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Full Length Research Paper

Assessment of the effect of effluent discharge from coffee refineries on the quality of river water in South-western Ethiopia

Tadesse Mosissa Ejeta[1]* and Alemayehu Haddis[2]

[1]Department of Natural Resource Management, College of Agriculture and Veterinary Medicine, Jimma University, Jimma, Ethiopia.
[2]Department of Environmental Health, Jimma University, Jimma, Ethiopia.

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The ecohydrological quality of water resource of Ethiopia is declining at an alarming rate, resulting in severe environmental degradation. This study finds out the effects of effluent discharge from intensive coffee refineries on river water quality based on physicochemical parameters and benthos assemblages as biological indicators. The experiment was done using complete randomized design (CRD) with three composite replicates in each refinery and on 24 river water sampling sites selected from four rivers in Limu Kosa District. A total of 72 water samples were collected from six sites: (upstream site (UPS), influent (INF), effluent (EFF), entry point (ENP), downstream one (DS_1) and downstream two (DS_2) in four rivers. Data analysis was performed by analysis of variance (ANOVA) using statistical analysis software (SAS). Spearman's median rank correlation among physicochemical and benthos assemblages as biological indicators of ecohydrological river water quality was characterized. Results reveal that there is a highly negatively significant difference in effect between the four rivers and 24 sites at $p<0.05$ and 0.01. The benthos assemblage communities of DS_2 and UPS of the ecohydrological rivers were more influenced by the effluents. Quality of DS_2 was more adversely affected compared to UPS. The alteration in river water quality parameters was more pronounced during the peak of coffee refineries. The impact of private refineries on receiving water was more significant than that of government refineries. Therefore, urgent attention should be given to the coffee refinery for effluent management options to avoid further damage to the ecohydrological river water quality using well-designed treatment technologies.

Key words: Biological indicators, benthos, ecohydrolological integrity, upstream downstream.

INTRODUCTION

Water is an essential and inevitable commodity for human growth and development than any other resource for life's sustenance. Although, the water resource of Ethiopia is declining at an alarming and accelerating rate,

*Corresponding author. E-mail: tadesse88@gmail.com. Tel: +251-0917082424.

resulting in severe environmental degradation (Beyene et al., 2011; Dejen et al., 2015). South-western Ethiopia is a major and famous coffee growing region in Ethiopia; it has a number of coffee refineries situated along the bank of rivers and/or streams with a varying degree of hydraulic gradients. Wet coffee requires considerable amount of water during processing to receive the cherries, transport them hydraulically through the pulping machine, remove the pulp, and sort and re-pass any cherries with residual pulp adhering to them. The rise in the number of wet coffee refineries has therefore resulted in the generation of enormous disposal of these wastes which are discharged unwisely into nearby natural water way that flows into rivers and/or infiltrates ground water, becoming main threat to surface and ground water qualities as reported by Dejen et al. (2015). With intensification of wet coffee refineries and rampant waste discharges into ecohydrological integrity of river water, an increased pressure on fauna and flora of ecohydrological integrity of river water bodies becomes evident. Water bodies are the primary dump sites for disposal of effluents from coffee refineries containing wide varieties of synthetic and organic wastes that are near them (Haddis and Devi, 2008; Beyene et al., 2011; Dejen et al., 2015). Water pollution is an acute problem in all water bodies, and major river water quality is the gloomy setback for development in coffee producing zone, especially in South-western Ethiopia. According to rough estimates, effluent from 1000 kg of parchment coffee is generated by wet-processing method compared to the human waste that can be generated by 3000-5600 people per day (Beyene et al. 2011). Alarmingly increasing rampant wet coffee refineries contribute to dwindling surface water quality in South-western Ethiopia to a greater extent. As a consequence, there is a risk to ecosystems structures and their functions which allow for regulation of ecosystem processes, and risk to local community health and welfare as they might take in pollutants through consumption of crops such as onion, tomato, potatoes and maize and using of this river for domestic purposes (Kassahun et al. 2010; Dejen et al. 2015).

