begin{bmatrix} 0.07 & 0.14 & 0.06 & 0.054 \\ 0.07 & 0.07 & 0.075 & 0.054 \\ 0.35 & 0.28 & 0.3 & 0.27 \\ 0.56 & 0.49 & 0.6 & 0.54 \end{bmatrix}$

Calculation of weighed columns by summing rows: $\begin{bmatrix} 0.324 \\ 2.69 \\ 1.2 \\ 2.19 \end{bmatrix}$

$$\frac{Weighed\ sum}{Priority} = \begin{bmatrix} 4.63 \\ 3.84 \\ 4 \\ 4.06 \end{bmatrix}, \sum = 16.53$$

The principle eigenvalue $\lambda_{\max} = \dfrac{16.53}{4} = 4.13$

To calculate the consistency index (CI):

$$CI = \frac{\lambda_{\max} - n}{n-1} = \frac{4.13-4}{4-1} = 0.044$$

To calculate the consistency ratio (CR):

$$CR = \frac{CI}{RI} = \frac{0.044}{0.9} = 4.9\%$$

3.6.3.2.7 International Non-Governmental Organization II

Table 3.10 Developed scale of impact of categorized constraints to water reuse in agriculture for International Non-Governmental Organization II.

Criteria		More important	Scale
A	**B**	**A or B**	**(0 to 9)**
Technical	Institutional	B	8
	Social	A	2
	Financial	B	7
Institutional	Social	A	7
	Financial	A	3
Social	Financial	B	6

Therefore, the developed AHP matrix is:

$$\text{Matrix A:} \begin{bmatrix} 1 & 0.13 & 2 & 0.1 \\ 8 & 1 & 7 & 3 \\ 1 & 0.14 & 1 & 0.2 \\ 7 & 0.33 & 6 & 1 \end{bmatrix}$$

To calculate RGMM:

$$\text{Matrix B: ln} \begin{bmatrix} 1 & 0.13 & 2 & 0.1 \\ 8 & 1 & 7 & 3 \\ 1 & 0.14 & 1 & 0.2 \\ 7 & 0.33 & 6 & 1 \end{bmatrix} = \begin{bmatrix} - & -2.04 & 0.69 & -2.3 \\ 2.08 & - & 1.95 & 1.1 \\ - & -1.97 & - & -1.61 \\ 1.95 & -1.11 & 1.79 & - \end{bmatrix}$$

For the first row: $\dfrac{\sum Row}{Number\ of\ criteria} = \dfrac{-3.65}{4} = -0.913$

The exponential is: $e^{-913} = 0.401$

$$\text{Results for all rows:} \begin{bmatrix} 0.401 \\ 3.6 \\ 0.4 \\ 1.93 \end{bmatrix}$$

The sum is: 6.33.

RGMM = $\frac{0.401}{6.33} = 6\%$

This process is repeated for all rows.

The resultant RGMM matrix: $\begin{bmatrix} 6\% \\ 56\% \\ 6\% \\ 32\% \end{bmatrix} \begin{Bmatrix} \textit{Technical Challenges} \\ \textit{Institutional Challenges} \\ \textit{Social Challenges} \\ \textit{Financial Challenges} \end{Bmatrix}$

To calculate the consistency ratio:

Prioritization results (the resultant RGMM matrix): $\begin{bmatrix} 0.06 \\ 0.56 \\ 0.06 \\ 0.32 \end{bmatrix}$

Calculation of weighed columns: $\begin{bmatrix} 0.06 & 0.07 & 0.12 & 0.032 \\ 0.48 & 0.56 & 0.42 & 0.96 \\ 0.06 & 0.08 & 0.06 & 0.064 \\ 0.42 & 0.18 & 0.36 & 0.32 \end{bmatrix}$

Calculation of weighed columns by summing rows: $\begin{bmatrix} 0.282 \\ 2.42 \\ 0.204 \\ 1.28 \end{bmatrix}$

$$\frac{\textit{Weighed sum}}{\textit{Priority}} = \begin{bmatrix} 4.7 \\ 4.3 \\ 3.4 \\ 4 \end{bmatrix}, \sum = 16.4$$

The principle eigenvalue $\lambda_{\max} = \frac{16.4}{4} = 4.1$

To calculate the consistency index (CI): $CI = \frac{\lambda_{\max} - n}{n-1} = \frac{4.1-4}{4-1} = 0.033$

To calculate the consistency ratio (CR): $CR = \frac{CI}{RI} = \frac{0.033}{0.9} = 3.7\%$

Summary and combined AHP results for each participating organization are given in Tables 3.11, and 3.12.

Table 3.11 Summary of the results for each participating organization for the challenges of water reuse in agriculture.

Organization	Technical challenges	Institutional challenges	Social challenges	Financial challenges	Consistency ratio
LA I	5%	17%	71%	8%	5.8%
LA II	17%	17%	15%	51%	7.3%
LA III	33%	4%	53%	10%	12%
LA IV	22%	12%	11%	55%	8%
AC I	7%	46%	41%	6%	6.2%
INGO I	7%	7%	30%	54%	4.9%
INGO II	6%	56%	6%	32%	3.7%

Note: LA: Local Authority, AC: Academia, INGO: International Non-Governmental Organization.

Table 3.12 Results of combined AHP for all participating organizations.

Constraints to water reuse in agriculture	Weights as percentage	Rank
Technical challenges	16%	4
Institutional challenges	24%	3
Social challenges	31%	1
Financial challenges	29%	2

3.6.4 Discussion

The results of the analytical hierarchy process show that social challenges were the main obstacle to treated wastewater reuse generally, and in agriculture specifically. The 31% overall perceived social challenges fell in line with local authorities I and III (at 71% and 53%, respectively), and to a lesser extent with academia I and international non-governmental organization I (at 41% and 30%, respectively). For LA I, the difference between perceived social challenges to the second ranked category (institutional challenges) was significant (54%), leaving a small impact for technical and financial constraints. A lack of a "Community Vision and Assessment" resulted in many projects to fail. LA I defined failure in communities, users or beneficiaries not owning the projects, because they were not part of the process in the first place.

A Community Engagement Scheme was established in partnership with an INGO, and it yielded significant results. LA III and INGO I confirmed that: "Now, most of [urban vegetable growers] are fetching water from the gutters". Wastewater in Ghana is generally considered as "just water", and therefore the public is less receptive to the idea of wastewater treatment and safe reuse, despite a recent Cholera outbreak (in 2014). IGNO I and AC I pointed to the scarcity of organic fertilizers and the high prices of chemical fertilizers encouraged large-scale farmers, particularly on the outskirts of Accra, to use raw wastewater as a substitute.

The agricultural sector, particularly farmers, will need to be convinced that treated wastewater and sludge can provide an attractive alternative resource for both irrigation water and, in some cases, imported unknown-source fertilizers, respectively, and that they can save money, besides resolving environmental, technical and institutional problems. Civil society organizations, such as environmental health and sanitation agency, in cooperation with concerned authorities, particularly the ministries of agriculture and environmental affairs, have to introduce programs directed to farmers so that they become aware of how to calculate the amount of nutrients required for their crops. This will help rationalize the amount of water used, thus protecting the environment and optimizing crop production.

Lack of skills and knowledge can cause failure in project implementation and, in the case of wastewater reuse projects, can potentially increase environmental and public health risks. Training programs should be an integral part of projects, and it should take into account technical, environmental, health and socio-economic aspects. The educational input must provide the farmers with an understanding of the details of the techniques and their associated hazards as well as any precautions. However, training the farmers to follow, at least visually or with simple tests, the quality of wastewater could be very helpful. International organizations including those interviewed can promote such ongoing programs with suitable technical and non-technical tools. Once the community has understood the need and importance of proper waste handling, it could be motivated to consider this as something valuable, and further utilize available treated wastewater in their private scheme.

LA II, LA III, and INGO I showed high correlation between social and financial factors (consistency ratios of 7.3%, 12%, and 4.9%, respectively). Farmers should be in a position to judge whether a particular wastewater has been properly treated or not. Changes in the colour of wastewater, and so is sludge, are indicators of the presence of high levels of chemicals and nutrients. Odour indicates insufficient treatment. In this area, institutions in Ghana introduced earlier training programs for agronomists and farmers. Market feasibility may refer to the ability to sell treated wastewater to crop producers, or it can refer to the marketability of produce cultivated with treated wastewater. The social acceptance of such produce must be assessed thoroughly. Generally, low-income consumers showed less willingness to pay for treated wastewater than high-income ones. Income is highly correlated with education and awareness. Low-income consumers often lack sufficient education, which is crucial in making the decision to reuse treated wastewater to achieve their food security.

Technical constraints received more impact from LA III than from all other participating organizations. However, the consistency ratio was 12% (higher than 10%), indicating that the interview resulted in judgement matrices to be slightly inconsistent. Social factors had the minimum impact for INGO II, which had the best consistency ratio (at 3.7%). This may be because this organization was involved in engineering wastewater treatment and reuse projects, funded by external donors. Interestingly, INGO II believed that institutional barriers were the main challenge (at 56%). The interviewee indicated that their organization's operations relied mainly on tripartite agreements, where whatever projects they implemented under any program were with external partners, mainly with local governments. He indicated that: "Ghana is fast urbanizing, so what is happening is that this situation [lack of wastewater treatment and reuse initiatives] is coming up and we have not planned the city [for wastewater treatment]".

In their efforts to overcome institutional barriers, INGO II used an Institutional Resource Mapping, so to learn from previous experiences, and establish connections with partners. Lack of certain regulations was noted, with specific emphasis on the polluter-pays principle. The perceived low social barriers (6%) was due to the belief that urbanization demystified many cultural perceptions, but acknowledged that guiding principles to preserve the environment were still required. The large number of institutions involved, and the complex system of wastewater treatment and reuse require the establishment of a sound institutional framework for coordination among the stakeholders. A reuse project should be a result of coordination between the stakeholders where each stakeholder has its part of shared responsibility. To overcome institutional barriers, the following framework could be adopted:

1. The Environmental Protection Agency (EPA) could be the leading institution, and it is to be responsible for wastewater collection, treatment, and disposal.
2. The Ministry of Health could be responsible for the regulation of the hygienic quality of wastewater reused for irrigation of marketed crops.
3. The Ministry of Food and Agriculture could be responsible for the implementation of the reuse projects, i.e. supply and development of irrigation schemes (pumping stations, reservoirs pipes, canals, etc.).
4. The Ghana Quality Organization (GQO) could be responsible for the preparation of environmental standards, guidelines, policies, and environmental impact assessment (EIA) legislation and enforcement. In addition, the GQO could be responsible for surveying and monitoring the environmental impact of collection systems, treatment and reuse of wastewater.
5. Master planning for water resources at the regional/national levels could be a shared responsibility between EPA, the National Development Planning Commission (NDPC), and the GQO.
6. Master planning of local areas would be at the local (municipal) level and could be a shared responsibility between local municipalities, Ministry of Local Government and Rural Development, and GQO in consultation with NDPC and GQO.

7. Ministry of Finance could represent the projects' counterpart cosigner, arrange project financing, and could be responsible for following up with the financial issues.

Unfortunately, and in general, the institutional arrangement in developing countries is very controversial and complex. Four organizations considered financial constraints to be the first or second major barrier to treated wastewater reuse. LA IV and INGO I ranked financial factors at the top (consistency ratios of 8% and 4.9%, respectively). However, there was significant difference in their perceptions of other factors. LA IV was the only organization to rank financial constraints at the top. It acknowledged that other water sources apart from rainfed agriculture were indeed required. Plans, however, were not in place mainly because the authority's strategic goals were directed to other priority schemes that were of interest to the public in their jurisdiction.

Because of the unique characteristics of wastewater, their impact on soils, crops, groundwater, humans and the integral environment should be regularly evaluated and carefully monitored, as indicated by AC I (consistency ratio 6.2%; social challenges 41%). Effective monitoring and record keeping are essential activities to ensure that sludge quality is compliant with standards and to identify any unacceptable environmental or health effects that may arise. Monitoring starts by checking the treatment plant for proving its proper functioning through some reliability tests. Monitoring will be extended at the farm level. There are several important quality parameters, including toxicity and pathogens, which should be regularly monitored.

The following Strengths, Weaknesses, Opportunities, and Threats (SWOT) can be drawn:

Strengths of treated wastewater reuse in agriculture are mainly adaptability and ability to incorporate new production methods and products, intense agricultural practices, existence of local markets, potential to export products, cooperative cultivation, and INGO presence and commitment to improve the livelihood of farmers and the productivity of the agricultural sector and water quality and quantity. The following list of facts may encourage farmers to use treated wastewater in agriculture:

1. Wastewater contains valuable nutrients and treated wastewater could be a competitive alternative to gutter water.
2. The desire to reduce environment pollution and consequent potential health risk using safe wastewater.
3. New environmentally-sustainable paradigms are compatible with the Ministry of Food and Agriculture trend and its strategies.

Weaknesses of the sector are: poor quality consciousness, lack of post-harvest processing activities, lack of developed marketing and distribution channels, lack of business management and marketing skills, lack of Research and Development, lack of testing and laboratory services, and poor access to financial services.

Opportunities for the water and agricultural sector are: favorable trade conditions in key export markets, strong markets, produce brand, traditional wet-farming is for the most part organic and can be developed into organic farming for the lucrative organic, new crops and varieties of existing crops can be developed for exports, and alternative marketing channels.

Threats are: willingness to pay for treated wastewater, fuel instability impacting export activities, regional competition and lack of compliance to technical requirements in export markets.

While each organization had several priorities on their current agenda which are in part based on successful and unsuccessful prior experiences, they all acknowledged that water quality and quantity are pressing issues that need to be tackled.

3.7 Part II: Water management

3.7.1 Research question

What informative evidence is there on the toxicological suitability of treated wastewater for reuse in agriculture?

3.7.2 Methodology

3.7.2.1 Site location and samples collection

Influent (raw sewage) and effluent (treated wastewater) samples were collected from the Legon Waste Stabilization Ponds in Accra (Figure 3.4). The selected wastewater treatment facility represents typical treatment plants in urban cities of a developing country, where the most common treatment process is anaerobic, facultative, and maturation ponds (Figure 3.5). The WSP treated domestic wastewater, with no significant industrial loading.

Figure 3.4 Side of the Legon Waste Stabilization Ponds in Accra (showing maturation ponds).

Composite samples were collected every three weeks between May and October 2016. Samples were stored in a cold room for 24 hours, and later kept in a fridge (at 4°C). All samples were collected in laboratory certified clean bottles and labeled with the name of the person who collected the sample and the method of collection, date and time of sample collection.

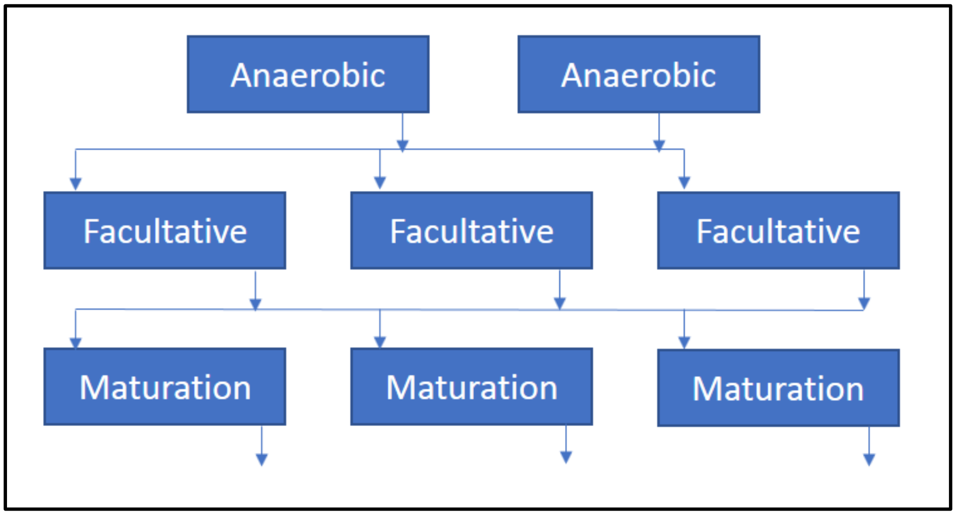

Figure 3.5 Schematic diagram of the Legon Waste Stabilization Ponds.

3.7.2.2 Laboratory work and wastewater analysis

The following parameters describing the performance and the characteristics of the WSP were determined: Ca, Na, Mg (to calculate SAR), pH, EC, B, and TN. An atomic absorption spectrophotometer (Perkin-Elmer, AAS 800 TM) was used to quantify Ca, Na, and Mg. The samples were diluted 5 times to ensure correct readings. Samples of the suspensions were withdrawn at different time intervals and immediately filtered through a 0.45 μm membrane filter. The filtrates were analyzed for various elements. Calculation was done as follows:

Amount of Mg (mg/L) = Readings x dilution factor

The sodium adsorption ratio is dimensionless and defined by the formula:

$$\mathrm{SAR} = \frac{\left[\mathrm{Na}^{+}\right]}{\left[0.5\left(\left[\mathrm{Ca}^{+}\right]+\left[\mathrm{Mg}^{2+}\right]\right)\right]^{0.5}}$$

where $[Na^+]$, $[Ca^{2+}]$ and $[Mg^{2+}]$ are the concentrations of sodium, calcium and magnesium ions in the irrigation water, expressed in milliequivalents per liter. For safe irrigation, the SAR should be less than 18 (Mara, 2004). Concentrations of $[Na^+]$, $[Ca^{2+}]$ and $[Mg^{2+}]$ ions in mg/L are converted to meq/L by multiplying by 0.044, 0.050 and 0.083 for sodium, calcium and magnesium, respectively.

With dilution, readings were multiplied by the dilution factor (e.g. if the AAS reading is 0.4 mg/L and dilution factor is 10, the concentration is 4 mg/L). Total Nitrogen was measured using 5 mL samples of wastewater. Samples were digested in concentrated H_2SO_4 with potassium sulphate catalyst, and a portable spectrophotometer was used (DR/2010, HACH, Loveland, USA), following the persulphate digestion method (HACH 1997). Samples were distilled with 40% sodium hydroxide and collected in 2% boric acid. Distillate was titrated against 0.01M HCl. Calculation was done as follows:

$$N\ (as\ percentage) = \frac{titer\ reading\ x\ 0.01\ x\ 0.014\ x\ V}{Volume\ of\ \ aliquot\ taken}$$

Where, 0.01 = molarity of the acid, 0.14 = a factor, and V = volume of digestate.

The digestion process is explained as follows:

- Degradation: Org-N + H_2SO_4 + catalyst + Heat (190ºC) → $(NH_4)_2SO_4$ + CO_2 + H_2O
- Distillation: NH_4^+ + $(NH_4)_2SO_4$ + 2NaOH → Na_2SO_4 + $2H_2O$ + $2NH_3$ (gas)
- Capture: H_3BO_3 + H_2O + NH_3 → NH_4+ $H_2BO_3^-$
- Back titration: NH_4 + $H_2BO_3^-$ + HCl → NH_4Cl

Aqueous boron was quantified using the potentiometric titration technique. $KMnO_4$ and EDTA were first added to the sample to break any organo-boron bonds, oxidize to borate, and chelate interferents such as aluminum and iron especially since they were employed for coagulation. D-mannitol was then added to produce an acid complex, which was then titrated using 0.02 N NaOH (ASTM, 1985; Sari and Chellam, 2015). EC and pH of each sample were measured using the electrometric method. HANNA pH meters were used, after calibration with buffers of pH 7 and 4, respectively.

3.7.2.3 Quality control

Analytical blanks and four samples with known concentrations of trace metals and anions were prepared and analyzed using the same procedures and reagents. Standard reference materials (SRM) from the National Institute of Standards and Technology of the USA (NIST: 1643c and 1643d) were used for the determination of ions and boron (NIST, 2010). The results of the analyses were reviewed in terms of milli-equivalent balance, which compares ionic charges of major anions and cations (APHA, 2005).

3.7.2.4 Data analysis

The mean and respective errors of metal and ion concentrations in wastewater samples were calculated for each parameter (i.e., for each set of three replicates, one from each parameter). To evaluate statistically significant differences, students' t-test ($p<0.05$) was performed.

3.7.3 Results and discussion

3.7.3.1 pH

The alkalinity of effluents fell in the acceptable range of irrigation wastewater (6.5 to 8.4; Figure 3.6). Treated effluents generally featured higher pH values than influents. Algae consume CO_2 through photosynthesis thereby causing an increase in pH in the pond. In anaerobic and anoxic processes, organic nitrogen is used as a food source, which causes a significant increase in alkalinity.

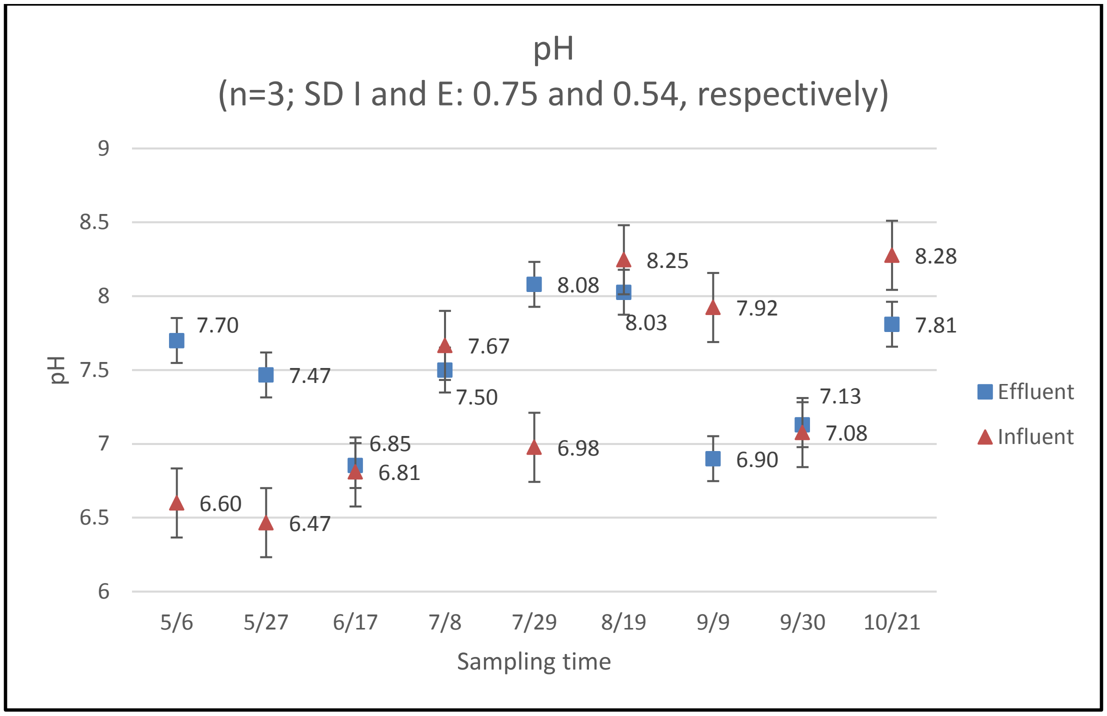

Figure 3.6 Measured pH levels in the influent and effluent of the Legon WSPs between May and October 2016.