This has often gradually rendered the ecohydrological quality of rivers of the Limu Kosa District unsuitable for various beneficial purposes as well as their maintenance and restoration. Benthos assemblages within ecological water quality are interrelated and excellent indicators of water quality; they easily respond to organic and inorganic pollution load from human interferences (Kassahun et al., 2010; Beyene et al., 2011). Few, if any, studies have investigated this issue in Ethiopia to assess the effect and extent of the problem and to suggest solutions and recommendations accordingly. Virtually, no studies have specifically addressed the spatial variation of different ecohydrological integrity of river water quality based on the physico-chemical parameters and benthos assemblages as biological indicators of receiving water bodies of South-west Ethiopia. The objective of this study is to determine the effect and extent of effluents generated from coffee refineries on ecohydrological integrity of river water quality based on the physicochemical parameters and benthos assemblages as biological indicators of river water quality in Limu Kosa District (Figure 1).

MATERIALS AND METHODS

Descriptions of the study area

The study was conducted in Limu Kosa District of Jimma Zone (Figure 1). Limu Kosa District is located 420 km southwest of Addis Ababa, the capital city of Ethiopia, lying between Latitude of 7°50 and 8°36′ North and Longitude of 36°44′ and 37° 29′ East. The altitude of district ranges from 1200 to 3020 m above sea level. It has an area of 2770.5 km^2. Several perennial rivers (Gibe, Awetu, Kebena, Ketalenca, Bonke and Dembi), intermittent streams, springs and notable landmarks including Cheleleki Lake and Bolo Caves were found in the Limu Kosa District (data from the Limu Kosa District Agricultural and Rural Development Office). The availability and quality of river water not only impact human health and wellbeing, but also the functioning of essential ecosystems, including rivers, wetlands, lakes and coastal ecosystems. Without sound ecohydrological of river basin management, human activities can upset the delicate balance between ecohydrological integrity and environmental sustainability. As might be expected, water quality in Limu Kosa District rivers and wetlands ranges from absolutely pristine to dangerously poor.

Methods

Study period

A cross sectional study was conducted to assess the impact of wastewater discharge on ecohydrological river water quality by coffee refineries in Limu Kosa District from August 2011 to December 2013. During the whole study period, the primary data (three days of a week from the chosen sampling points) were collected through direct measurement of river water quality parameters of the selected study sites *in-situ* and under laboratory condition.

Experimental design of the study and selection of sampling sites

The experiment was conducted using complete randomized design (CRD) with three composite replicates to minimize the variation of all sample collected from the same sample site. In order to assess the ramification of coffee refineries effluent being discharged, physico-chemical samples were taken from the 24 ecohydrological river water sites (12 among each private and government refineries). Six sampling sites were selected for physico-chemical samples along each ecohydrological river. These sites were upstream site (UPS), influent (INF), effluent (EFF), entry point (ENP), downstream one (DS_1) and downstream two (DS_2). In order to understand the influence of effluent discharge by coffee refineries on ecohydrological river water quality, benthos assemblages as biological indicators of river water quality samples were also taken from the upstream (UPS) and downstream two

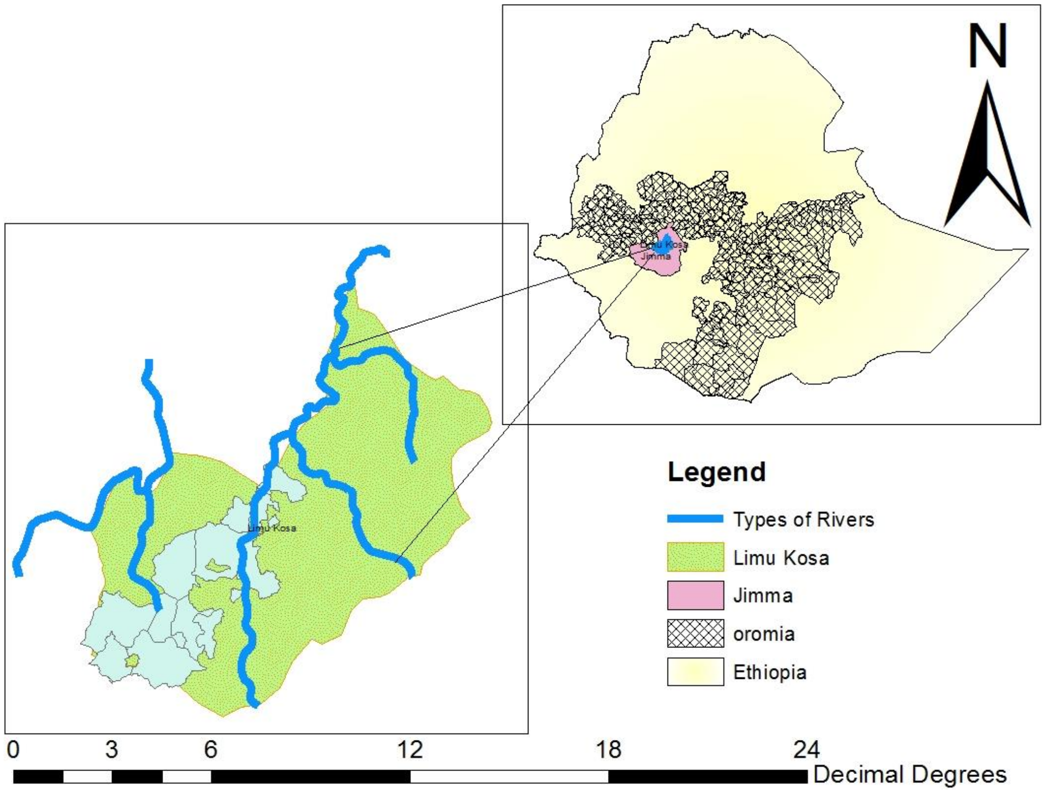

Figure 1. Map of Kosa District indicating sampling sites.

(DS_2) of the discharge points of ecohydrological rivers. UPS was the control sites without any effects from the effluent because of their sites. Influent (INF) was the point at which waste water enters the treatment plants; in this case lagoon. Effluent (EFF) is wastewater leaving the lagoon before it enters the river water. Entry point (ENP) is highly impacted; it is located after the EFF and the point at which lagoon effluent enters the river. Downstream one (DS_1) is located 500 meters below ENP. Downstream two (DS_2) is located 500 meters below DS_1. The aim of taking samples at different sites of the downstream is to analyze spatial variations and determine the rivers' recovery potential. At each sampling point, three samples were taken cross sectionally (corners and center) and three similar sampling campaigns were conducted. This makes the total analyzed samples 180. The distance between UPS, ENP, DS_1 and DS_2 was set at an interval of 500 m. Also, samples were taken from INF and EFF. No actual distance was determined because it depends on the coffee refineries designed. Specially, these wastewater samples were collected at the peak hours of coffee refineries three days in a week from the chosen sampling points (Figure 2) (Kobingi et al., 2009; Kassahun et al. 2010; Akali et al., 2011; Dejen et al. 2015).

Sampling procedure of physicochemical parameters data

Samples were collected in sterilized plastic BOD and glass bottles to maintain accuracy or minimize contamination of physicochemical changes that can occur between time of collection and analysis as indicated in APHA standard method (APHA) (2005). The water samples were collected by inserting the plastic and glass bottles to the opposite direction of the river flow and capped tightly immediately after filling to the tip of the mouth of this bottle by using depth-integrated sampling technique. Determinations of pH, EC, temperature, turbidity and DO fixing were carried out *in-situ* as APHA (2005). These samples were properly and carefully labeled, sealed and transported to the laboratory of the Department of Environmental Health Sciences and Technology, Jimma University. Cold storage was maintained throughout the process till analysis.

Sampling method of macro-invertebrates (benthos) from river water sites

A triangular D-frame Dip-Net (mesh size = 500 µm, sampled area = 0.9 m^2) was used to collect benthos by kick sampling method. In this method, the river bed was disturbed for a distance of about 100 m for 3-5 min. Benthos sample was conducted three times from each riffle and run sample site. These samples were properly and carefully labeled, sealed and transported to the laboratory of the Department of Environmental Health Sciences and Technology, Jimma University, Jimma, Ethiopia. Cold storage was maintained throughout the process till analysis. Identification to a family level was done using a compound light microscope and assisted by a standard identification key (Bouchard 2004; Kobingi et al., 2009).