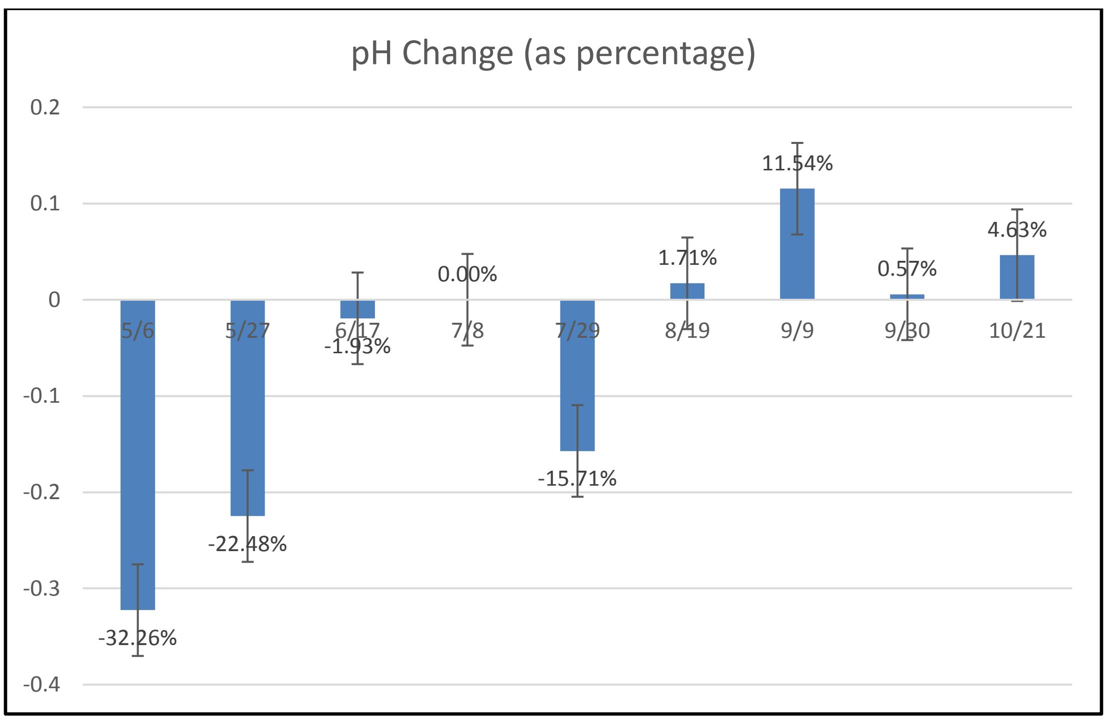

Figure 3.7 Change of pH levels between influents and effluents.

The levels of pH increase as ammonia nitrogen decreases. This pH increase is common in similar treatment systems, such as the WSPs in Loyalist Township, Ontario. However, aerobic processes (i.e. nitrification) result in a decrease in alkalinity. Because the wastewater treatment system depends on natural processes, seasonal and even diurnal differences (day verses night) can affect microbial photosynthesis and respiration, water temperature and water movement, which can subsequently produce significant swings in effluent characteristics including pH levels. Treatment occurs best under alkaline conditions, with optimum pH range being typically between 6.5 and 10.5.

The pH of the irrigation water and consequently the application of wastewater with a low pH could lead to decreased soil pH, and this in turn could cause an increase in the mobility of heavy metal which would then become available for plant uptake or leaching to lower soil layers. Soil pH has great influence on the mobility and bioavailability of heavy metals and, in general, metals are more available to plants from acidic soils than from neutral or alkaline soils.

The efficiency of chemical precipitation of metals can be affected by pH and the presence of other ions. However, it is ineffective when metal concentration is very low. When the capacity of soil to retain heavy metals is reduced, as a result of continuous application of wastewater or a change in soil pH, the metals enter a mobile phase, and may be released to groundwater or be available for plant uptake.

3.7.3.2 Electrical conductivity

The average electrical conductivity of treated effluent was 62 mS/m, less than 70 mS/m (Figure 3.8). The average electrical conductivity improvement was 24% (with an average influent EC concentration of 82 mS/m). The highest ion removal difference was recorded in early September at +42% (with effluent EC at 59 mS/m), and the lowest was in mid-June at +9% (with effluent EC at 73 mS/m). Therefore, treated effluent is non-saline, and salinity levels is suitable for all crops as indicated in Table 3.2. The hydraulic retention time in combination with climatic conditions directly affect salinity of the effluent. The loss of water via evaporation from facultative and maturation ponds is less than from anaerobic ponds, and, combined with the rainy weather especially in September and

October, this maintained lower salinity levels. However, low salinity was associated with the presence of emergent grass and *Culex* mosquitoes and *An. subpictus*.

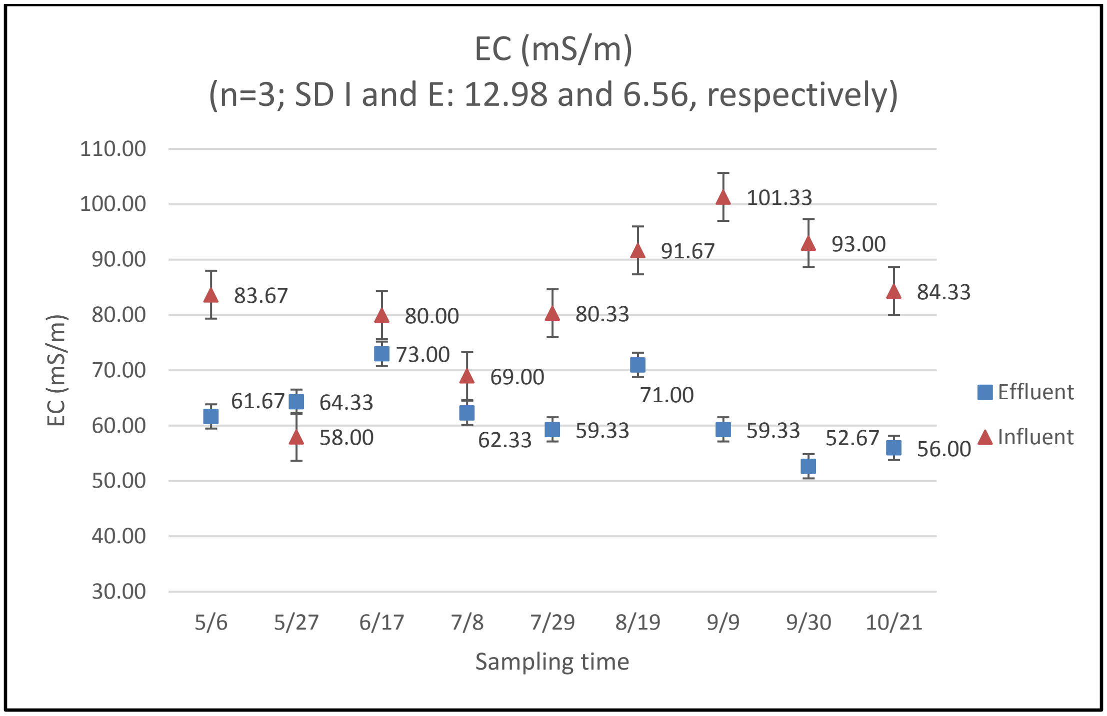

Figure 3.8 Measured EC levels in the influent and effluent of the Legon WSPs between May and October 2016.

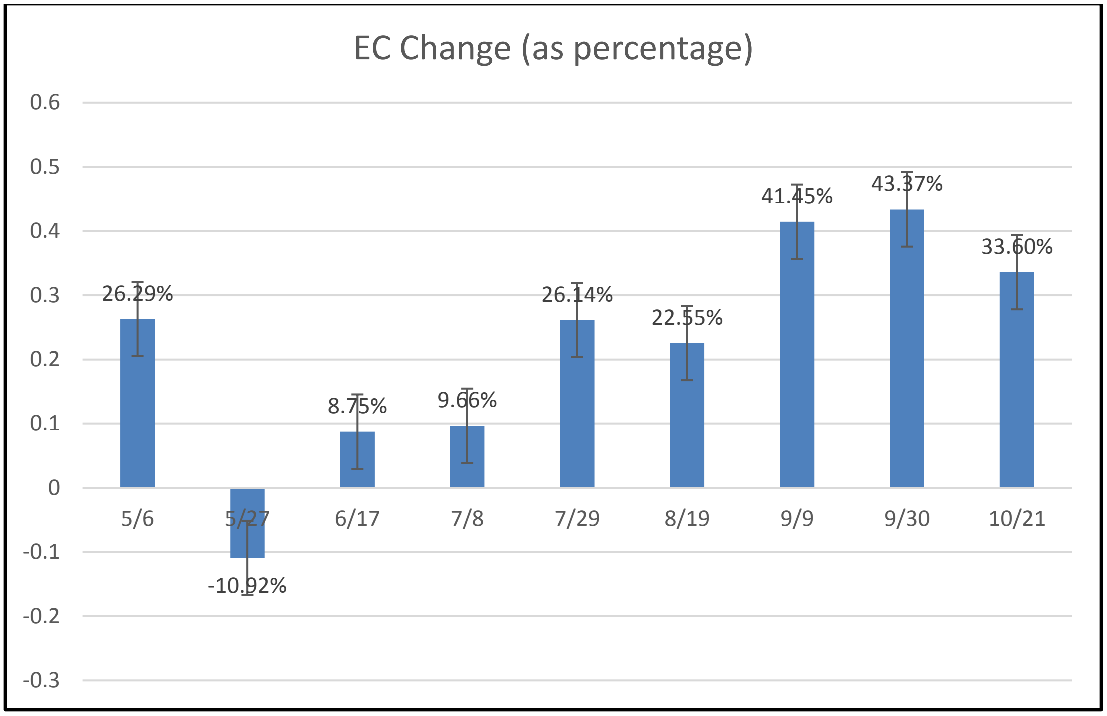

Figure 3.9 Change of EC levels between influents and effluents.

As noted in Section 3.5, groundwater exhibited high salinity levels (280 mS/m was recorded). Recharging groundwater with treated wastewater via application on agricultural land has strong potential to alleviate the issue. The suitability of reusing wastewater for irrigation is not only affected by the quantity of dissolved salts but also by the type of salts that are present. Generally, wastewater can be classified into saline wastewater that contains excess levels of soluble salts and total dissolved solids, sodic wastewater containing excess sodium Na^+, and saline-sodic wastewater which is characterized by both excessive salt and sodium Na^+. The type and the degree of the effects will vary depending on the type of wastewater being reused.

Soil structure could be negatively impacted by saline treated wastewater irrigation. Salinity and sodicity-related characteristics and impacts are affected by many factors including: the type of salt, the efficiency of leaching and the drainage system, irrigation system type, sensitivity of crops, and soil properties. Significant long-term problems of soil salinity and/or salinity due to the application of saline irrigation water results primarily from poor irrigation management and inadequate soil drainage systems. The type of irrigation system used directly affects both the efficiency of water use and the way salts accumulate. Each irrigation technique has certain advantages and disadvantages, and these should be considered if treated effluent had salinity levels above the recommended value.

3.7.3.3 SAR

The maximum SAR was found in raw wastewater at 6.65, which is significantly less than 18 (Figures 3.10 to 3.17). Therefore, treated effluent SAR is suitable for irrigation of all crops. The average influent SAR was 5.2, and the average effluent SAR was 4.78. Maximum SAR difference was in late October at 34% (after a heavy rainfall), and the least was in late May at only 0.8% (during the dry season). At a given SAR, the infiltration rate increases as salinity increases. Therefore, SAR and TDS should be used in combination to evaluate the potential permeability problem (Pedrero *et al*., 2010).

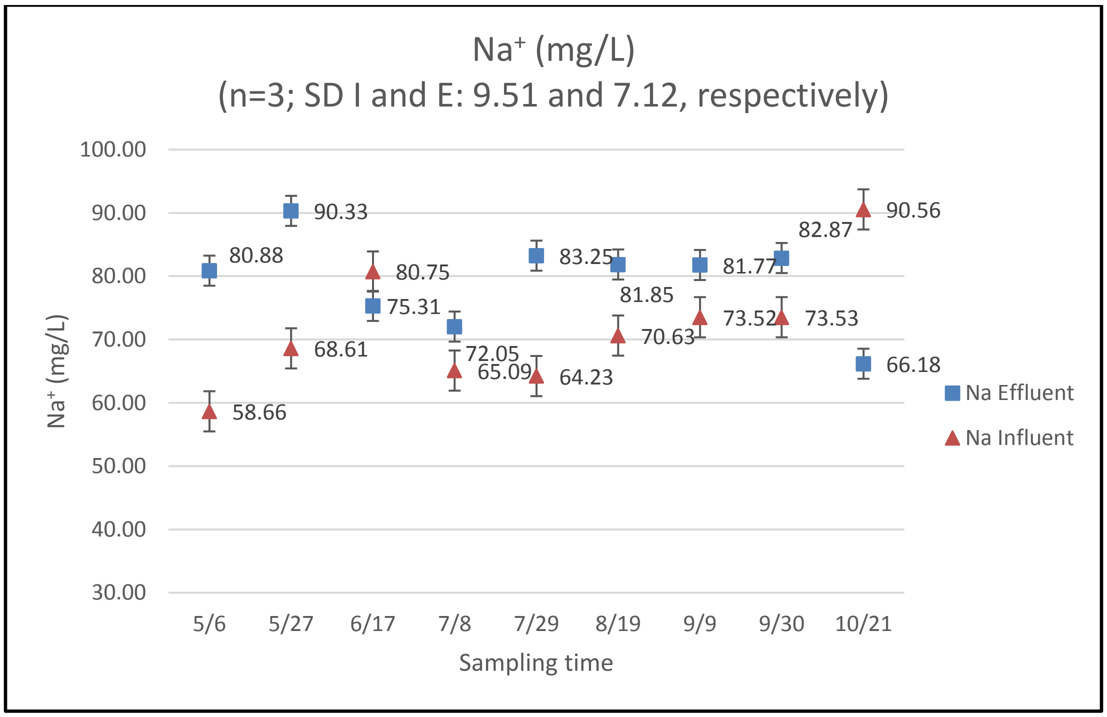

Figure 3.10 Measured Na^+ concentrations in the influent and effluent of the Legon WSPs between May and October 2016.

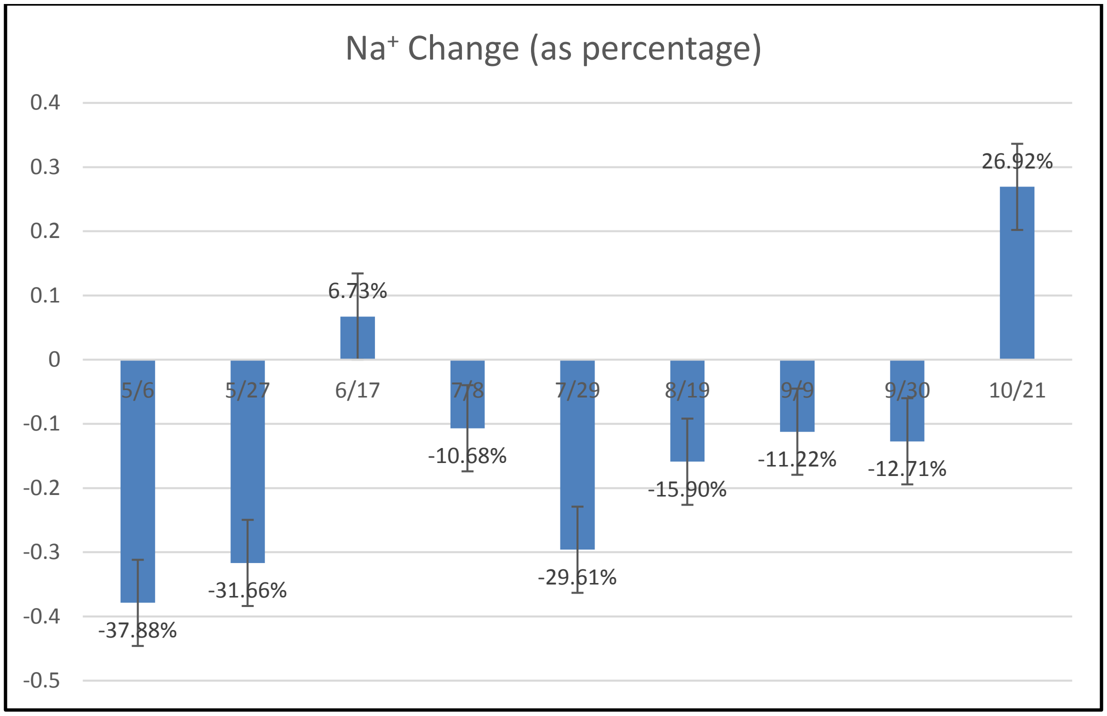

Figure 3.11 Change of Na^{+} levels between influents and effluents.

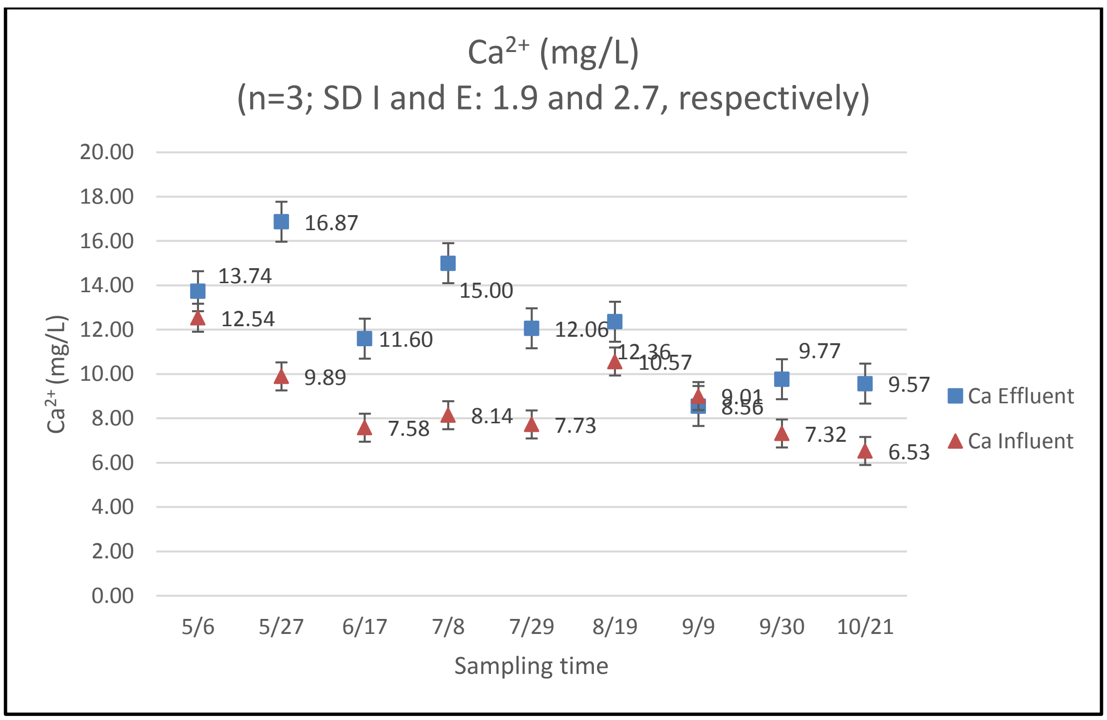

Figure 3.12 Measured Ca^{2+} concentrations in the influent and effluent of the Legon WSPs between May and October 2016.

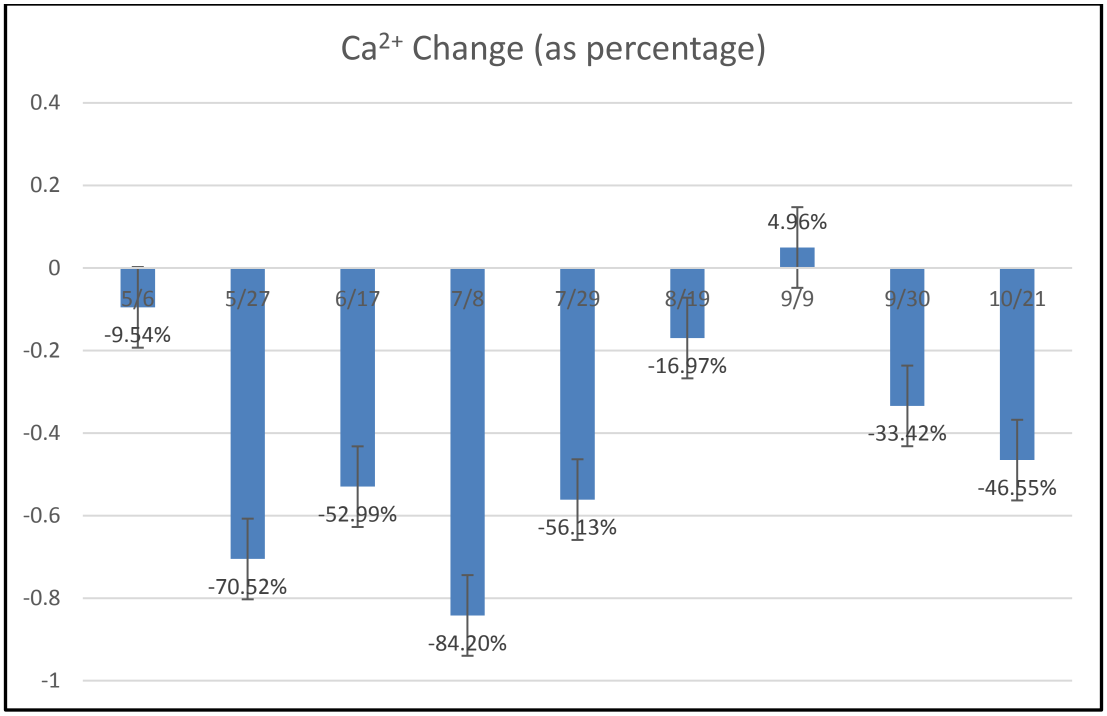

Figure 3.13 Change of Ca^{2+} levels between influents and effluents.