Statistical analysis

The data were subjected to different statistical analysis such as analysis of variance (ANOVA) using SAS version 9.2, Minitab

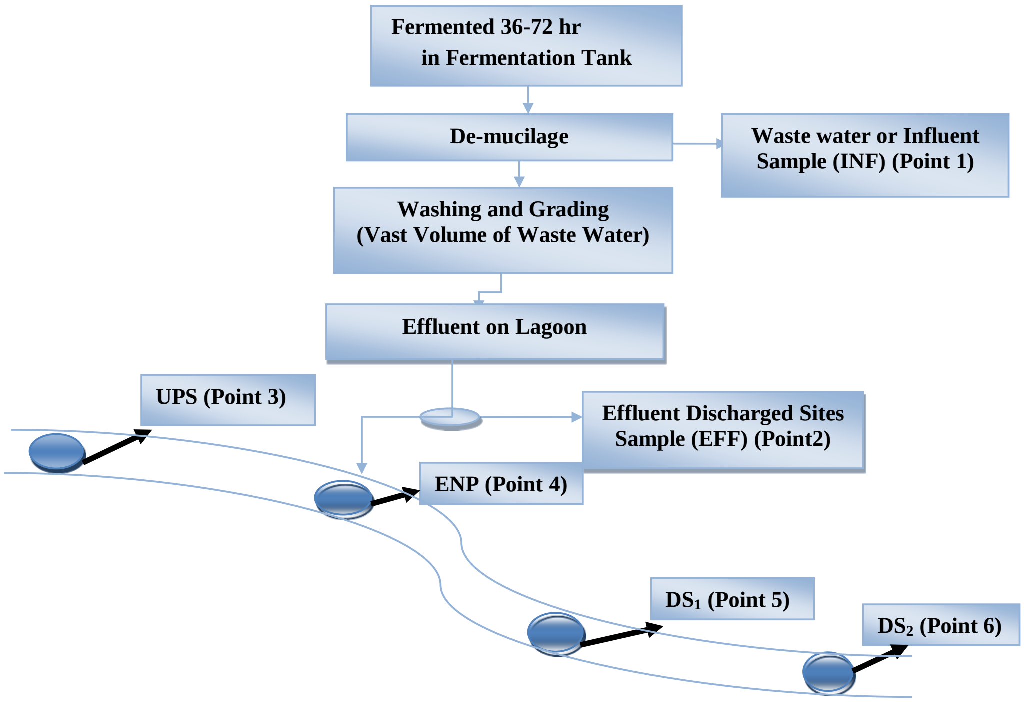

Figure 2. Map indicating general flow diagram of coffee refinery and effluent sampling points.

Version 16.0 software and MS Excel. When significant interaction effects were observed among the four rivers with river water and sites using a two-way ANOVA, One-way ANOVA was computed to see significant difference between each sample site for the physico-chemical parameters and benthos assemblages as biological indicators. Mean separation of different sources of variation among each river water and site was done using Tukey's test at α = 0.05 level of minimum significance difference (MSD). Pearson correlation matrix analysis was used to reveal the magnitude and direction of relationship between different physic-chemical parameters within and among benthos assemblages as biological indicators of river water quality. Benthos assemblages as biological indicators of ecohydrological river water quality samples were determined by using benthos assemblages multimetric indices.

RESULTS

Physical parameters and their significance level in four river water of the study sites

The average mean values of river water temperature ranged between 12.11±0.78-43.09±0.78˚C at Kebena UPS and Awetu EFF respectively. This result showed that there was highly significant difference in all sampling sites, but very high 43.96°C in the Awetu EFF, indicating much stress from the coffee refineries disposal at $p<0.05$ and 0.01. There was highly significant difference in the concentration of EC among the four river water and sites at $p<0.05$ and 0.01. The average mean values of EC ranged from 167.65±15.38-1187.26±15.38µS/cm among all sites. DS_1 to DS_2 exhibited non- significant variation of EC and TDS in contrast to other sites. The EC alarmingly increased with increase in TDS and water temperature (Tables 1 and 2).

The observed turbidity mean values ranged from 3.3±11.05-1363.67±11.05 NTU at Bonke UPS and Kebena INF respectively. The maximum average mean value obtained from the polluted sites (1397NTU) was higher than 2.86NTU recorded at UPS. The turbidity mean concentration at DS_1 to DS_2 was 114.10±11.05 - 980.58±11.05NTU which significantly exceeded the allowable limit set by WHO and EPA (10 mg/L). Consequently, various analytical mean values of TSS and TDS fluctuated between 756.35 ± 15.31 - 1063.35 ± 15.31 mg/L to 394.14 ± 15.31 - 342.09 ± 15.31 mg/L and 1095.64 ± 53.71 - 1197.37 ± 53.71 mg/L to 435.26 ± 53.71 - 481.92 ± 53.71 mg/L amongst the polluted sites of Kebena and Ketalenca DS_1 to DS_2, respectively. These mean values of TSS and TDS obtained from the polluted sites were higher than 16.79 ± 15.31 - 10.02 ± 15.31 mg/L to 302.96 ± 53.71 - 235.04 ± 53.71 mg/L recorded at Kebena and Ketalenca UPS, respectively. There were highly significant differences ($p<0.05$ and 0.01) in the values of TSS among the different sampling sites across the river water course. These results show significant increased values from DS_1 to DS_2 sites of the river water in TSS, but non- significant differences from DS_1 to

Table 1. Physico-chemical parameters selected for the study site and techniques used for methods of analysis.

Physico-chemical parameters	Abbreviation	Methods of analysis	Unit
Water temperature	WT	Probes multi parameter methods	°C
Turbidity	TURB	Turbidity meter	NTU
Electrical conductivity	EC	Probes multi parameter methods (EC meter)	µS/cm
pH	pH	Probes multi parameter methods (pH meter)	-
Total dissolved solids	TDS	Gravimetric Method, dried at 180°C	mg/L
Total suspended solid	TSS	Gravimetric Method, dried at 103-105°C	mg/L
Total solid (TS)	TS	Gravimetric Method, dried at 103-105°C	mg/L
Dissolved oxygen (DO)	DO	Probes multi parameter methods (DO meter)	mg/L
Biological oxygen demand	BOD_5	The Azide Modification of the Winkler Method	mg/L
Chemical oxygen demand	COD	Kit (Hachlange cuvette test, LCk 614 &114)	mg/L
Nitrate-nitrogen	NO_3-N_2	Phenoldisulfonic Acid Method	mg/L
Ammonia-nitrogen	NH_3-N_2	Direct Nesslerization Method	mg/L
Total nitrogen	TN	Kit (Hachlange cuvette test, LCK 138 & 338)	mg/L
Orthophosphate	Orth-P	Stannous Chloride Method	mg/L

Source: APHA, 2005.

DS_2 in TDS (Table 2).

Chemical parameters and their significance level in four river water of the study sites

The average mean values of pH in all six sites of river water were acidic and ranged between 3.12 ± 0.10-7.67 ± 0.10 at Kebena EFF and Awetu UPS respectively. The lowest values obtained from the EFF (2.9) were very lower than 7.93 recorded at UPS. Acidity was found to be potent at ENP than DS_1 which in turn was stronger than DS_2 (Table 3). The pH has shown significant differences among DS_1 and DS_2 river water at p< 0.05 and 0.01.

The average mean values of DO were fluctuated between 0.00±0.10 to 8.04±0.10 mg/L in river water samples collected among the four river water with river water and sites. Kebena EFF and INF showed the lowest value of DO as 0.00±0.10 mg/L. These variations may be attributed to oxygen consumption by aerobic organisms due to increase in oxygen demanding wastes. Level of DO in the river water was almost normal in the UPS. DO concentrations below 5 mg/L may also adversely affect the functioning and survival of biological communities and hence all pollution-sensitive taxa failed to retrieve (Table 3). There were highly significant inconsistencies of interaction effect of BOD and COD among all river waters at (p<0.05 and 0.01). The maximum average mean values of BOD and COD were recorded (2972.67±30.27 to 2576.05±30.37 mg/L) at Kebena EFF and INF;h minimum values were recorded (2.36 ± 30.27 to 3.99 ± 30.37 mg/L) at Ketalenca and Bonke UPS. BOD and COD showed alarming increment from 1773 ± 30.27-1719.83 ± 30.37 mg/L to 1797.89 ± 30.27-1836.40 ± 30.37 mg/L at Kebena DS_1 and $DS_{2;}$ then decreased slowly towards the rest of the Ketalenca and Bonke of DS_1 and DS_2 respectively.