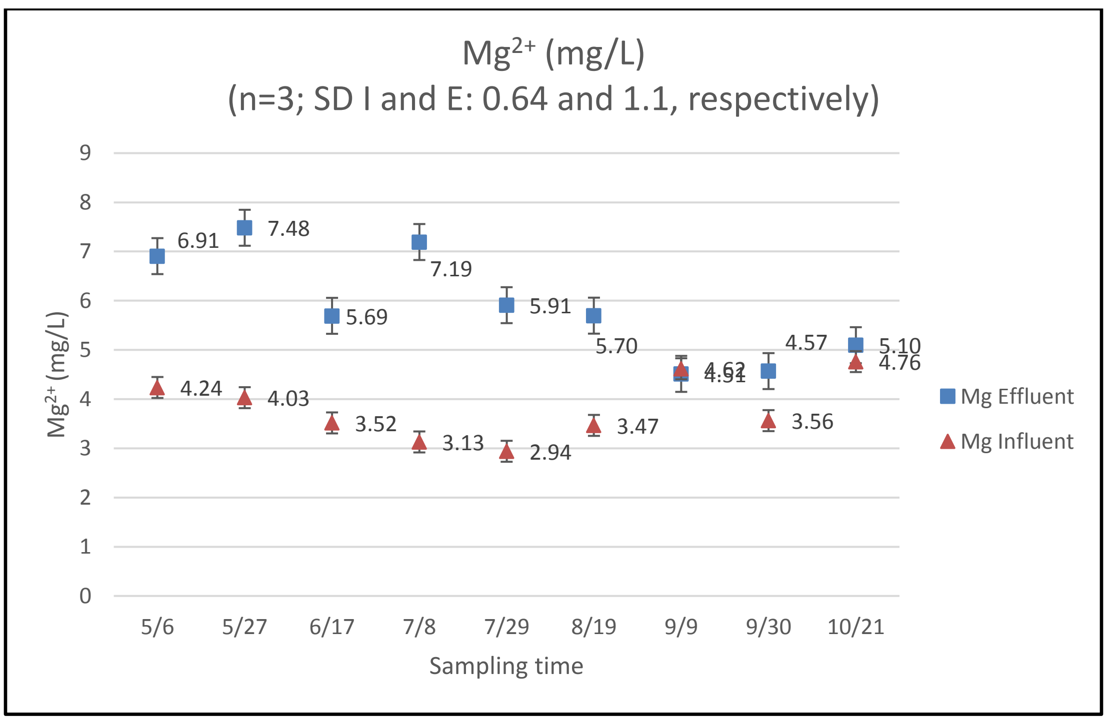

Figure 3.14 Measured Mg^{2+} concentrations in the influent and effluent of the Legon WSPs between May and October 2016.

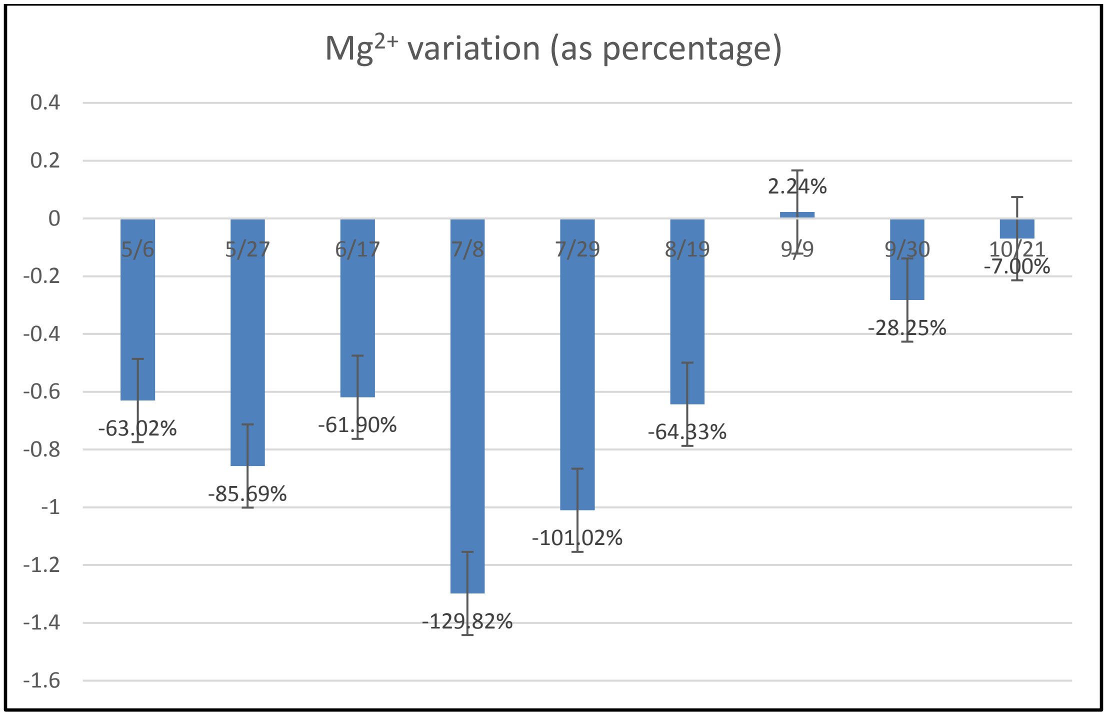

Figure 3.15 Change of Mg^{2+} levels between influents and effluents.

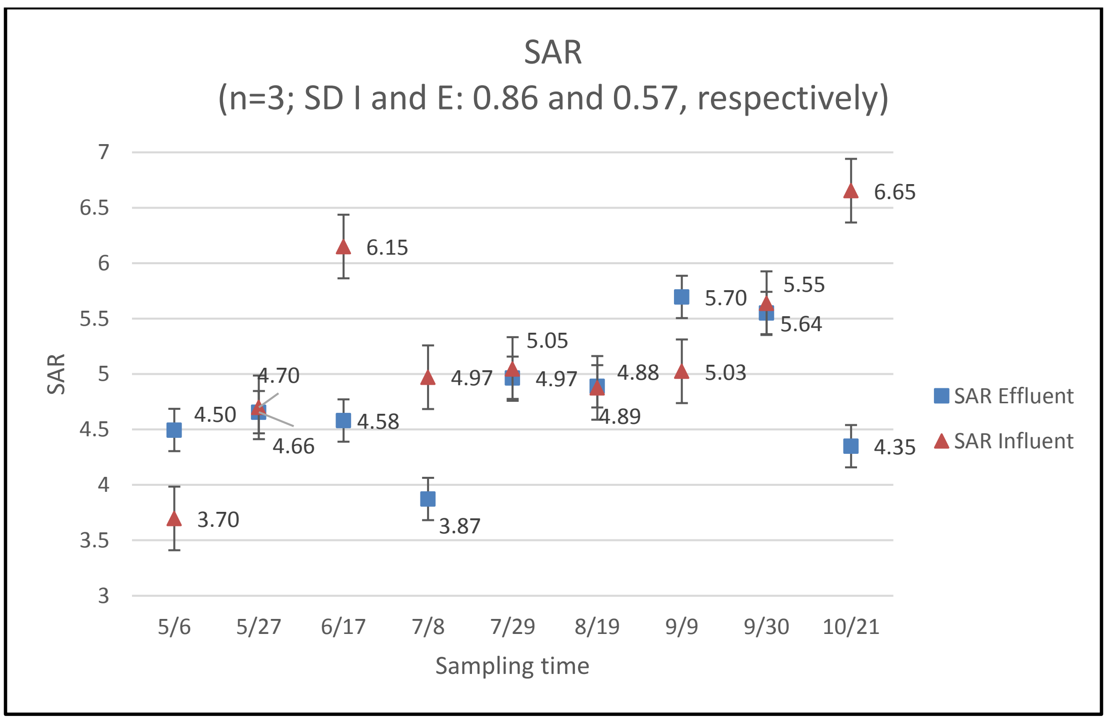

Figure 3.16 Calculated SAR levels in the influent and effluent of the Legon WSPs between May and October 2016.

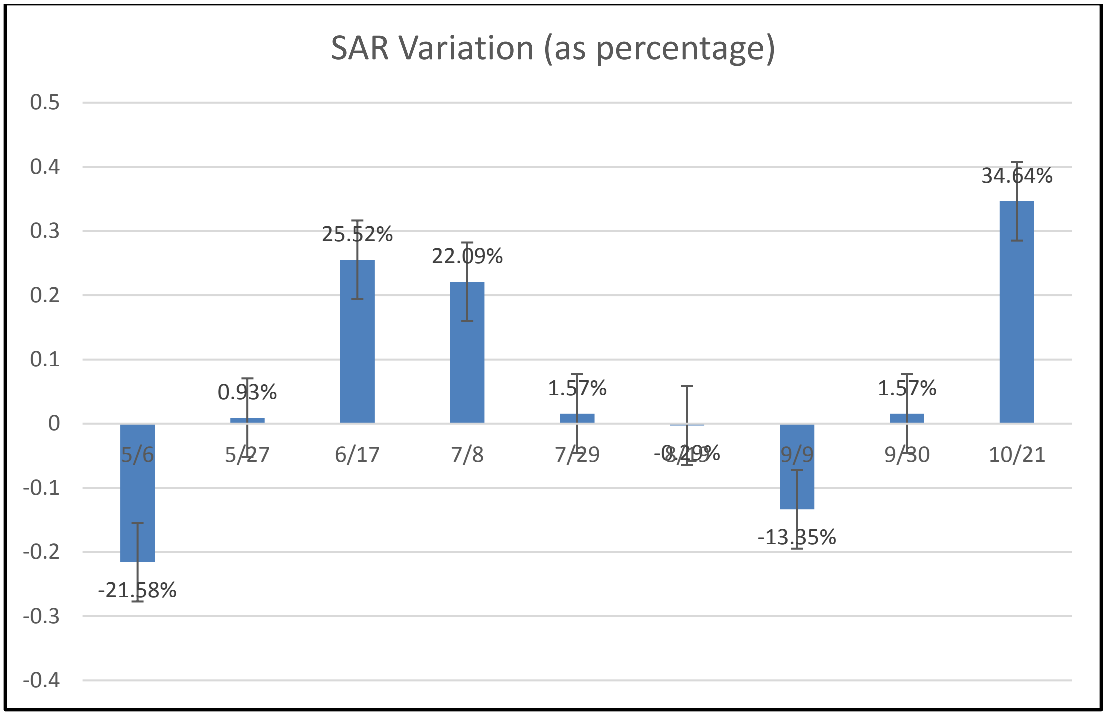

Figure 3.17 Change of SAR levels between influents and effluents.

Agriculture and Agri-Food Canada however sets SAR limit to 9, with slight to moderate degree of suitability of wastewater reuse for SAR being between 4 and 9. Na^+ concentrations above 70 mg/L are considered slight to moderate, with no bottom limit to severity. Due to its similar physicochemical properties, Na^+ competes with K^+ in plant uptake specifically through high-affinity potassium transporters (HKTs) and nonselective cation channels (NSCCs). In this case, tested effluents are considered safe for reuse in agriculture (Figure 3.12).

Sodium in water can displace calcium and magnesium in soil. This will cause a decrease in the ability of the soil to form stable aggregates and a loss of soil structure and tilth. Excessive exchangeable Na^+ concentrations relative to magnesium and calcium leads to sodicity problems which can cause deterioration of soil structure, clay dispersion with subsequent blocking of pores, negative effects of hydraulic properties such as causing soil impermeability. It may lead to elevated pH of the soil solution, and it could dissolve humus and sodium humate precipitates which imparts the dark alkali soils.

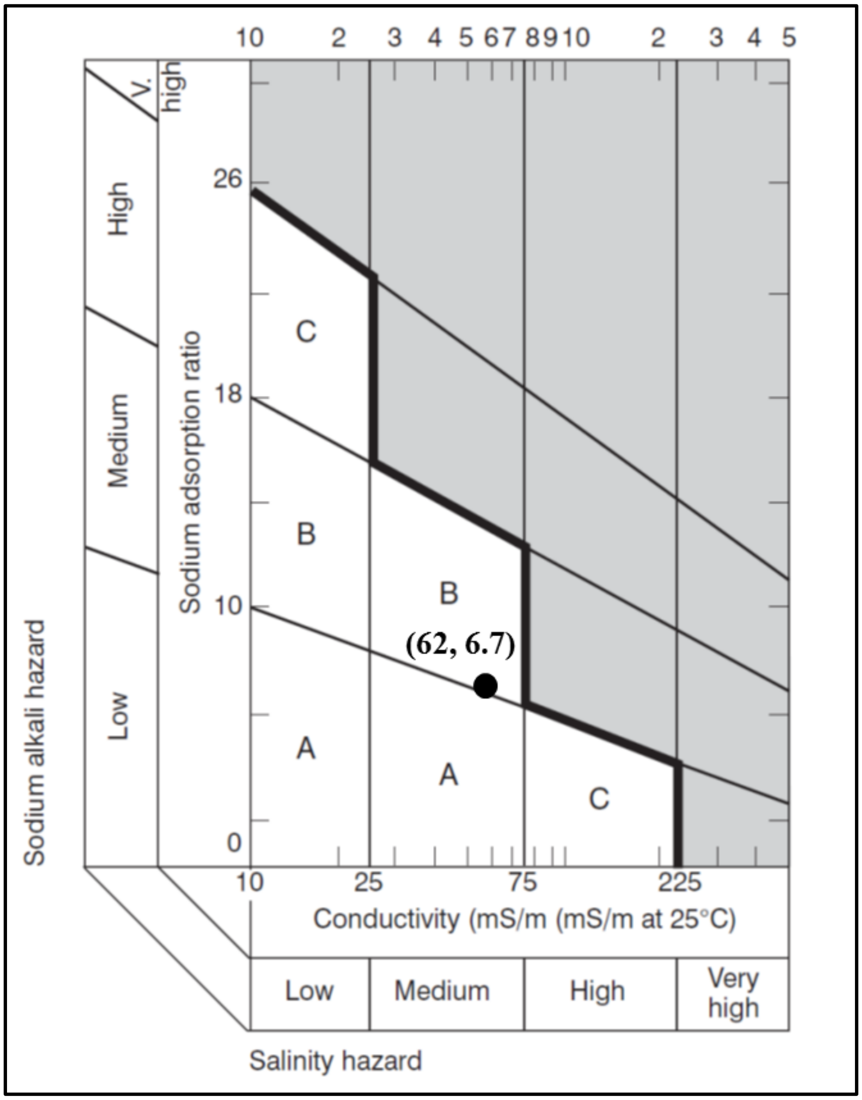

Figure 3.18 Waters in regions A and B are acceptable for almost all irrigation purposes. Adopted from Mara (2004).

3.7.3.4 Boron

The average boron concentration in the effluent was 1.1 mg/L (Figure 3.19). The average concentration in influents was 2.8 mg/L. The maximum removal efficiency of 84% was in late October. The lowest removal efficiency was in early July at 49%. High removal efficiency of boron may be attributed to boron-accumulator plants, especially with low salinity waters. Marin and Oron (2007) found that wetland species *Lemna gibba* removed boron efficiently. Selection of adequate wetland plant species for trace element removal, however, is also based on the known accumulation capacities of the different species (Zayed *et al*., 1998).

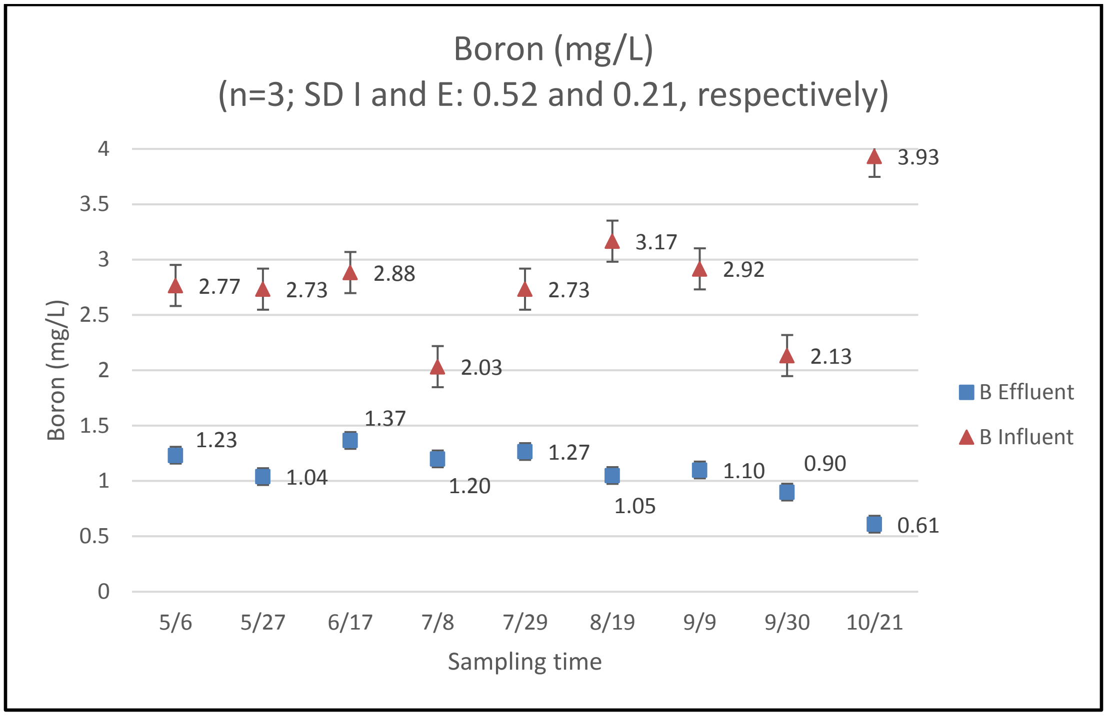

Figure 3.19 Measured B levels in the influent and effluent of the Legon WSPs between May and October 2016.

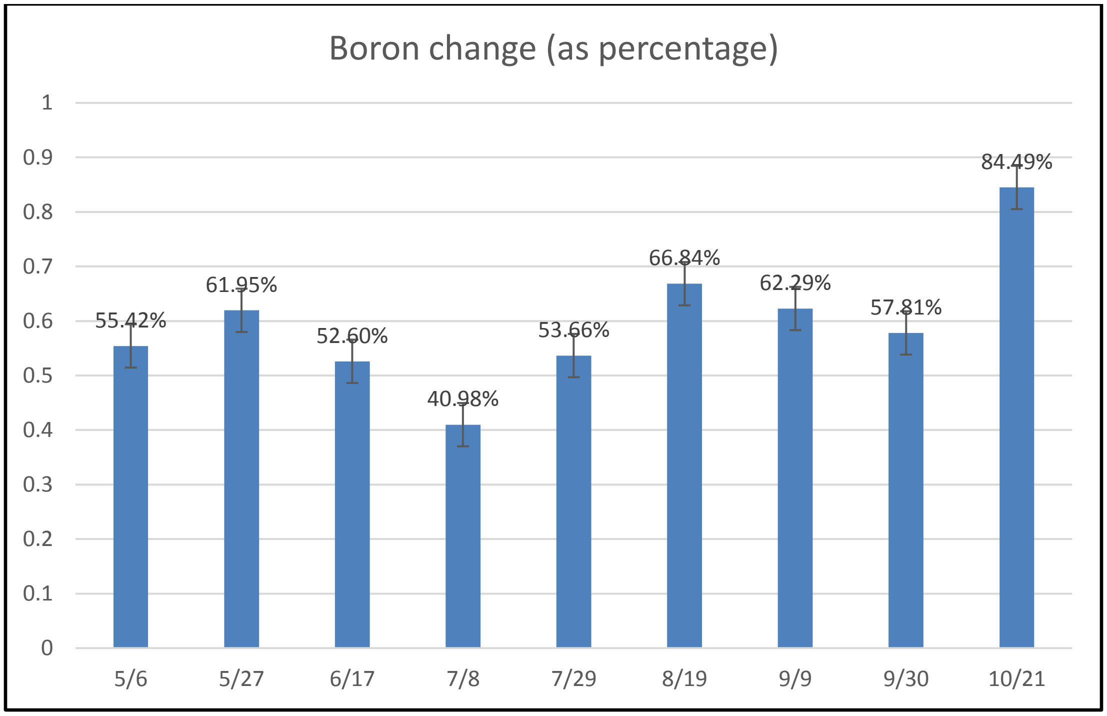

Figure 3.20 Change of B levels between influents and effluents.

Boron concentrations of more than 2 mg/L have adverse effects on most crops and plants. Concentrations are generally low in this study because there were no significant industrial discharges, and boron which was present in flows was mainly from domestic detergents. With the maximum effluent boron concentration being 1.37 mg/L, the following crops can be irrigated: sweet pepper, pea, carrot, radish, potato, cucumber, squash lettuce, cabbage, celery, turnip, oats, maize, sorghum, tomato, alfalfa, parsley, beetroot, sugar beet, and cotton (Table 3.13).

Table 3.13 Recommended maximum concentrations of boron in irrigation waters according to crop tolerance.