TN concentration analysis revealed that there was highly significant difference in interaction effect among the four rivers but not at Kebena River of ENP, DS_1, DS_2 and Bonke ENP as well as EFF and PUS of all river water at p ≤ 0.05 and 0.01. This is due to highly mobility or fixation of TN concentration among each river water site. The concentrations of NO_3-N and NH_3-N_2 in the river water were found to be statistically highly significant and the average mean values ranged from 2.43±0.03 to 4.99±0.07 mg/L. They were higher concentrations in all INF and alarmingly increment from DS_1 to DS_2 due to high coffee refineries' activities that ultimately discharge almost untreated effluent to the river (Table 3). The average mean values of orthophosphate (Orth-P) showed significant difference in all river water, but not DS_1 and DS_2 in all river water (Table 3).

Benthos assemblages as biological indicators of river water quality

UPS and DS_2 benthos assemblages of fauna from 8 taxonomic orders were collected from Limu Kosa District rivers. A total of 30 families fewer than 8 orders representing classes and comprising 1293 individuals were collected from the eight sampling sites. A total number of individuals found in the DS_2 were 387 compared to 906 individuals collected from the UPS. The pollution sensitive taxa of Ephemeroptera, Hemispheres, Trichoptera, Plecoptera and Coleoptera were present in greater number in the UPS. On the other hand, pollution

Table 2. Interaction effects of effluent discharges by coffee refineries on physical characteristics between the all river waters and among sites of river water.

River	Mean separation of physical parameters						
	Site	TSS	TDS	TS	EC	TURB	WT
Kebena	EFF	1800.35^{A}	2239.30^{B}	4039.64^{BA}	1045.80^{B}	1335.23^{A}	28.12^{EF}
	INF	1527.23^{B}	2681.23^{A}	4208.46^{A}	1160.68^{A}	1363.67^{A}	37.267^{B}
	ENP	1460.03^{CB}	2052.26^{B}	3512.29^{ED}	858.65^{C}	1190.48^{B}	24.27^{FIHG}
	DS_2	1063.35^{D}	1197.37^{E}	2260.72^{HG}	661.09^{D}	980.58^{C}	19.60^{JK}
	DS_1	756.35^{E}	1095.64^{E}	1851.98^{JI}	616.73^{D}	972.10^{C}	18.67^{K}
	UPS	16.79^{I}	302.96^{GH}	319.75^{M}	188.65^{H}	3.99^{H}	12.11^{L}
Awetu	EFF	1778.87^{A}	1508.64^{DC}	3287.51^{EF}	1035.56^{B}	1195.25^{B}	43.09^{A}
	INF	1126.52^{D}	2773.59^{A}	3900.1^{BAC}	1187.26^{A}	1188.10^{B}	36.75^{B}
	ENP	586.98^{F}	1537.99^{C}	2124.97^{HI}	844.00^{C}	675.94^{D}	34.97^{CB}
	DS_2	431.65^{G}	762.07^{F}	1193.72^{K}	505.65^{E}	514.38^{E}	25.40^{FHG}
	DS_1	434.23^{G}	753.82^{F}	1188.05^{K}	513.28^{E}	514.56^{E}	29.747^{ED}
	UPS	33.24^{I}	335.24^{GH}	368.48^{M}	197.94^{H}	6.99^{H}	20.08^{JIK}
Bonke	EFF	757.29^{E}	2298.43^{B}	3055.72^{F}	890.99^{C}	1316.66^{A}	37.82^{B}
	INF	1382.24^{C}	2202.67^{B}	3584.9^{EDC}	1151.17^{A}	1202.01^{B}	37.27^{B}
	ENP	578.45^{F}	1227.24^{DE}	1805.68^{J}	582.78^{ED}	520.62^{E}	36.86^{B}
	DS_2	543.76^{F}	577.02^{F}	1120.78^{LK}	395.69^{F}	128.70^{G}	23.64^{JIHG}
	DS_1	584.03^{F}	569.84^{GF}	1153.87^{K}	393.62^{F}	128.39^{G}	35.77^{B}
	UPS	28.07^{I}	230.00^{H}	258.07^{M}	167.65^{H}	3.30^{H}	27.32^{FEG}
Ketalenca	EFF	1381.05^{C}	1177.33^{E}	2558.38^{G}	1015.70^{B}	1352.37^{A}	31.29^{CED}
	INF	1051.68^{D}	2746.18^{A}	3797.86^{BDC}	1179.37^{A}	1237.58^{B}	33.95^{CBD}
	ENP	419.74^{HG}	753.73^{F}	1173.47^{K}	318.35^{GF}	334.88^{F}	30.10^{ED}
	DS_2	342.09^{H}	481.92^{GFH}	824.02^{L}	240.14^{GH}	122.30^{G}	21.36^{JIHK}
	DS_1	394.14^{HG}	435.26^{GH}	829.39^{L}	226.71^{H}	114.10^{G}	19.60^{JK}
	UPS	10.02^{I}	235.04^{H}	245.06^{M}	169.10^{H}	5.12^{H}	14.28^{L}
	Mean	770.34	1257	2028	647.77	683.64	28.31
	Max	1812	2816	4302	1227.16	1397	43.96
	Min	9.70	222.27	236.84	165.43	2.86	11.70
	WHO	500	1000	500	1000	10	-
	CV (%)	3.44	7.39	4.98	4.11	2.80	4.78
	MSD	83.45	292.79	318.08	83.85	60.24	4.26
	SEM(±)	15.31	53.71	58.35	15.38	11.05	0.78
	River	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	River*Site	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001