Boron concentration (mg/L)	**Crops**
< 0.5	Lemon, blackberry
0.50–0.75	Avocado, grapefruit, orange, apricot, peach, cherry, plum, fig, grape, walnut, pecan, cowpea, onion
0.75–1	Garlic, sweet potato, wheat, barley, sunflower, beans, strawberry, peanut

1–2	Sweet pepper, pea, carrot, radish, potato, cucumber
2–4	Lettuce, cabbage, celery, turnip, oats, maize, squash, barley, cauliflower
>4	Sorghum, parsley, tomato, alfalfa, asparagus, parsley, beetroot, sugar beet, cotton

Source: Mara (2004) and FAO (2007).

3.7.3.5 Total nitrogen

The maximum total nitrogen in tested effluents was 16 mg/L, compared to a minimum influent concentration of 97 mg/L (Figure 3.21). The maximum and minimum nitrogen removal efficiency occurred in mid-August and late July at 90% and 82%, respectively. The average concentrations in influents and effluents were 121 and 15 mg/L, respectively. Performance of the ponds in removing nitrogen matched conventional activated sludge treatment systems which has an average effluent concentration of 15 to 35 mg TN/L.

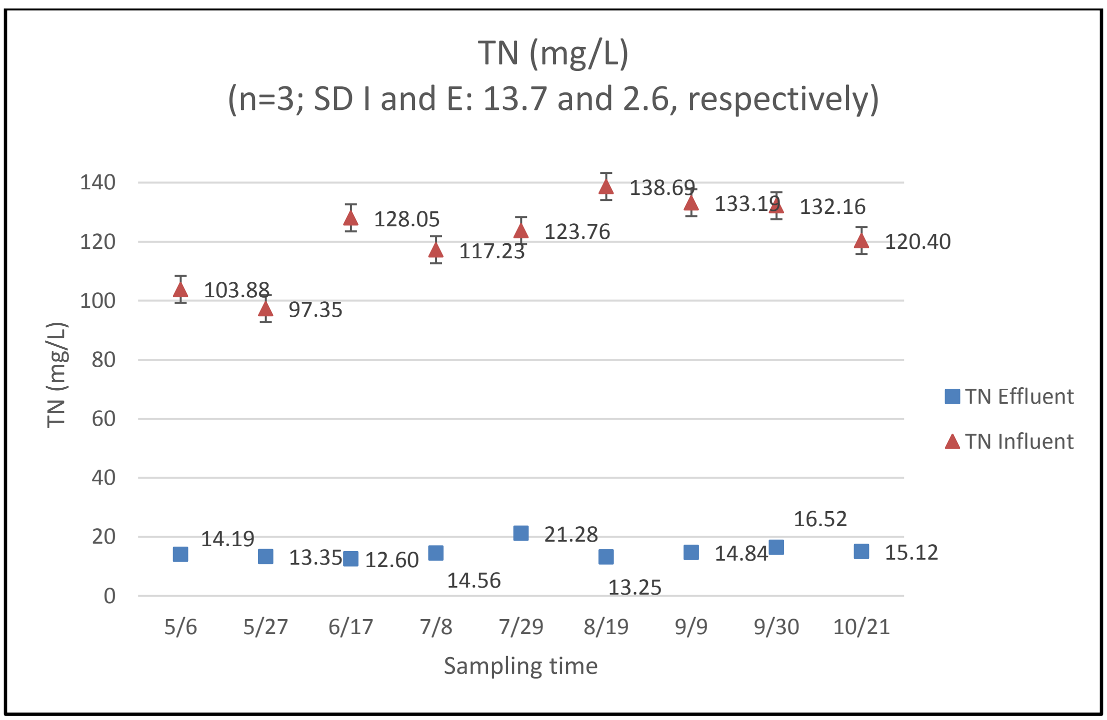

Figure 3.21 Average measured TN levels in the influent and effluent of the Legon WSPs between May and October 2016.

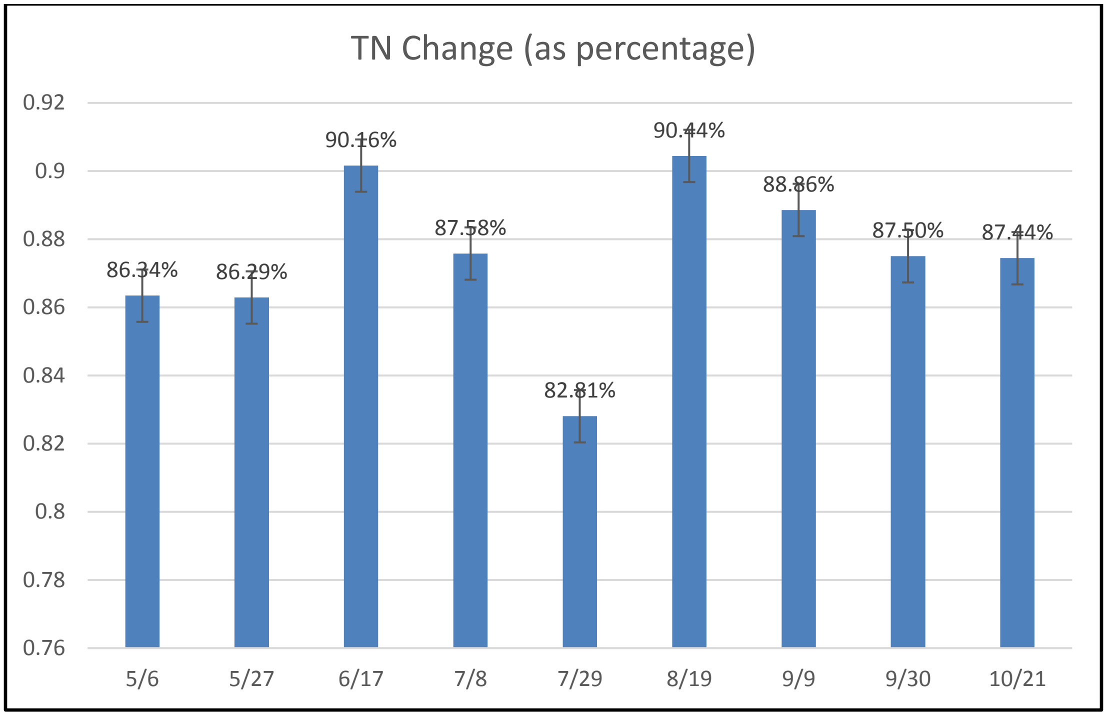

Figure 3.22 Change of TN levels between influents and effluents.

Nitrogen removal occurs only in facultative and maturation ponds through the incorporation of ammonia into algal cells. In anaerobic ponds, only nitrogen transformation with some of the organic nitrogen (principally urea and amino acids) is converted to free and saline ammonia. Fish was present in maturation ponds, indicating free ammonia concentrations of less than 0.5 mg NH_3-N/L. Low total nitrogen indicates high dissolved oxygen, which contributes to fish and plant growth. This is consistent with higher alkalinity levels (Section 3.7.3.1).

It is anticipated that most crops can tolerate 30 mg TN/L, but some only 5 mg TN/L (Mara, 2004). However, plant species tolerance through quantification and categorization has not been found in literature. Given the fertilizer value of treated wastewater, nitrogen levels may not need to be reduced any further. However, excessive nitrogen in irrigation water may impact soil microbial communities, in particular the microbial activities associated with cycling this element.

The amount of nitrogen taken up by the plant, leached to groundwater, or lost via soil erosion and volatilization depends on the nitrogen concentration in the effluent and the type of soil, crop demand, soil permeability, irrigation rate and the vulnerability of the aquifer. Nitrogen supplied via irrigation is removed primarily through nitrification and subsequent ready uptake by plants as ammonium NH_4^+-N and nitrate NO_3-N. The concentration of NH_4^+ in treated wastewater is normally greater than nitrate and it usually binds to soil particles and is not leached. However, it can easily be converted to nitrate nitrification by soil bacteria. Nitrates are highly dissolved in the soil solution and they can easily be moved through wastewater irrigated soils especially highly permeable soils.

3.8 Conclusions and recommendations

1. Treated wastewater contains valuable nutrients necessary for plant growth. Ghana is currently water vulnerable, and at current trends, it will become a water-stressed country in the near future. The deficit can be completely substituted by reusing locally-generated treated effluent.
2. Treated effluent generated from the Legon waste stabilization pond is toxicologically suitable for reuse in agriculture. This applies to almost all crops.
3. Substituting treated wastewater for potable water supplies in agriculture is a competitive sustainable option to conserve the environment, protect public health and drive economic development.
4. Treated wastewater application on land is multi-dimensional. Reusing treated wastewater in agriculture is technically feasible and environmentally friendly. However, social and financial challenges exist.
5. Community engagement schemes proved effective. Partnerships with local governments may likely have resulted in greater institutional barriers.
6. The complex system of wastewater treatment and reuse require the establishment of a sound institutional framework for coordination among the stakeholders. The proposed institutional framework could be adopted.
7. Fertigation could make substantial contributions beyond the provision of livelihoods and food. However, there are strengths, weaknesses, opportunities, and threats to treated wastewater reuse in agriculture.
8. A multiple-barrier approach can be adopted, and safe reuse practices can be developed through action research involving a number of stakeholders at different levels along the food chain.

9. The pollution expected from wastewater reuse would be considerable when irrigation efficiency is low. Even effluents of acceptable quality could adversely affect health and environment if not applied properly. Proper management of treated wastewater application should be taken into account.
10. pH was within the normal range of treated effluents, whereas EC was significantly lower than average concentrations in effluents, possibly due to the rainy season. Alkalinity and salinity met requirements for reuse in agriculture. Rainfall likely resulted in swings of SAR and pH levels in effluents.
11. SAR threshold was different for Canada and FAO. However, the ponds met both limits.
12. For total nitrogen, performance of the studied ponds in removing nitrogen matched conventional activated sludge treatment systems.
13. Knowing the quantities of treated wastewater applied on land, the amount of nutrients can be calculated to determine how much fertilizers are needed for a particular crop. This could save on fertilizers for farmers and consequently driving their market.
14. "Little drops of water, little grains of sand, make the mighty ocean and the pleasant land". Rational use of treated wastewater ensures public health and crop health protection.

3.9 Future work

1. Utilization of the Analytical Hierarchy Process for social dimensions (as the main criterion) of treated wastewater reuse in agriculture.
2. Life cycle assessment to determine externality costs of the Golden Rule.
3. Assessment of the microbiological quality of treated wastewater.

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Chapter 4

Biochar-Modified Soil Aquifer Treatment for Water Reuse in Agriculture

4.1 Introduction

Canadians are worried their bucolic backyard is fast becoming their outhouse, and are concerned that septic systems may be leaking into groundwater (The Globe and Mail, 2013). On-site septic treatment of sanitary waste is proliferating throughout the Great Lakes basin as they serve more than 50% of new housing in some areas, with effluent filters, required on newly installed septic systems, are not common on existing systems (IJC, 2010; GCC, 2015). Moreover, at least 40% of existing systems fail to treat wastes adequately (GCC, 2015). Leaky sewer and waterlines are of concern with 30% conveyance loss being common, and thousands of line breaks occur in the basin every year. Few jurisdictions monitor or regulate these systems in any systematic way.

One of the motivations for the present study is to investigate the suitability of a biochar-modified SAT for wastewater treatment in rural areas of Canada where on-site sanitation systems (OSSs), known as septic tanks, are used. Many rural communities rely on aging individual septic systems or drain tile networks that discharge sewage directly to surface waters, even though direct discharge of untreated sewage is illegal (IJC, 2010). In Ontario alone, there are an estimated 1.2 million septic systems potentially posing a threat to public health and the environment (AMO, 2008). Approximately 25,000 new or replacement OSSs are installed annually in Ontario with similar numbers installed in each of the Great Lakes states each year (IJC, 2010).

The Soil Aquifer Treatment (SAT) system is defined as a three-stage wastewater treatment process involving infiltration zone, vadose zone, and aquifer storage. Removal of pollutants occurs via physical, chemical and biological processes in the unsaturated and saturated zones (Abel, 2014). SAT is a relatively low-cost system, and can be an alternative to fresh water sources (van Bruggen, 2014) for agricultural and park irrigation and municipal uses. SAT systems have been found to be capable of (Fox *et al*., 2006):

- facilitating the removal of organic carbon, nitrogen, and pathogens;
- providing anaerobic ammonia oxidation environment; and

- being resilient to pathogens and pathogen indicators.

Sharma and Kennedy (2017) identified the following main advantages of SAT: (i) SAT improves the physical, chemical and microbial quality of source water during soil passage by removing particles, microorganisms, heavy metals, nitrogen, bulk organic matter and organic micropollutants, (ii) it can be integrated with other conventional and advanced wastewater treatment systems to produce water of a desired quality for intended use, and (iii) it can serve as environmental and psychological barrier, thus increasing public acceptability of reclaimed water and promoting water recycling and reuse. However, local hydrogeological conditions may affect SAT permeability.

Biochar is used for soil application because of its capacity to improve soil organic content, and it was reported to have increased crop yield. Biochar is also credited for its water holding capacity, thus improving soil structure. Other applications include bio-energy to reduce dependency on fossil fuel, while offsetting carbon emissions which contributes to climate change strategic plans. However, literature lacks information on the use of biochar in commercial wastewater treatment, which could indicate lack of its assessment and or application in the removal of contaminants from wastewater, so it can be considered for application on land.

Latest wood waste survey conducted in 2004 concluded that almost 1 million tonnes are being disposed of (excluding that reused for bioenergy and recycling) yearly in Canada, with a rate of 120,000 tonnes per year for Ontario alone (Kelleher, 2007). Biochar, produced from the pyrolysis of agricultural waste and forest industry by-products as feedstock, can be used in several applications, including: (1) soil amendment (Hunt *et al*., 2010), (2) carbon sequestration (Lehmann and Joseph, 2015), (3) pollution prevention through better management of agricultural waste and run-off control, and (4) reuse potential as energy source.

4.1.1 Aim and objectives

The aim of this study is to investigate a Biochar-Modified SAT (BMSAT) system for the treatment of wastewater for reuse on land, as per the 2012 US Environmental Protection Agency (EPA) guidelines. The objectives are to: (a) critically discuss post-treatment effluent toxicity against the EPA criteria guidelines for reuse in agriculture, and (b) evaluate the microbiological quality of treated effluent against these guidelines.

4.2 Nutrients required for plant growth

Plant nutrients are divided into three subgroups (Table 4.1). All fourteen elements are essential for crop growth, yield and the quality of the crop produce (MOAFF, 2000). However, balanced nutrition is an important aspect (Laegreid *et al*., 1999; Magen, 2006).

Table 4.1 The essential elements for plant growth.

Element	Taken Up as	Function
Primary nutrients		
Nitrogen	NO_3^-, NH_4^+	Structural component of proteins, DNA, enzymes, etc.
Phosphorus	$H_2PO_4^-$, HPO_4^{2-}	Structural component of DNA; involved in energy conversion
Potassium	K^+	Essential for many chemical reactions in plants
Secondary nutrients		
Sulfur	SO_4^{2-}	Structural component of some proteins
Magnesium	Mg^{2+}	Central component of chlorophyll
Calcium	Ca^{2+}	Influences permeability of cell membranes
Micronutrients (Trace elements)		
Iron	Fe^{2+}	Structural component of a number of essential enzymes
Manganese	Mn^{2+}	Involved in enzymes for respiration
Boron	H_3BO_3	Required for protein synthesis
Chlorine	Cl^-	Involved in carbohydrate metabolism
Zinc	Zn^{2+}	Component of enzyme for decomposition of carbonic acid
Copper	Cu^{2+}	Component of enzymes for oxidation reactions
Molybdenum	MoO_4^{2-}	Component of enzyme that reduces nitrate to nitrite
Nickel	Ni^{2+}	Plant cannot complete the lifecycle without Nickel

Sources: Laegreid *et al*., 1999 and Shakhashiri, 2011.

Nitrogen has a larger effect on crop growth, yield and quality than any other nutrient (MOAFF, 2000). However, too much nitrogen will be financially wasteful and can aggravate problems such as lodging of cereals (the permanent displacement of plant stems from the vertical which affects all cereal species and is a major limiting factor on grain production worldwide), foliar diseases and poor silage fermentation (CEH, 2015).

Phosphorus is an essential element for the metabolism of living organisms because it is a component of nucleic acids, the phospholipids that compose cellular membranes, ATP and ADP molecules and intermediate compounds of respiration and photosynthesis (Taiz and Zeiger, 1998). Particular care should be taken to avoid the buildup of unnecessary high phosphorus in soil causing environmental problems.

4.3 Study site

The Vauxhall Wastewater Treatment Plant (WWTP), London, Ontario, Canada was the site of this study (Figure 4.1). The plant is an activated sludge treatment plant with two sections that treat municipal and industrial flows. In 2016, the plant experienced 10 overflow events with a total volume of 6,000 m^3 of raw sewage and 2,000 m^3 of primary treated sewage bypassing to the Thames River. Based on a desktop review, the lands immediately surrounding the plant, including the Thames River, potentially include habitat for species protected under the Provincial Endangered Species Act (2007). The annual average daily flow is 13,000 m^3/day. More than 16,000 m^3 of sludge were incinerated or disposed of at a local landfill. Table 4.2 summarizes wastewater characteristics of the plant.

The monthly average concentrations and loading compliance criteria for BOD_5, *E. coli*, ammonia, phosphorous and suspended solids were achieved in 2016. Effluent pH was between 6.7 and 7.8 which were within the compliance range of 6.0 to 9.5. A treatment optimization of the Vauxhall sewersheds was studied through a municipal class environmental assessment master plan. Interconnection of the sewersheds was found to provide additional operational flexibility and facilitate future upgrades as the assets approach the end of their service life. The Upper Thames River Source Protection Area Assessment Report identifies that this project site is located within a Highly Vulnerable Aquifer Area, and is within a Significant Groundwater Recharge Area (City of London, 2017).

Figure 4.1 Arial view of the Vauxhall WWTP. Source: City of London (2017).

Table 4.2 Wastewater characteristics of Vauxhall WWTP in 2016.

Parameter	Unit	Average	Minimum	Maximum
Temperature	°C	15.1	10.7	20.8
BOD_5 raw	mg/L	228	161	354
BOD_5 effluent	mg/L	17	12	26
Suspended solids raw	mg/L	354	210	715
Suspended solids effluent	mg/L	3	1	7
Total phosphorus raw	mg/L	6.3	3.5	9.5
Total phosphorus effluent	mg/L	0.25	0.17	0.3
E. coli effluent*	CFU/100mL	32	8	51
Total Kjeldahl Nitrogen raw	mg/L	28	13	57
Total Kjeldahl Nitrogen effluent	mg/L	1.1	0.45	1.6
Ammonia raw	mg/L	17	7	31
Nitrate effluent	mg/L	17	11	22

* Measured from April to September. Adopted from City of London (2017).

4.4 Design, setup, and methods

A laboratory scale Biochar-Modified Soil Aquifer Treatment (BMSAT) system was built using a polyvinyl chloride column with an internal diameter of 5 cm and effective length of 90 cm (Figure 4.2). Dimensions of the column were selected based on previous laboratory scale SAT studies (Velasquez, 2016). Primary wastewater, collected from the study site, was pumped into the column at a constant head of 20 cm, allowing for gravity flow conditions. An overflow weir maintained the head throughout the experiments. A valve was installed at the bottom of the column to allow for sample collection. The column was operated at 20°C (± 1°C).

Barco silica sand (ISO 9001) was obtained from Opta Mineral Inc. (Waterdown, Canada). The selected mesh number was 32 so sand particles were larger than 0.125 mm and smaller than 0.5 mm which represents high permeability aquifer recharge zones with fine to medium grain size distribution. The sand was washed, dried for 72 hours at 65°C, and then packed into the column. The effective depth of sand was 30 cm, with dry bulk density of 1.59 g/cm^3, and specific gravity of 2.65. The bottom 10 cm of the column were filled with gravel to provide stability for the column operation. Major oxides composition is given in Table 4.3.

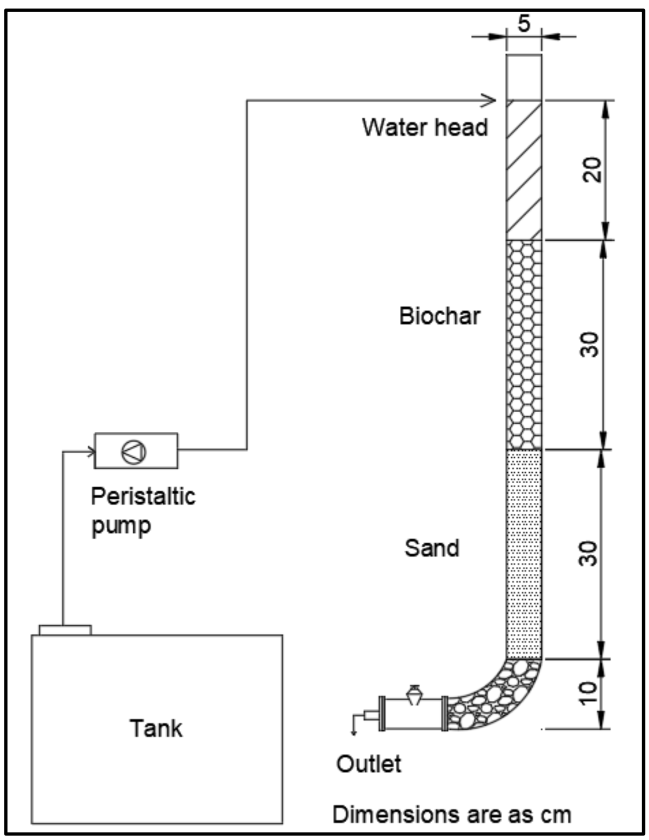

Figure 4.2 Schematic diagram of the biochar-modified SAT column.

Table 4.3 Major oxides composition of selected sand.

Element	Percentage	Element	Percentage
SiO_2(total)	99.7%	Na_2O	<0.01%
Al_2O_3	0.14%	MgO	<0.01%
Fe_2O_3	0.016%	CaO	<0.01%
K_2O	0.04%		

Biochar was obtained from Passive Remediation Services Ltd. (Armstrong, BC). Sufficient 5 M NaOH solution and 1 M HCl were added to the biochar to adjust its pH to 7. It was then cleaned with deionized water (DI) to remove residual surface particles, dried at 65°C for 72 hours, and sealed in an airtight container until use. The remaining 30 cm effective depth of the column was then packed with the prepared biochar.

Secondary effluent samples were collected from the Vauxhall WWTP, and stored in 2-gallon high density polyethylene drums. The gallons were stored at room temperature (20°C) to acclimatize feed wastewater to the column operating conditions. The experiment spanned a period of three months, and site visits to the Vauxhall WWTP were conducted weekly. Effluent column samples were collected weekly and stored at 4°C until analysis. All experiments were repeated three times, and the results reported represent average values.

4.5 Analytical methods

4.5.1 Scope and application

For total metal analysis, Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES) was used (Agilent 5100 Synchronous Vertical Dual View, SVDV). This was preceded by Aqua Regia Digestion (SM3120). The application of ICP-OES in the analysis of environmental materials, such as natural water, wastewater, sediments and air born particles is widely known. ICP-OES is especially effective for analyzing refractory metals (such as Cr, V) and has low susceptibility to interelement interferences other than spectroscopic interferences.

For the determination of anions such as chloride, fluoride, nitrate, and nitrite, Ion Chromatography (IC) was used (Dionex Ion Chromatograph Integrion System; Thermo Fisher Scientific; SM4110C). Ion Chromatography eliminates the need to use hazardous reagents, and effectively distinguishes among the halides (Br, Cl, F) and the oxy-ions (SO_3, SO_4, NO_2, NO_3). This method is applicable, after filtration to remove particles larger than 0.45 µm, in surface, ground, and drinking water as well as wastewater. It is also applicable to the determination of water leachable anions in soils, biosolids and on filters, and to the determination of Toxicity Characteristic Leachate Procedure (TCLP) anions in solids/soils using TCLP-001 procedure.

For NH_3-N, Automated Segmented Flow AA3 Colorimetric Analyzer (seal) was used following MOE E3364 standards. For TKN, the Sulphuric Acid Digestion method was used, followed by the Automated Segmented Flow AA3 Colorimetric Analyzer (MOE E3367). For *E. coli* analysis, the membrane filtration method was used using EC medium incubated at 44.5°C (MOE E3407). EC and pH were determined using a Man-Tech Automated Meter PC-Titrate System™ (Mandel Scientific, Canada), following standard methods SM2510B and SM4500+, respectively. The analytical range for EC, for a 1.0 cm cell length, is approximately 0.5 µS/cm to 1999 mS/cm. The relative detection difference is 0.02 pH units. Results were reported at a corrected temperature of 25°C.

4.5.2 Principles

Atomic emission spectrometry is a branch of the analytical spectrometry which derives analytical information from atomic spectra in the optical region of the electromagnetic spectrum (i.e. the ultraviolet, the visible, and the near infrared). The atomic spectra in this region originate from energy transitions in the outer electronic shells of free atoms and ions.

The emission source is an Inductively Coupled Argon Plasma, and is provided by a 2.5 KVA crystal-controlled RF Generator operating at 40.0 MHz. The high frequency current is applied to a water-cooled copper coil placed around the outside of a quartz torch assembly. Streams of argon gas flow through each of the three concentric tubes of the torch. At ignition time the gas stream is seeded with electrons by a Tesla coil and discharge occurs through the argon flow. Electrons together with ions formed by collisions start moving rapidly, in circular orbits, due to the electromagnetic field surrounding the coil. After a few popping flashes at the open end of the torch the plasma is ignited. It forms a cone shaped flame. The inductively coupled plasma formed is maintained by inductive heating of the flowing gas in a way similar to the inductive heating of a metallic cylinder placed in the axle of the induction coil. The temperature in the core of the plasma can reach 10,000°K.

In Ion Chromatography, wastewater samples were injected into a stream of carbonate-bicarbonate eluent, and passed through a series of ion exchangers. The anions of interest are separated on the low capacity, strongly basic anion exchanger (Guard and Separator columns). The separated anions are directed through a hollow fibre cation exchanger membrane or micromembrane suppressor, bathed in continuously flowing, strongly acidic solution (Regenerant solution). In the suppressor, the separated anions are converted to their highly conductive acid forms. The separated anions in their acid forms were measured by conductivity. Quantification is by measurement of peak area or peak height.

In the membrane filtration method, a known volume of sample was passed through a 0.45 μm membrane filter using vacuum filtration and applied to a nutrient selective medium, m-Tech agar with BCIG. These tests were incubated for 24 ± 2 hours at 44.5 ± 0.2°C. During this incubation period, *E. coli* Colony Forming Units (CFUs) emerged as light blue or dark blue colonies. All colonies that met the above criteria were counted and used to calculate the number of CFUs per 100 mL of sample. Colonies may arise from pairs, chains, clusters or single-cells, all of which were included in the count.

E. coli is one of the principle species making up the thermotolerant coliform group capable of fermenting lactose at 44.5 ± 0.2°C. *E. coli* is the preferred indicator because it excludes most other *Klebsiella* organisms (fecal coliforms) which may or may not originate from fecal sources. *E. coli* is the definitive organism for demonstrating fecal pollution of water because it is the only member of the coliform group that is unquestionably an inhabitant of the intestinal tract and does not persist for long in the environment as a free-living organism. Therefore, when found in environmental samples, *E. coli* represents evidence of recent fecal contamination and thus the possibility of the presence of enteric pathogens that may also be associated with fecal material.

E. coli meets the criteria of a valid fecal pollution indicator in that it is present in the intestine in numbers larger than those of enteric pathogens; it behaves similarly to enteric pathogens within the aquatic environment, and it is more susceptible than most enteric pathogens to treatment or disinfection procedures (Cliver and Newman, 1984). The presence of *E. coli* in a water supply indicates contamination with fecal material from warm-blooded animals. It must be assumed that, if *E. coli* has gained access to a waterway, enteric pathogens also may have entered the water.

The determination of conductivity is performed by measuring the resistance occurring in an area of the test solution defined by the probe design. A voltage is applied between the two electrodes immersed in the test solution, and the voltage drop caused by the resistance of the solution is used to calculate its conductivity per centimetre. The basic unit of measure for conductivity is the siemens (or mho), the reciprocal of the ohm in the resistance measurement.

For pH, the glass electrode responds to the hydrogen ion concentration (activity) by developing an electrical potential at the glass/liquid interface. At a constant temperature, this potential varies linearly with the pH of the solution being measured. pH is defined as $-\log [H^+]$; it is the intensity of acidity. At 25°C, pH 7.0 is neutral (i.e. the activities of the hydrogen ions (H^+) and hydroxyl ions (OH^-) are equal, each corresponding to an approximate activity of 10–7 M. The neutral point is temperature dependent, varying from pH 7.5 at 0°C to pH 6.5 at 60°C. The glass pH electrode system includes a temperature probe, and in many modern pH meters or ‘multi-meters’, the pH of the sample was temperature corrected to 25°C.

4.5.3 Sample preparation, calibration, and quality control

For total metals, samples were collected in a 250 mL plastic bottles. The samples were not filtered, and nitric acid (5% v/v) was used. The maximum holding time was 30 days prior to analysis. Pipettes of 20 mL were used to extract samples into 200 mL volumetric flask and make up to volume with 5% v/v HNO_3. This comforts the maximum analysis limit of 10 mg/L for analytes.

Samples were mixed, and aliquots (10 mL) were taken into polypropylene test tubes, then 1 mL of concentrated Trace metal grade nitric acid was added, and samples were covered with plastic watch glass and heated for 30 minutes in a hot block set at 97 ± 3°C. After 30 minutes, samples were left to cool to room temperature, and diluted to original volume (10 mL) with DI water. Samples were then filtered using a pre-rinsed Whatman #1 filter as required before analysis. A Blank and ICP wastewater Quality Control solution, and matrix spike were digested along with the samples.

Wavelengths (in nm) and intensities (in used were as follows, respectively: Aluminum: (396.152, 200), Arsenic (193.696, 125), Beryllium (313.107, 96000), Boron (249.772, 5000), Cadmium (214.439, 2400), Chromium (267.716, 3939), Cobalt (238.892, 1457), Copper (327.395, 4000), Iron (238.204, 3800), Lead (220.353, 246), Lithium (670.783, 41000), Molybdenum (202.032, 1105), Nickel (231.604, 518), Vanadium (311.837, 1100), and Zinc (213.857, 5341).

For chloride, fluoride, nitrate, and nitrite, suspended particulates were removed by filtering through a 0.45 μm membrane filter. Samples were diluted if necessary with reagent grade water. Samples were analysed on IC as soon as possible after collection. Samples that were filtered to remove particulate were accompanied by a filter blank to show no indication of contamination above Method Detection Limit (MDL). Sodium bicarbonate stock solution (0.5 M) was used as an eluent solution. The solution was prepared by dissolving 8.4 g of sodium bicarbonate ($NaHCO_3$) in 100 mL of reagent grade water, in a 200 mL volumetric flask.

For NH_3-N, wastewater samples were collected in 100 mL plastic bottles, and stored at 4 ± 3°C. Samples were preserved with H_2SO_4 at pH <2. Dichloroisocyanuric acid (DCI) was used as a detection reagent solution. The solution was prepared by dissolving 20 g of NaOH and 3 g of DCI sodium salt dihydrate into 1000 mL of DW. Samples were extracted using 0.5 mL pipettes into pre-labelled 100 mL Class A volumetric flasks containing about 50 mL of DW and 1 mL of 20% H_2SO_4.

For TKN, an aliquot of the sample was digested in a block digester with reagents to convert trivalent organic nitrogen compounds to ammonium ions (highly acidic media). The determination of ammonia was based on the colourimetric method in which a blue colour was formed by the reaction of salicylate sodium hypochlorite ($NaClO$) and sodium nitroprusside ($Na_2[Fe(CN)_5NO]$). The intensity of the blue colour varied linearly with the concentration of ammonium ions through the analytical range. The absorbance was measured using a 1.0 cm flow cell and a 660 nm filter.

Samples were collected in 100 mL plastic bottles with plastic-lined caps, preserved with H_2S0_4 at a pH <2, and stored at 4 ± 3°C. The digestion was done using 20% H_2SO_4 solution. 200 mL of concentrated H_2SO_4 were added to about 600 mL of DW in a properly labelled 1000 mL amber glass bottle, cooled to room temperature, and then diluted to 1000 mL in a fume hood. For buffer TKN, 50 g of sodium potassium tartrate ($C_4H_4KNaO_6.4H_2O$) and 4 NaOH pellets were dissolved in 600 mL of DW. To prevent contamination, heat was applied, and boiling temperature was maintained for 1 hour. 27.2 g Disodium Hydrogen Phosphate-7-Hydrate ($Na_2HPO_4.7H_2O$) was then added to 64 g of NaOH solution, and 4.0 mL of 15% Brij-35 solution, and diluted to 1000mL.

For *E. coli*, samples were collected in approved sterilized 200 mL plastic bottles containing $Na_2S_2O_3$. Certified bottles are verified for sterility upon receipt of each new shipment before being utilized for sampling. Sterile membrane filters with 0.45 μm pore size, and 47 mm grid was used. Samples were incubated within 30 minutes of the filtration step, place it into the incubator at 44.5±0.2°C, and incubated for 24 hours.

4.6 Results and discussion

4.6.1 Electrical conductivity

The electrical conductivity should be less than 70 to 300 mS/m, and the maximum tested EC is 186 mS/m (EPA, 2012; Figure 4.3). Several studies found that electrical conductivity ameliorates salt stress effects on plants through salt sorption (Thomas *et al*., 2013; Akhtar *et al*., 2015). Usman *et al*. (2016) studied the effect of *Conocarpus* biochar (BC) and organic farm residues (FR) at different application rates on yield and quality of tomatoes grown on a sandy soil under drip irrigation with saline or non-saline water.

They found that salt stress adversely affected soil productivity, as indicated by the lower vegetative growth and yield components of tomato plants. However, this effect tended to decline with application of FR or BC, especially at high application rate and in the presence of biochar. Under saline irrigation system, for instance, the total tomato yield increased over the control by 14.0 to 43.3% with BC, and by 3.9 to 35.6% with FR.

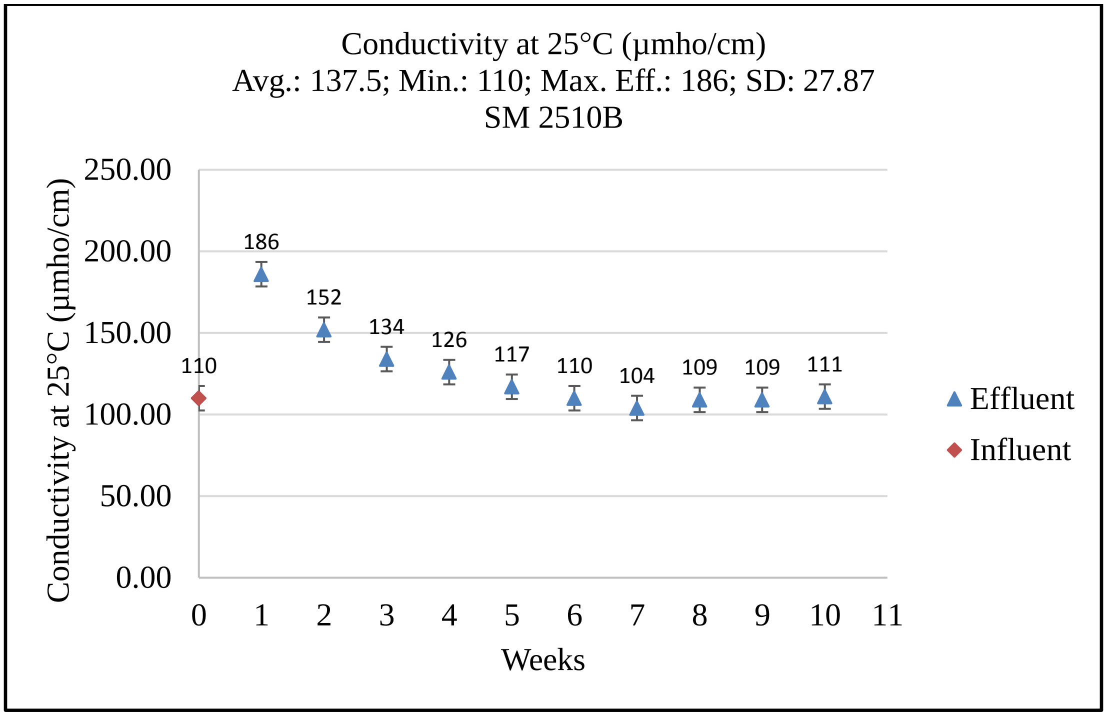

Figure 4.3 Analysis of the electrical conductivity in the Biochar-Modified Soil Aquifer Treatment System.

Therefore, biochar has the potential to be used for adsorption of salts in added units to existing wastewater treatment plants if salinity of treated wastewater exceeds the maximum recommended limits. SAT has not been studied distinctly for its impact on electrical conductivity of wastewater. However, Job and Hardaha (2016) found the effect of long term sewage application on soil was cumulative increase in various parameters of soil such as electrical conductivity, nitrogen, phosphorous, organic carbon, micro-nutrients, and heavy metals.

4.6.2 Sodium adsorption ratio (SAR)

For safe irrigation, the SAR should be less than 18. The maximum tested SAR is 2.47 (EPA, 2012; Figures 4.4, 4.5, 4.6, and 4.7). Several studies suggest that biochar's ability to remove metals via adsorption is high. For calcium and magnesium, removal efficiencies of 0.51 (± 0.08) mg/g and 0.49 (± 0.05) mg/g were reported, respectively (Bian, 2011). In a study conducted on an Abu Dhabi arid soil, biochar was applied at different concentrations (10, 50, and 100 g/kg of soil), and was found to (1) increase the Cation Exchange Capacity (CEC) from 2.5 to 6.7 meq/100g, and (2) lower SAR of soil to below sodic levels (<13) (Khalifa and Yousef, 2015).

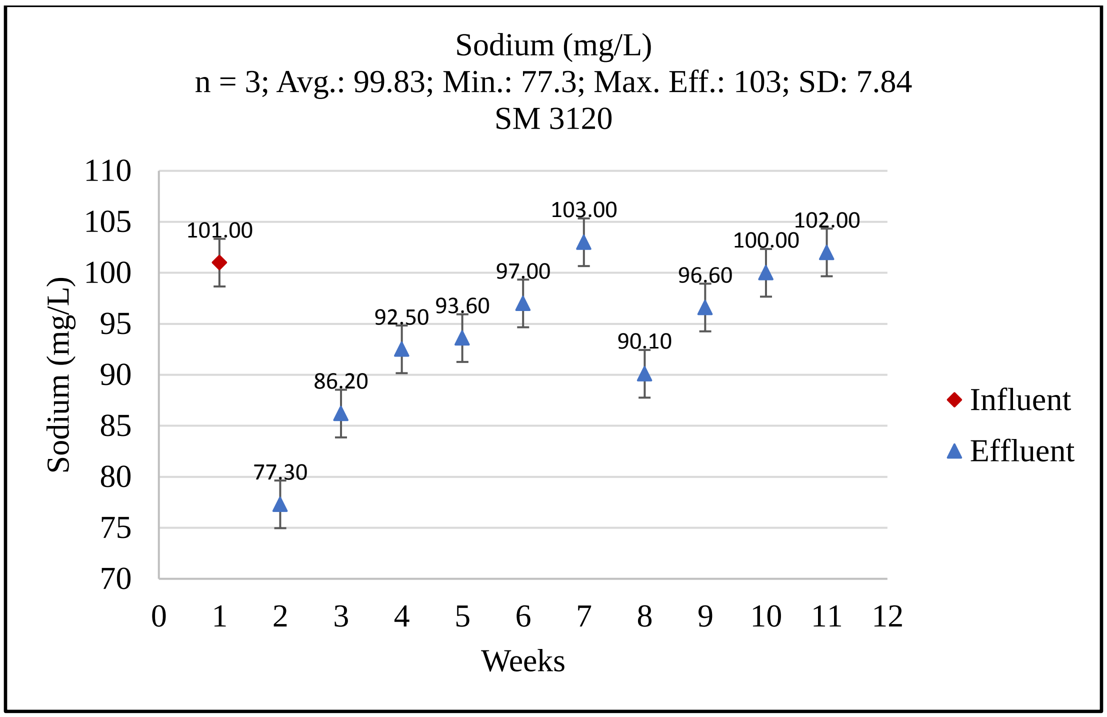

Figure 4.4 Analysis of sodium in the Biochar-Modified Soil Aquifer Treatment System.

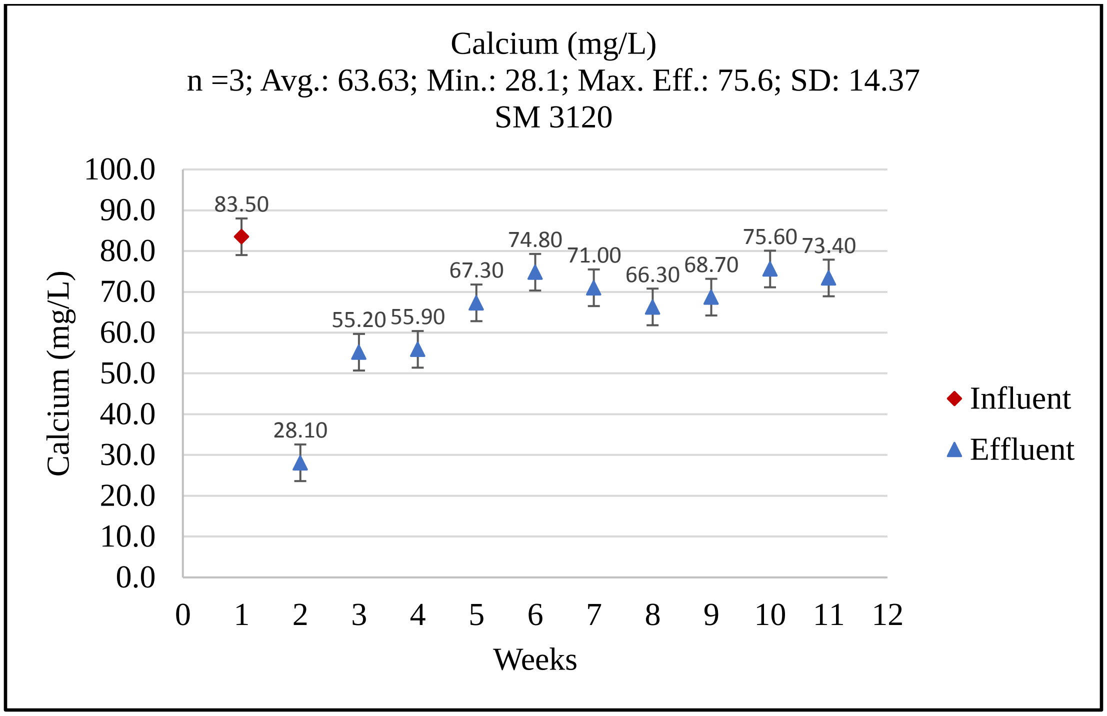

Figure 4.5 Analysis of calcium in the Biochar-Modified Soil Aquifer Treatment System.

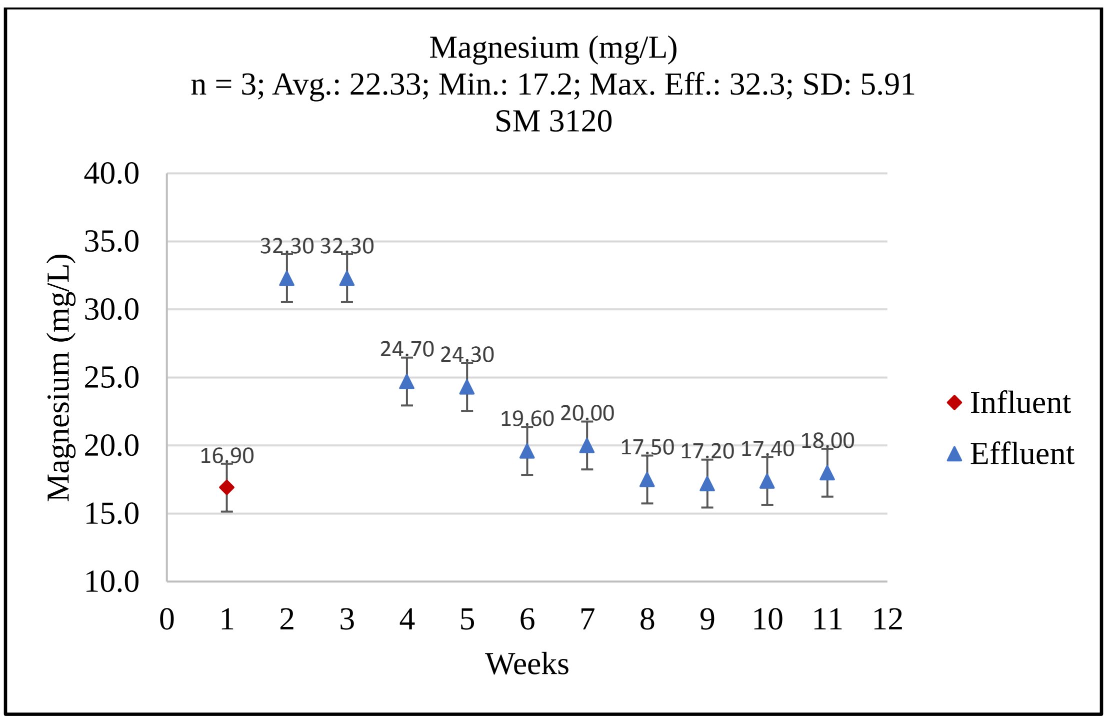

Figure 4.6 Analysis of magnesium in the Biochar-Modified Soil Aquifer Treatment System.

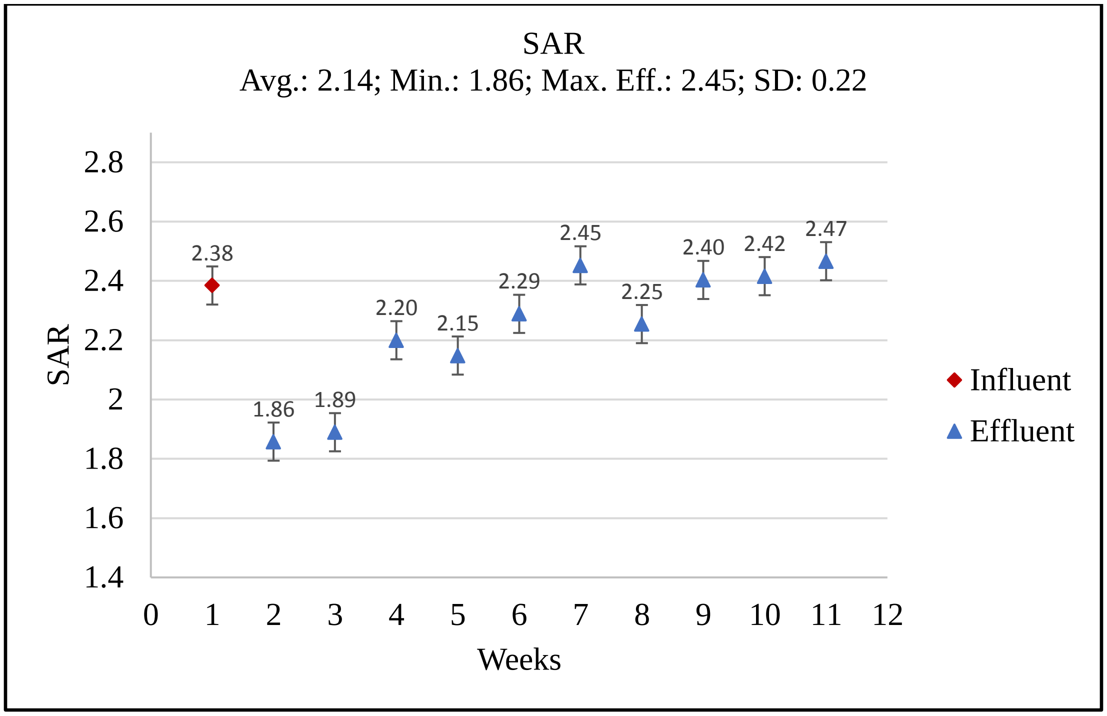

Figure 4.7 Analysis of SAR in the Biochar-Modified Soil Aquifer Treatment System.

Biochar made from the pyrolysis of agricultural residues (e.g. pecan shells, pecan orchard prunings, cotton gin trash, and yard waste) was found to increase SAR in both sandy loam and clay loam soils, but remained in the range of 2.9 to 6.9 (Zhang *et al*., 2016).

A simplified treatment scheme for municipal wastewater reuse in agriculture was investigated (De Sanctis *et al*., 2017). The Sequencing Batch Biofilter Granular Reactors were found to remove > 80% of total nitrogen and 90% of the total suspended solids, and decrease electrical conductivity in the effluent. However, SAR increased slightly but remained under the 18 threshold. Sand filtration was combined with the reactors, and had positive effects on plant effluent quality and the latter could even comply with more restrictive reuse criteria.

4.6.3 Boron

Boron adsorption depends largely on the type of soil. Although boron is mobile in sands and gravels, it can be adsorbed on clay due to cation adsorption and ion exchange when the SAT system is first put into operation (Pescod, 1992; Figure 4.8). Therefore, SAT systems can significantly reduce the concentrations of trace elements in sewage effluent

(FAO, 1985). Excessive liming of soil can make boron less available to plants at higher pH (Parks and Edwards, 2005).

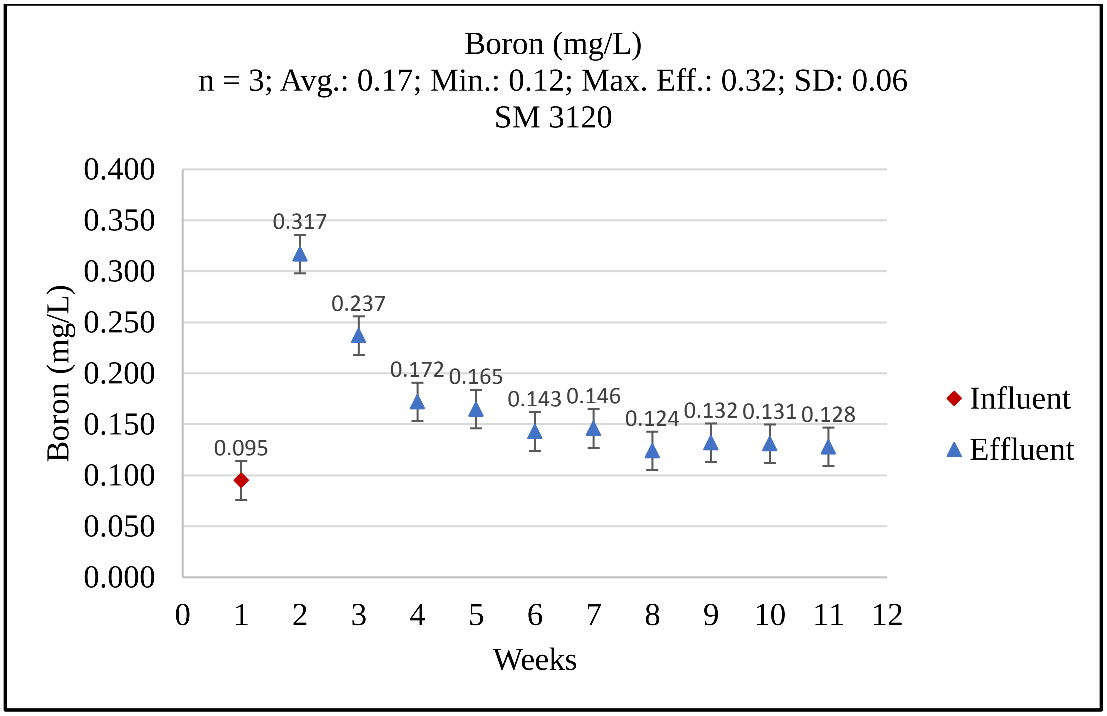

Figure 4.8 Analysis of boron in the Biochar-Modified Soil Aquifer Treatment System.

The following concentrations (ppm, dry matter) of boron in leaf tissue are considered excessive (Gough *et al*., 1979): >260, Navel and Valencia oranges; >80, pears and plums; >85, almonds; >90, apricots; apples and peaches; >250, avocados; >300, grapes; and >250, rutabaga. If boron content of water is 0.15 to >0.30 mg/L, and 45 cm/ha (or more) of water is used, boron deficiency is unlikely. But if the boron content of the water is greater than 1.1 mg/L, some sensitive crops will begin to show toxicity symptoms (Bradford, 1966). Therefore, boron concentration in treated effluent of the BMSAT system is suitable for irrigation of crops.

4.6.4 Total nitrogen

Total nitrogen in raw wastewater was measured at 26.8 mg TN/L, and the BMSAT effluent maximum tested concentration was 18.6 mg TN/L (Figures 4.9, 4.10, 4.11, 4.12, and 4.13). However, nitrogen threshold is related to total concentration in soil. With the application of nitrogen to agricultural soils through fertilizers, consideration could be given to further reduce total nitrogen concentration in treated wastewater by means of applying a low-cost additional treatment unit. Nitrogen balance can be calculated using the following formula:

Total Nitrogen = Total Kjeldahl Nitrogen (Ammonia as Nitrogen + reduced Nitrogen + Organic Nitrogen) + Nitrate as Nitrogen + Nitrite as Nitrogen.

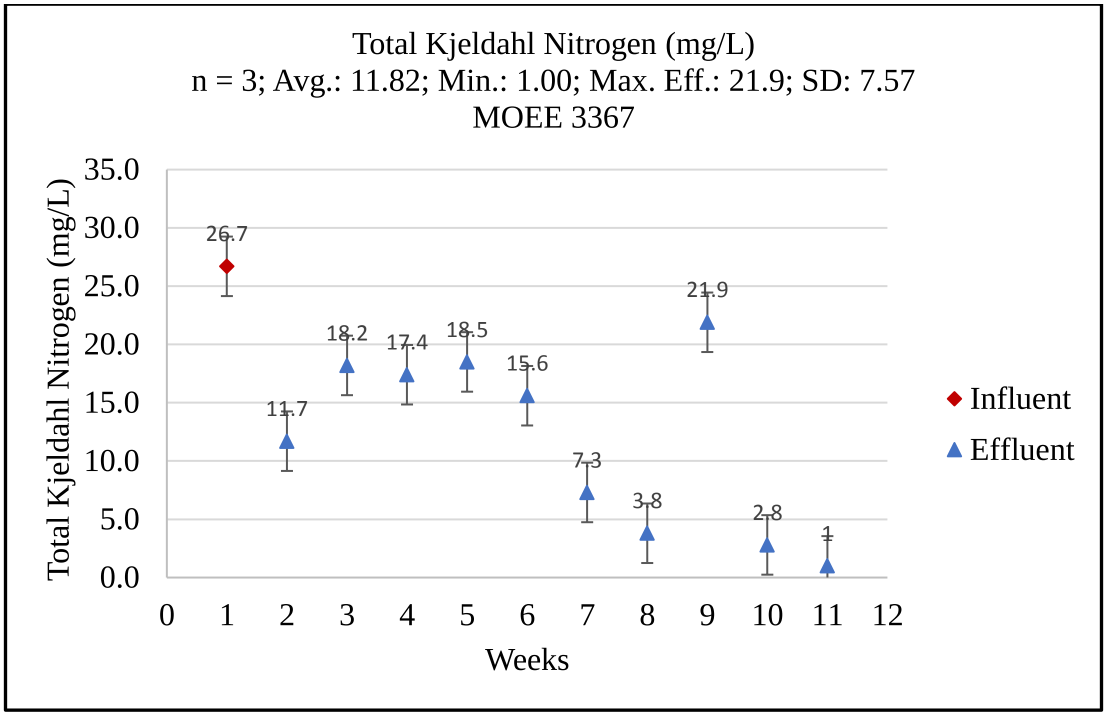

Figure 4.9 Analysis of Total Kjeldahl Nitrogen in the Biochar-Modified Soil Aquifer Treatment System.

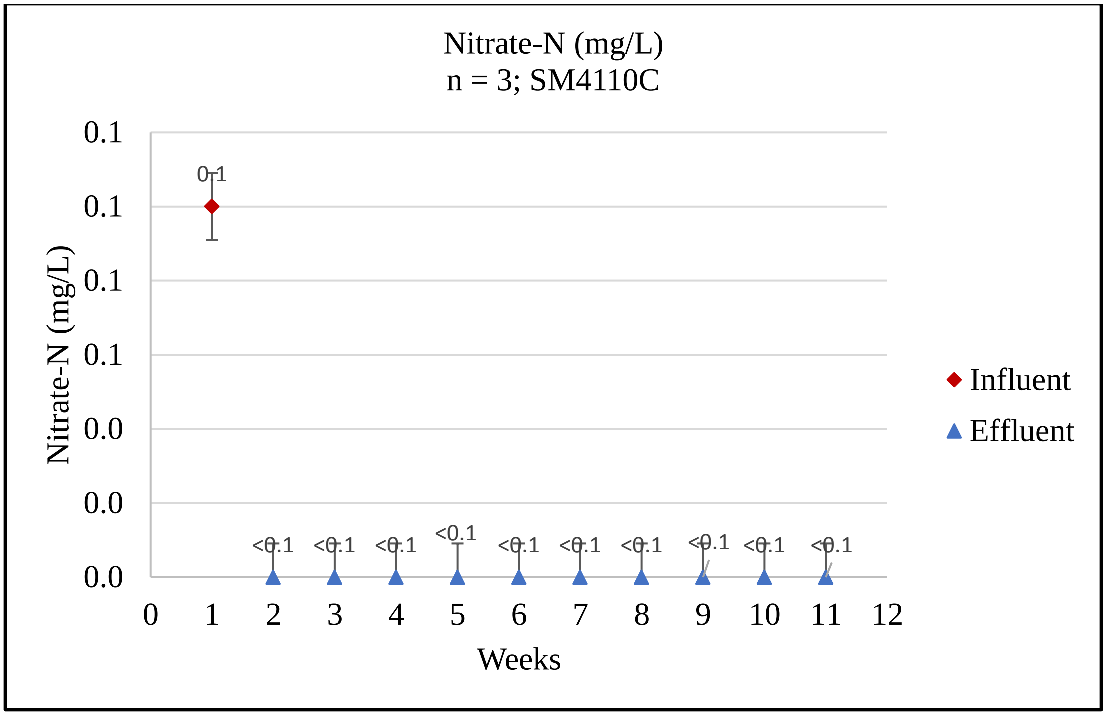

Figure 4.10 Analysis of Nitrate in the Biochar-Modified Soil Aquifer Treatment System.

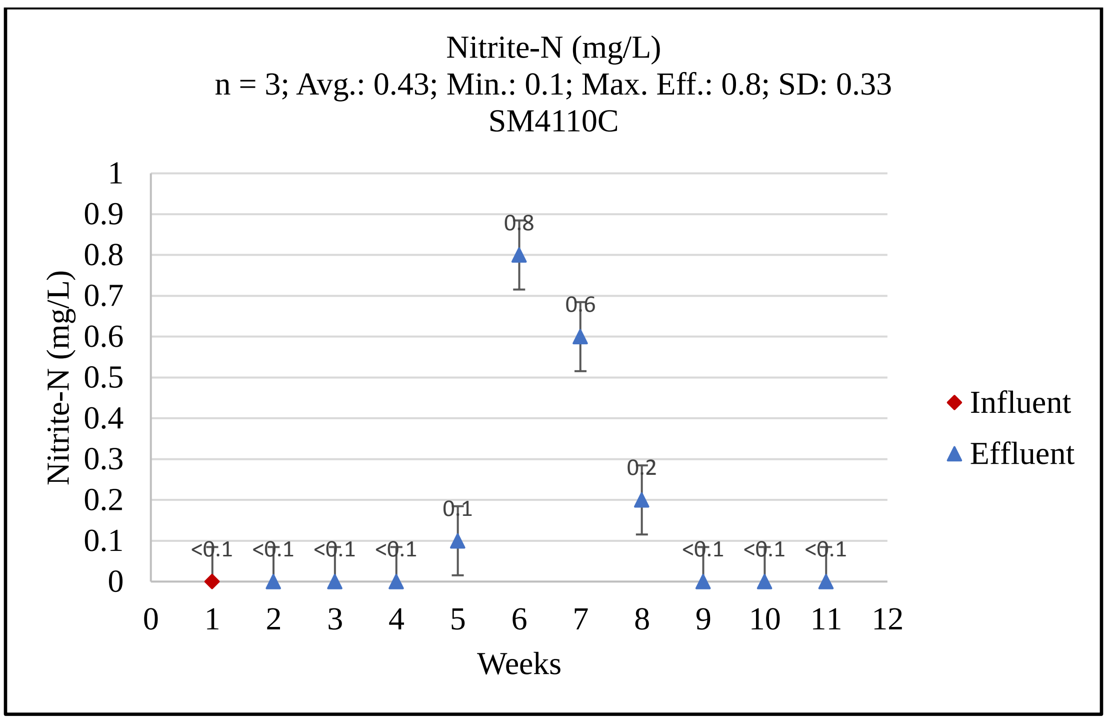

Figure 4.11 Analysis of Nitrite in the Biochar-Modified Soil Aquifer Treatment System.

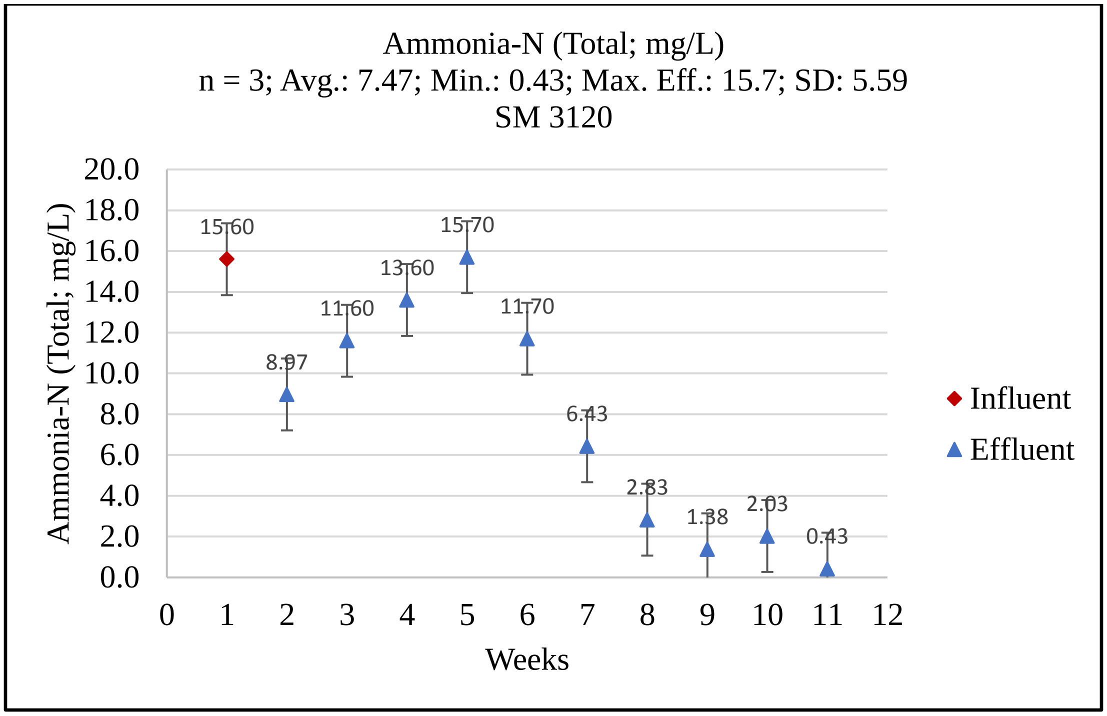

Figure 4.12 Analysis of ammonia-N in the Biochar-Modified Soil Aquifer Treatment System.

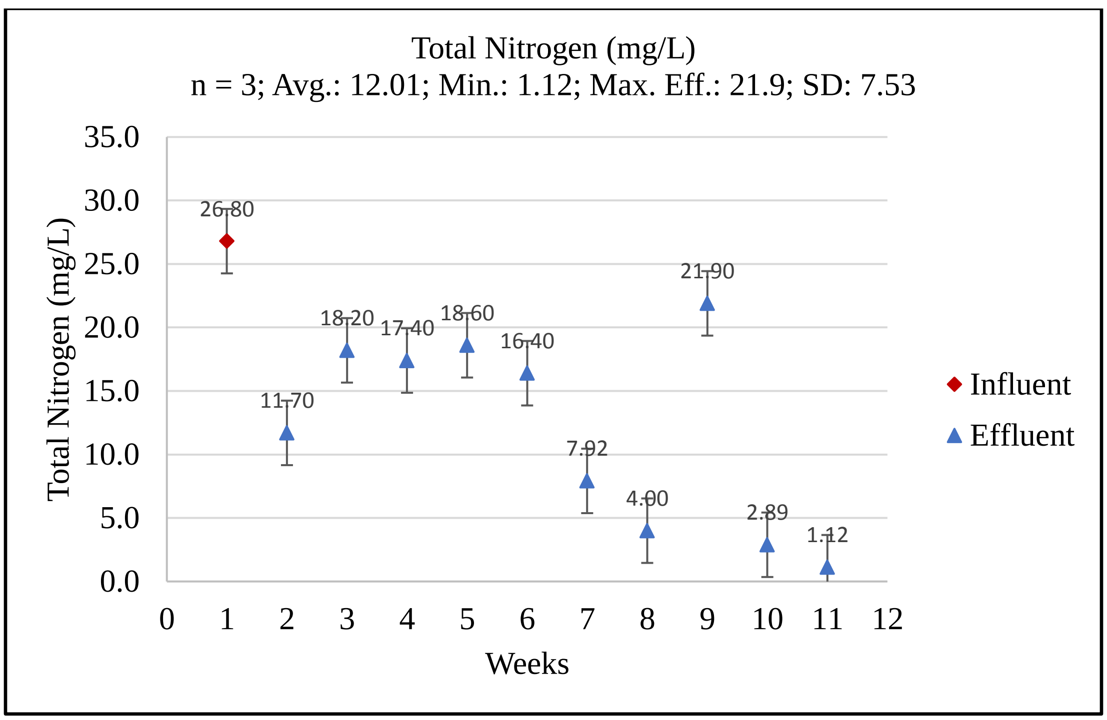

Figure 4.13 Calculated Total Nitrogen in the Biochar-Modified Soil Aquifer Treatment System.

Nitrogen may have likely reduced in the BMSAT system effluents due to a combination of physical and microbiological processes. This includes adsorption on to biochar as well as consumption by anoxic bacteria. Using batch sorption experiments, Ghezzehei *et al*. (2014) found that biochar can adsorb up to 20–43% of ammonium and 19–65% of phosphate in dairy wastewater in 24 hours. Gupta *et al*. (2015) found that biochar-amended wetlands outperformed gravel, with excess removal of 18-28 mg TN/L. Biochar is known for its ability to retain nitrogen (Ding *et al*., 2010). Williams (2016) found that biochar reduced runoff concentrations of TKN and NH_3-N for municipal sludge treatments, and TKN, NH_3-N, TSS, and FC for horse manure treatments. Hence, biochar will likely have greater potential if combined with slow sand filtration.

Soil-biochar amendment can affect nitrogen transfer and soil nitrogen cycling processes, reducing N_2O emissions (Zheng *et al*., 2010). Biochar contains high C/N ratios, which aids in the biological immobilization of inorganic N, thus decreasing ammonia volatilization (Lehmann and Rondon, 2006). Taghizadeh-Toosi *et al*. (2011) found a reduced NO_3^--N pool in biochar-amended soil plots (more than 70%), and concluded that incorporating biochar into the soil can significantly diminish ruminant urine-derived N_2O emissions. Adsorption of NO_3^- by the biochar surface is responsible for a reduction in N_2O emissions (van Zwieten *et al*., 2010).

4.6.5 pH

The pH range for irrigation water is from 6.5 to 8.4, which normally does not present any problems for treated effluent (EPA, 2012; Figure 4.14). For example, activated sludge effluents have a mean pH of 8, and the pH for effluents of Vauxhall WWTP ranged from 6.7 to 7.8. Biochar has been found to reduce soil salinity which would decrease the need for liming at the wastewater treatment stage (Abewa, 2013).

Biochar potential impact is likely to be positive if properly treated and or activated, in light of previous studies. However, if biochar was undergone pyrolysis, the influence of temperature can be significant; the pH was 7.9 and 8.6 at 400 and 500°C, respectively, in peanut hull biochar; 5.4 and 8.0 at 250 and 500°C, respectively, in pecan shell biochar;

5.9 and 7.2 at 350 and 700°C, respectively, in switch grass biochar; and 8.7 and 10.3 at 350 and 700°C, respectively, in poultry litter biochar (Novak *et al*., 2009).

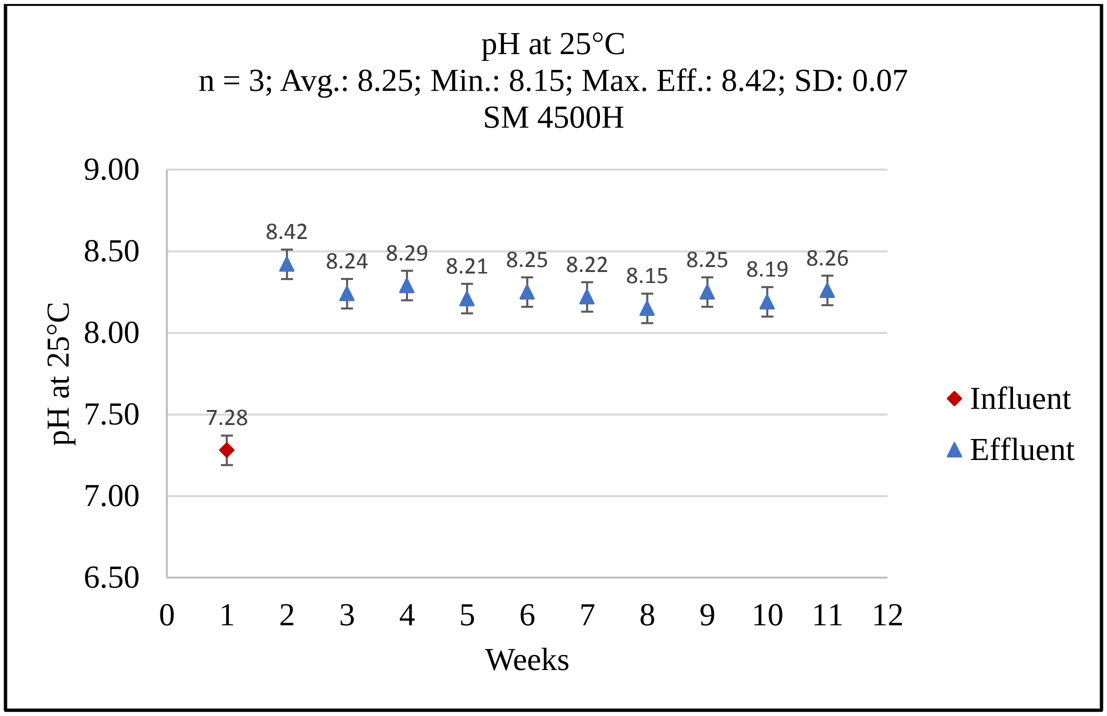

Figure 4.14 Analysis of pH in the Biochar-Modified Soil Aquifer Treatment System.

4.6.6 Microbiological quality

The influent *E. coli* was 37 million CFU/100 mL (Figure 4.15). After 10 weeks of analysis, *E. coli* dropped to less than 2 CFU/100 mL. Bacteria die-off may likely be significant. However, *E. coli* may have been consumed by other pathogens such as protozoa, and other worms. This requires in-depth microbiological assessment, therefore biochar reuse may still need to be sterilised for safe use.

For treated wastewater application on land, the World Health Organisation requires that (WHO, 2006):

- for restricted irrigation, i.e. irrigation of all crops except salad crops and vegetables which may be eaten uncooked, treated wastewater must contain $\leq 10^4$ *E. coli* per 100 mL, and ≤ 1 human intestinal nematode per liter; and

- for unrestricted irrigation, i.e. irrigation of all crops including those which may be eaten uncooked, treated wastewater must contain ≤ 1000 *E. coli* per 100 mL, and ≤ 1 human intestinal nematode per liter.

The development of a stable soil biomat appeared to provide the best on-site water treatment or protection for subsequent groundwater interactions of OSSs (Tomaras *et al*., 2008). However, Appling *et al*. (2013) found *E. coli* concentration was higher by 100-fold in a septic tank effluent than influent wastewater samples, indicating the growth of E. coli inside the tank. This suggests that microbial load of the wastewater is potentially enhanced during its storage in the tank. Pathogenic deactivation is influenced by natural parameters such as sunlight exposure, temperature, pH, and retention time, and manmade processes such as tertiary treatment. Aerobic units were installed in some septic tanks to facilitate the processes of nitrification and denitrification to sequentially remove nitrogen (Rich, 2008). The impact on effluent microbial quality and quantity remains largely unknown.

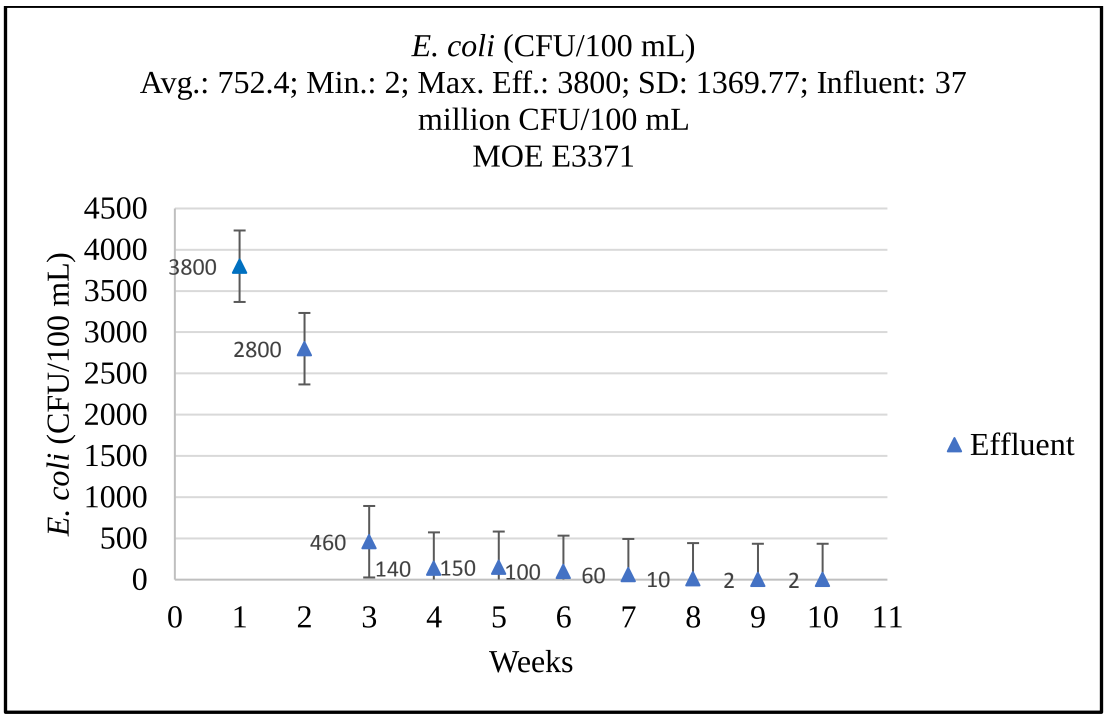

Figure 4.15 Analysis of *E. Coli* in the Biochar-Modified Soil Aquifer Treatment System.

The real question, however, is not how many pathogens and or *E. coli* are present in treated wastewater and sludge, but how many pathogens can be ingested without exceeding the tolerable infection risk, which should be the basis of strategic OSS standards (Mara, 2010). Pathogens in raw wastewater are reduced by both treatment and post-treatment, but pre-ingestion, health-protection control measures. These include the type of wastewater treatment, method of irrigation (drip or sprinkler), die-off of pathogens since last irrigation and food preparation (such as washing, peeling, and cooking).

This means that latest microbial assessments should consider the geophysical conditions of wastewater treatment units as well as local irrigation practices, and a conclusive result can only be drawn upon from localised assessment. However, the overall health impact of reusing treated wastewater in agriculture has been shown to be acceptable and post-treatment health control measures could play a key role in reducing pathogens (The World Bank, 2010). In order to properly interpret and apply the guidelines in a manner appropriate to local conditions, a broad-based policy approach is required that will include OSSs legislation as well as positive and negative incentives to support the adoption of good non-treatment or post-treatment practices.

4.6.7 Aluminum

The recommended limit for aluminum in irrigation water is 5 mg/L, but soils at pH 4.7 to 7.8 will precipitate the ion and eliminate toxicity (EPA, 2012; Figure 4.16). However, certain crops cannot be grown on some naturally acidic soils without applications of lime. Sensitivity of crops to aluminum in culture solutions are as follows: sensitive (depressed by 2 mg/L) barley, beet, lettuce, and timothy; intermediate (depressed by 7 mg/L) cabbage, oats, radish, rye, and sorghum; and tolerant (depressed by 14 mg/L) corn, redtop, and turnip (Samantaray *et al*., 1998).

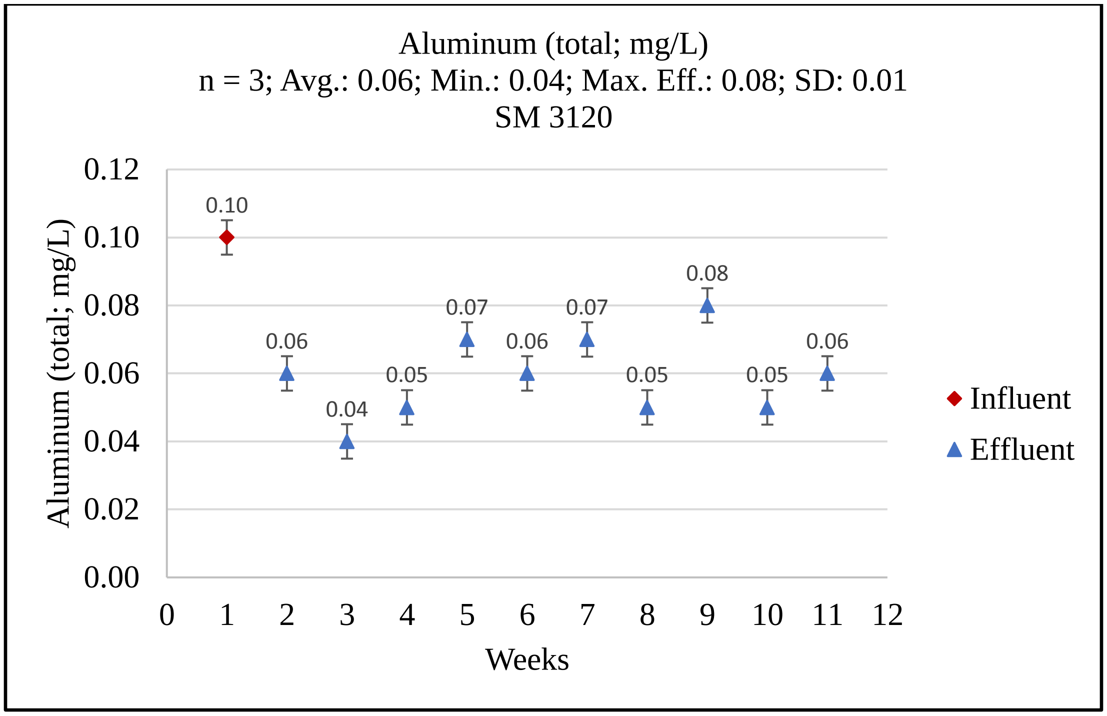

Figure 4.16 Analysis of aluminum in the Biochar-Modified Soil Aquifer Treatment System.

Biochar surface can contain numerous chemically active groups, such as OH, COOH, and ketones that produce potential for the adsorption of toxic chemicals such as aluminum (Al) and manganese (Mn) in acid soils, and arsenic (As), cadmium (Cd), nickel (Ni), copper (Cu), and lead (Pb) in heavy metal-contaminated soils (Berek *et al*., 2011). Aluminum concentrations in shoot also indicated the beneficial effects of biochar applications. Across three lime levels (0, 3, and 6 tons/ha), shoot Al averaged 129 ppm (or µg/g) without biochar as compared to 85 ppm and 70 ppm at 2.5% and 5.0% biochar (Berek *et al*., 2011).

4.6.8 Arsenic

The recommended arsenic limit in irrigation water is 0.1 mg/L (EPA, 2012). All tested samples of the BMSAT system had arsenic concentrations of less than 0.02 mg/L. However, toxicity to crops can be as low as 0.05 mg/L for rice. Certain arsenic metabolites, principally sodium arsenite and arsenic trioxide, are very toxic to plants that these compounds were used as herbicides for many years. Other As compounds such as calcium arsenate, lead arsenate, and cupric arsenite were widely used as insecticides. For example, Wyllie (1937) reported illness and one fatality in a farm family that used water from a deep well having 0.4–10 ppm As (as As_2O_3).

Biochar may increase the cation and anion exchange capacities (CECs and AECs) of soil, thus encouraging greater root development and decreasing aluminum for transport to groundwater (Chan *et al*., 2008). Mohan *et al*. (2007) Sorption isotherms studies were conducted in broad concentration ranges (1–1000 ppb for arsenic, 1 x 10^{-5} – 5 x 10^{-3} ppm for lead and cadmium). They found oak bark biochar (10 g/L) removed about 70% of arsenic, 50% of cadmium, and 100% of lead, all from aqueous solutions, with maximum adsorption occurring over pH range of 3–4 for arsenic and 4–5 for lead and cadmium.

4.6.9 Beryllium

The maximum recommended Beryllium limit in irrigation water is 0.1 mg/L (EPA, 2012). All tested samples of the BMSAT system had Beryllium concentrations of less than 0.002 mg/L. Beryllium is able to replace magnesium as an essential element in some fungi and partly in tomatoes. Further, it apparently stimulates the growth of some plants (ryegrass and kale; 5 mg/L) while inhibiting others (bean; 0.5 mg/L). beryllium (as beryllium chloride) at levels greater than 2 mg/L in nutrient solutions reduces the growth of alfalfa, lettuce, peas, and soybeans (Samantaray *et al*., 1998).

4.6.10 Cadmium

The maximum recommended limit for Cadmium is 0.01 mg/L (EPA, 2012). All tested samples of the BMSAT system had Cadmium concentrations of less than 0.005 mg/L. The conservative limit remains unexplained, especially that Cadmium is a rare element. The general toxicity of this element to plants is considered moderate. Growth depression occurred in soybeans when shoots had 3–5 ppm cadmium. Cadmium was reported to reduce plant growth at the following nutrient solution concentrations: 0.2 mg/L, beets, beans, and turnips; 1 mg/L, corn and lettuce; 5 mg/L, tomato and barley; and 9 mg/L, cabbage. Dong *et al*. (2005) studied Cd effects on the growth and photosynthesis of two tomato cultivars. On average, exposure to 1 and 10 μM Cd for 33 days reduced plant height by 18.9% and 46.4%. Cd may be toxic to beans, beets, and turnips at concentrations as low as 0.1 mg/L (Samantaray *et al*., 1998).

4.6.11 Chromium

The maximum recommended limit for Chromium is 0.1 mg/L (EPA, 2012). All tested samples of the BMSAT system had Chromium concentrations of less than 0.002 mg/L. The concentrations (dry weight) of chromium in plants growing in serpentine soils and showing toxicity symptoms follow: 18–24 ppm in leaves and 375–410 ppm in roots of tobacco; 4–8 ppm in corn leaves; and 252 ppm in oat leaves. In soil, 16 ppm (chromate) reduced growth in tomatoes, oats, kale, and potatoes whereas 10 ppm was toxic to corn, and 5 ppm was toxic to tobacco (Samantaray *et al*., 1998). Significant reduction in dry matter yields of two cultivars of soybean was recorded with as low as 0.5 ppm of chromium in nutrient culture. The 2007 Canadian Soil Quality Guidelines for the Protection of Environmental and Human Health recommends soil should contain less than 64 mg/kg of total chromium, and 0.4 mg/kg of the form hexavalent chromium.

4.6.12 Cobalt

The maximum recommended limit for Cobalt is 0.05 mg/L (EPA, 2012). All tested samples of the BMSAT system had Cobalt concentrations of less than 0.005 mg/L. Small amounts of cobalt in solution cultures, sometimes as small as 0.1 mg/L, produced adverse or toxic effects on many crop plants such as tomato. The symptoms of cobalt excess include depressed growth, chlorosis, necrosis, and even death of the plant. In a radiometric study, straw-derived biochar was demonstrated as a heavy metal-immobilizing agent for contaminated soils or water through sorption separation of cobalt and cadmium (Pipíška *et al*., 2017).

4.6.13 Copper

The recommended upper level of Cu in irrigation water for long-term use is 0.2 mg/L (EPA, 2012; Figure 4.17). It may become toxic to some plants at 0.1 to 1 mg/L. Severe Cu toxicity may occur when there is excess Cu accumulation in the roots, but its translocation to top parts of the plants is limited. Toxicity to copper in plants is usually manifested by yellowish foliage, caused by the interference of excessive copper with iron metabolism in the plant (Feigin *et al*., 2012). Increased levels of copper could likely be due to reaching maximum biochar adsorption capacity.

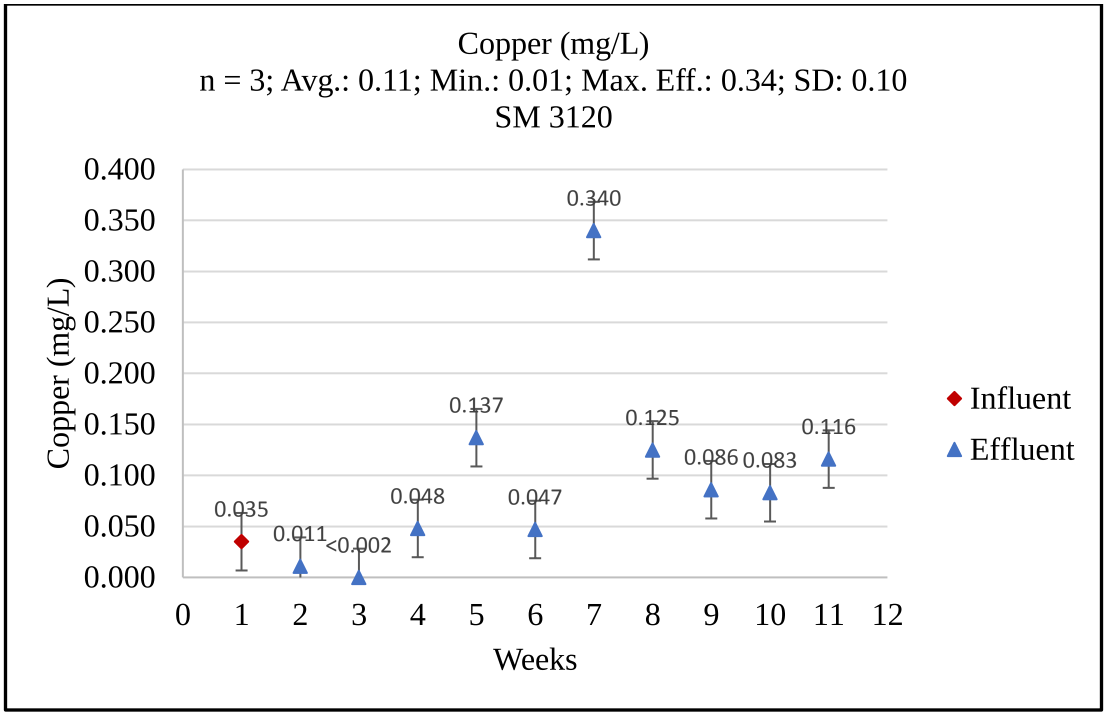

Figure 4.17 Analysis of copper in the Biochar-Modified Soil Aquifer Treatment System.

4.6.14 Fluoride

The maximum recommended limit for fluoride is 1.0 mg/L (EPA, 2012; Figure 4.18). Fluoride is inactivated in neutral and alkaline soils. Superphosphate and elemental phosphorus uptake by plants is often injured by fluorine emissions. Depending on species and growth conditions, 30–300 ppm fluorine (dry weight) generally reduces growth.

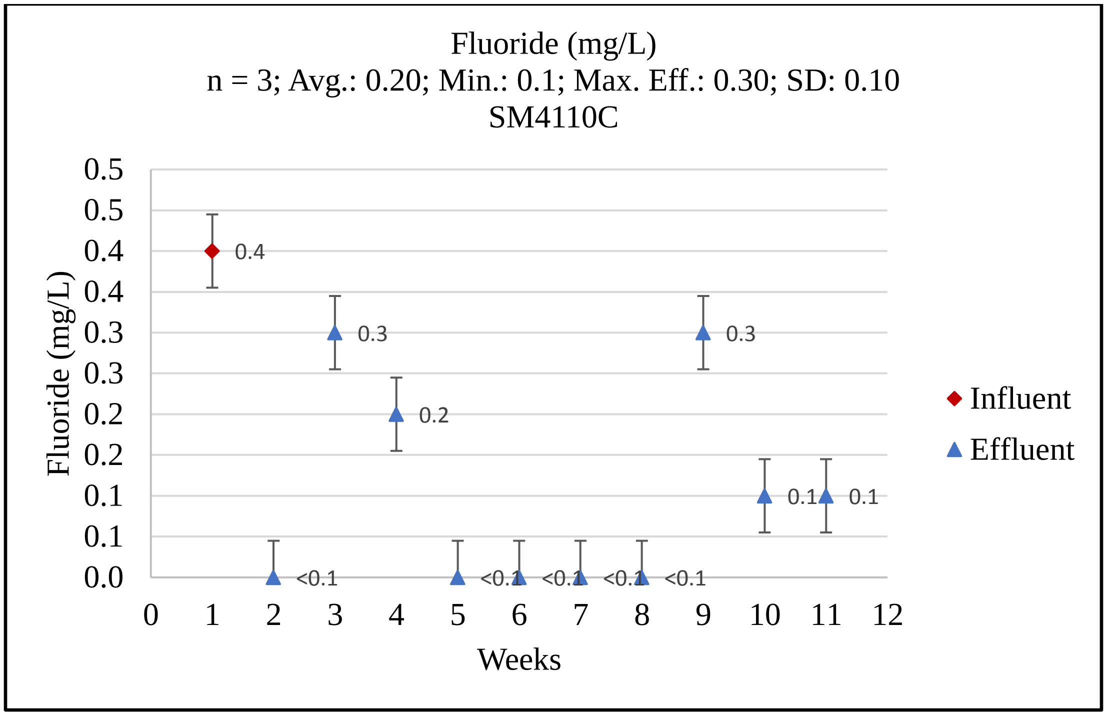

Figure 4.18 Analysis of fluoride in the Biochar-Modified Soil Aquifer Treatment System.

4.6.15 Iron

The maximum recommended limited for iron is 5 mg/L (EPA, 2012; Figure 4.19). Iron toxicity is most commonly associated with highly acidic soil as low pH increases its availability. Excess phosphorus, potassium, molybdenum, and nickel can inhibit the uptake of iron.

Iron sharply increased then continued on a decreasing pattern for the rest of the experiments. The sudden increase may likely be a result of a flush in iron oxides in soil and biochar.

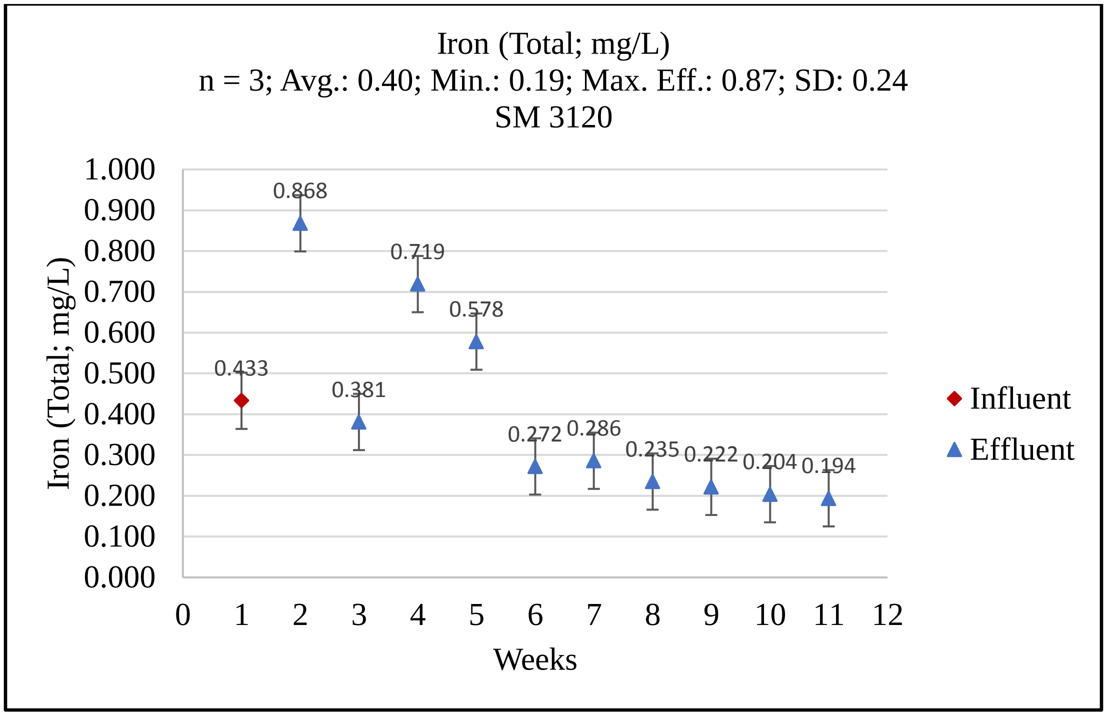

Figure 4.19 Analysis of iron in the Biochar-Modified Soil Aquifer Treatment System.

4.6.16 Lead

All tested samples of the BMSAT system had lead concentrations of less than 0.02 mg/L. The maximum recommended limit for lead is 5 mg/L (EPA, 2012). At higher concentrations, it can inhibit plant cell growth at very high concentrations. Lead, zinc, and other heavy metals often occur together; therefore, it is difficult to associate observed plants toxicity symptoms with lead alone. Plants at these sites may absorb large amounts of lead without exhibiting toxicity symptoms. Although some plant species (such as Broadbean plants) adsorb lead, most of the lead in soils is sparingly soluble and largely unavailable to plants. The availability of lead to plants is generally low, mainly because lead reacts with many soil constituents including phosphate, clay, carbonate, hydroxides, sesquioxides, and organic matter. Dairy-manure derived biochar was found to effectively sorb lead and organic contaminant atrazine (Cao *et al*., 2009).

4.6.17 Lithium

All tested samples of the BMSAT system had Lithium concentrations of less than 0.01 mg/L. The maximum recommended limit for lithium is 2.5 mg/L (EPA, 2012). Up to 5 mg/L is tolerated by most crops, but can be toxic to citrus at low doses (with a recommended limit of 0.075 mg/L). Lithium toxicity has also been observed in avocado, celery, corn, olive, and wheat (Samantaray *et al*., 1998). In contrast, cotton seems to be very resistant to lithium toxicity. Lithium excesses often occur in soils derived directly from igneous rocks rich in ferromagnesian minerals, and in soils derived from sedimentary deposits rich in clays or micas.

4.6.18 Molybdenum

All tested samples of the BMSAT system had Molybdenum concentrations of less than 0.01 mg/L. The maximum recommended limit for molybdenum is 0.01 mg/L (EPA, 2012). Although it is nontoxic to plants at low levels, it can be toxic to livestock if forage is grown in soils with high molybdenum concentrations. Nevertheless, at 1,000–2,000 ppm molybdenum, tomato plants developed golden yellow color in their leaves, and seedlings of cauliflower turned purple (Samantaray *et al*., 1998). Deficiency levels for plants are usually indicated by less than 0.10 ppm in their tissues.

4.6.19 Nickel

The maximum recommended limit for nickel is 0.2 mg/L (EPA, 2012; Figure 4.20). It is toxic to a number of plants at 0.5 to 1.0 mg/L. Nickel can be toxic to plants even at low concentrations. A concentration of 40 ppm (dry weight) in tomato plants was toxic, and 150 ppm stopped its growth. Similar levels were also toxic to corn and tobacco. At week 7, several elements experiences increased levels (Section 4.6.22). This may be due to a flush of those metals in the tested batch.

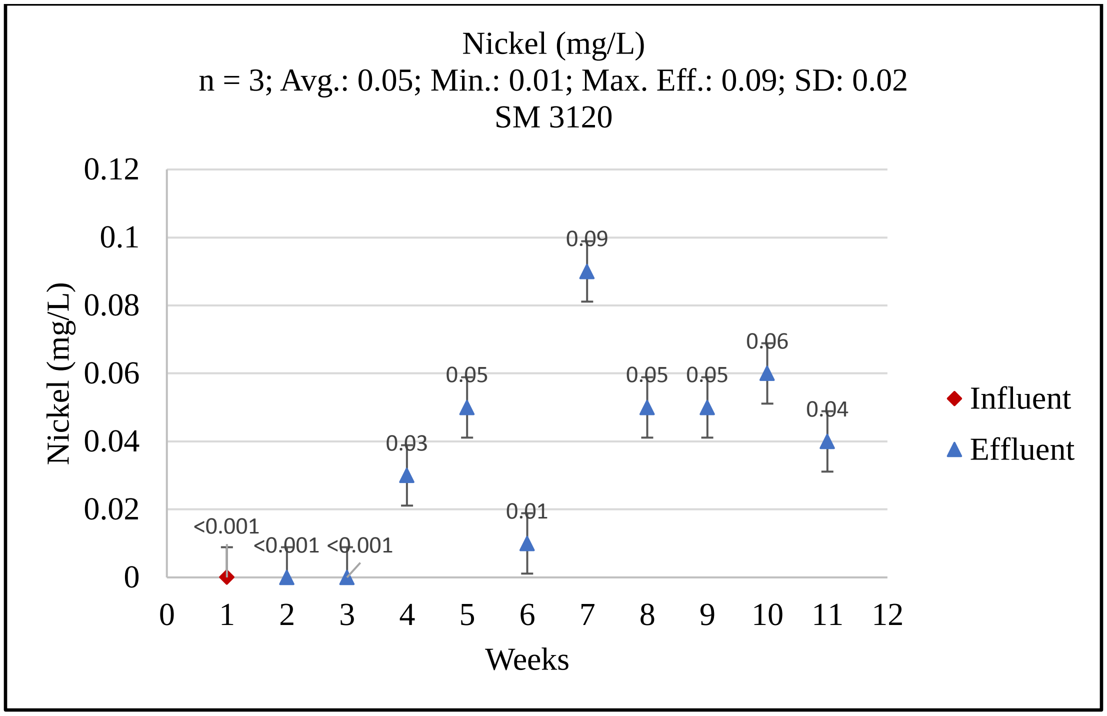

Figure 4.20 Analysis of Nickel in the Biochar-Modified Soil Aquifer Treatment System.

4.6.20 Selenium

All tested samples of the BMSAT system had Selenium concentrations of less than 0.005 mg/L. The maximum recommended limit for selenium is 0.02 mg/L (EPA, 2012). Acid conditions minimises solubility of Se in soils. Under neutral or alkaline conditions, selenate ion is formed which does not form any insoluble salts.

4.6.21 Vanadium

All tested samples of the BMSAT system had Vanadium concentrations of less than 0.005 mg/L. The maximum recommended limit for vanadium is 0.1 mg/L (EPA, 2012). Vanadium is toxic to germinating seeds, but even more toxic at later stages of growth. The toxicity symptoms exhibited by these plants were extreme dwarfing and chlorosis.

4.6.22 Zinc

The maximum recommended limit for zinc is 2 mg/L (EPA, 2012; Figure 4.21). Zinc is an essential plant micronutrient, and is added at a rate of 0.5 mg/L in nutrient solutions. Typical levels of toxicity in soil are more than 200 mg/kg.

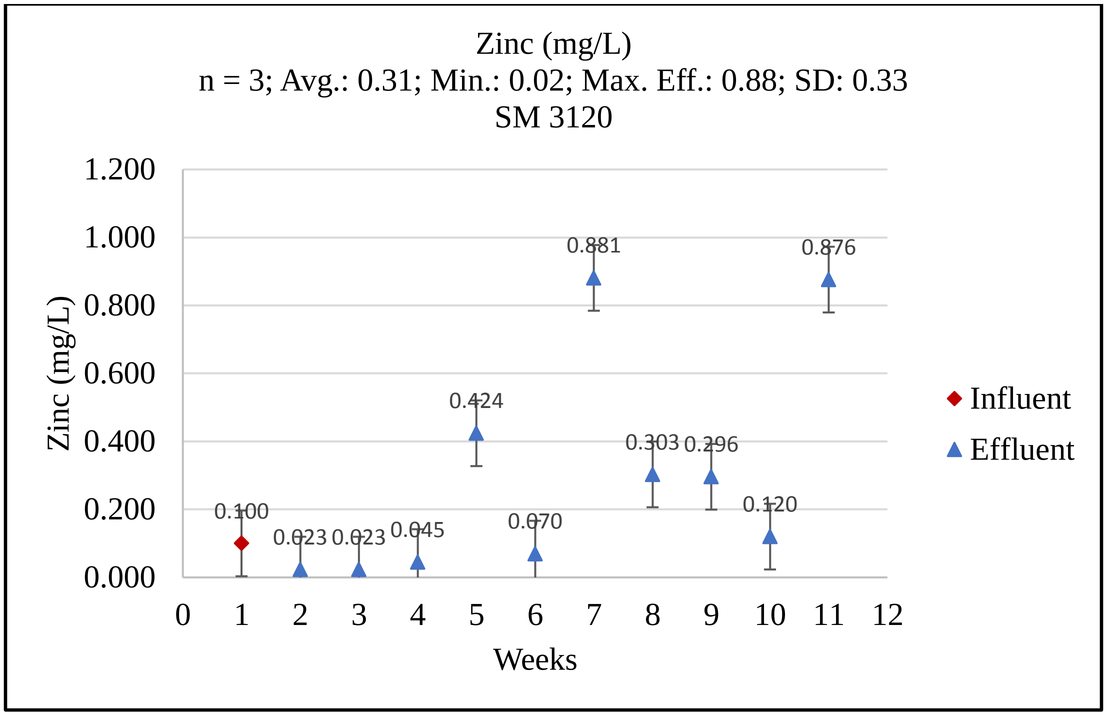

Figure 4.21 Analysis of zinc in the Biochar-Modified Soil Aquifer Treatment System.

4.7 Conclusions and recommendations

1. The BMSAT system performed exceptionally well in improving the microbiological quality of treated effluents. *E. coli* concentration in the final stage was significantly below the recommended limit of 1000 CFU/100 mL for unrestricted irrigation.
2. All tested metals were below the recommended limits of treated effluents for reuse in agriculture. The BMSAT system significantly reduced certain metals and ions such as aluminum and fluoride.
3. Due to the nitrification and denitrification processes in the BMSAT system, total nitrogen concentration in effluents are considered typical of activated sludge wastewater treatment plants.
4. pH remained within the normal range of treated effluents from activated sludge wastewater treatment plants, and waste stabilization ponds. Therefore, BMSAT system could provide a competitive alternative to these systems, especially in rural areas.

5. Boron concentrations increased in the BMSAT system, but remained below the maximum limit for irrigation.
6. SAR was initially reduced in the BMSAT system, then increased but remained significantly below the maximum limit for irrigation.
7. Electrical conductivity increased in the BMSAT system, but remained below the maximum limit for irrigation.
8. The BMSAT system could be a cost-efficient alternative to existing wastewater treatment systems, especially OSSs. Treated effluents were suitable for reuse restricted and unrestricted irrigation of crops.

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Chapter 5

Conclusions and Recommendations

5.1 Summary

This thesis made progress in our understanding of wastewater governance and management (e.g. treatment and reuse). Chapter 2 critically reviewed some of the most pressing causes to reuse water, and examined the potential of treated wastewater application on land as a measure to alleviate issues associated with current practices. It provided better understanding of the impacts of current treated wastewater management actions on ecosystems, human and animal health, and the economy at large, with emphasis on the Great Lakes basin. It then examined suitability of treated wastewater reuse from financial, social, and institutional perspectives.

Wastewater governance presents a challenge to Ghana, as is the case in many other developing countries. There are economic benefits and institutional challenges related to applying treated wastewater to agricultural land. A review was presented on treated wastewater status and current characteristics of wastewater treatment plants. Chapter 3 developed a hierarchy of the main challenges to treated wastewater reuse in agriculture, and developed better understanding of the scale of various challenges using the analytical hierarchy process as a decision-making tool. Results were summarized in Tables 3.11 and 3.12. The chapter then provided informative evidence on treated wastewater toxicological suitability for reuse in agriculture. Toxicological assessment determined WSPs performed as well as activated sludge plants, and effluents were suitable for the proposed end use.

Millions of OSSs, commonly known as septic tanks, in Canada and the United States pose a threat to ground water quality and natural surface water bodies, and have the potential to elicit an effect on human health over the long term. OSSs serve more than 50% of new housing in some areas of the Great Lakes area, and at least 40% of existing systems fail to treat wastes adequately. Biochar-modified SAT (BMSAT) system was investigated for its toxicological and microbiological performance as per the 2012 U.S. Environmental Protection Agency guidelines for water reuse in agriculture. Raw wastewater, containing industrial and domestic discharges, was collected from a local wastewater treatment plant which is located within a Highly Vulnerable Aquifer Area, and is within a Significant Groundwater Recharge Area. The new BMSAT system presents a practical application for rural areas where OSSs mainly exist.

5.2 Conclusions and recommendations

- Two of the most pressing causes to reuse treated effluents are eutrophication and the emergence of PPCPs.
- Aquatic organisms are exposed to aqueous contaminants, including PPCPs, throughout their entire lifetimes, leading to feminization of fish, altered fish behaviour, and endocrine disruption, even at nanogram per litre concentrations.
- Treated wastewater discharge to streams and rivers can promote the spread of ARBs and ARGs. The impact can be as far as 20 km downstream to the point discharge. Chlorination may not be effective.
- Advanced treatment systems, such as ozonation and filtering techniques, can not fully eliminated ARBs and ARGs. Prioritization of the issue and mobilized action to deliver a scale of change are needed.
- In humans, PPCPs and endocrine disruptors could inhibit the action of hormones, or alter the normal regulatory function of the endocrine system.
- It may not be affordable to address PPCPs by only upgrading wastewater treatment plants.
- The application of treated wastewater on land could present a new multi-barrier approach to manage PPCPs, and could likely allow natural processes (such as biodegradation, photodegradation, physical adsorption to soil and sediments, and plant uptake) to more integrally alleviate the issue. However, the combined effect of these processes requires further investigation.
- Sewage effluents may well provide a greater risk of river eutrophication than diffuse sources from agricultural land. Phosphorus is the limiting factor for the growth of algae in lakes.
- Hypoxia and HABs can cause morbidity and mortality of birds and marine mammals, kill fish or shellfish directly, cause loss of submerged vegetation, and affect aquaculture and biodiversity. HABs poisoning, including paralytic shellfish

poisoning, ciguatera fish poisoning, diarrheal shellfish poisoning, neurotoxic shellfish poisoning, amnesic shellfish poisoning, has been reported in humans and animals.

- The socio-economic consequences of sewage-driven eutrophication can be difficult to quantify. However, the cost of eutrophication on drinking water treatment infrastructure, fishing and recreational industries, and public health are likely prohibitive. The willingness to pay for eutrophication generally exists.
- The source-pathway-recipient pollution control implies that direct discharge of treated wastewater to water courses is a shorter path to human than application on land.
- Recent federal strategies have set ambitious targets. Measures, such as producer-responsibility and feed-in tariffs, could not only contribute to achieving these targets, but also promote resource recovery.
- In Ghana, treated effluent generated from the Lagon waste stabilization pond is suitable from a toxicological perspective for reuse in agriculture. This applies to almost all crops.
- pH was within the normal range of treated effluents, whereas EC was significantly lower than average concentrations in effluents, possibly due to the rainy season. Alkalinity and salinity met requirements for reuse in agriculture. Rainfall likely resulted in swings of SAR and pH levels in effluents.
- SAR threshold was different for Canada and FAO. However, the ponds met both limits.
- For total nitrogen, performance of the studied ponds in removing nitrogen matched conventional activated sludge treatment systems.
- The BMSAT system proved efficient in removing most metals and ions. However, a few elements had higher concentrations in treated effluents.
- The BMSAT system performed notably well in improving treated wastewater microbiological quality. Treated effluents were suitable for restricted and unrestricted irrigation of crops.
- The BMSAT effluents met the 2012 U.S. EPA guidelines for reuse of treated wastewater in agriculture.

5.3 Future work

4. In environmental economics, a study on the willingness to pay index (WPI) for reduced eutrophication would provide insight to regulators and practitioners in Canada into the issue, and give an indication of perceptibility and elasticity of demand.
5. The Analytical Hierarchy Process could be utilised for social dimensions (as the main criterion) of treated wastewater reuse in agriculture.
6. Life cycle assessment of the Golden Rule, as defined in Section 3.4, would help determine externality costs of different governance modes.
7. An assessment of the microbiological quality of treated wastewater in the tested local stabilisation pond can determine its suitability for restricted or unrestricted irrigation.
8. The synergistic behaviour of PPCPs in soil could provide insight into their degradation over the long term.
9. The BMSAT system could be re-configured to include increased depth of biochar. This will likely further improve treated wastewater microbiological and toxicological quality.

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