CV, Coefficient of variation in percent; MSD, minimum significance difference at 5 and1%; SEM, standard error mean. Mean with different letters in the same column were significantly different (withTukey's test at 5 and 1% level of probability) as established by MSD test. Except EC (µS/cm), TURB (NTU) and WT (°C) the others parameters were expressed in mg/L. These six river sites were averages among each site. Awetu and Kebena river water from private and the other two were from the government refineries. Significant interactions and main effects were explored by Tukey's test, using the GLM procedure at P<0.05 and 0.01as established by MSD test.

tolerant species of families Chironomidae, Simuliidae and Leeches present in greater number in the DS_2 sections throughout the experimental period reflected the coffee refineries' stresses of the ecological status of rivers in its DS_2 sections (Appendix Table 1).

Analysis of the results of benthos assemblages as

Table 3. Interaction effects of effluent discharges by coffee refineries on chemical water quality between the all ecohydrological river waters and among sites of river water.

River	Mean separation of chemical parameters							
	Site	**pH**	**BOD**	**COD**	**DO**	**TN**	**NO_3-N**	**NH_3-N**	**Ort-P**
Kebena	EFF	3.12^{I}	2972.67^{A}	2735.50^{A}	0.00^{H}	98.40^{A}	3.36^{C}	7.04^{C}	13.18^{E}
	INF	3.33^{HI}	2689.67^{B}	2576.05^{A}	0.01^{H}	92.60^{BA}	3.86^{A}	8.11^{A}	22.90^{A}
	ENP	3.36^{HI}	2478.88^{C}	1940.57^{B}	0.02^{H}	78.61^{DE}	3.08^{D}	6.92^{DCE}	10.87^{F}
	DS_2	4.06^{DGEF}	1797.89^{E}	1836.40^{CB}	0.05^{H}	76.22^{DE}	2.81^{E}	6.83^{DCE}	10.83^{F}
	DS_1	4.28^{D}	1773.00^{FE}	1719.83^{C}	0.07^{H}	76.66^{DE}	2.74^{FE}	6.65^{DE}	10.34^{F}
	UPS	7.43^{A}	6.70^{I}	4.57^{G}	8.04^{A}	0.31^{K}	0.03^{J}	0.07^{K}	0.34^{I}
Awetu	EFF	3.59^{HGIF}	2254.95^{D}	1850.27^{CB}	0.11^{H}	88.72^{BC}	3.09^{D}	7.00^{DC}	11.47^{F}
	INF	3.31^{HI}	2205.32^{D}	1982.94^{B}	0.12^{H}	94.57^{BA}	3.60^{B}	7.49^{B}	20.37^{B}
	ENP	3.71^{HGEF}	1868.24^{E}	1525.88^{D}	1.49^{E}	82.56^{DC}	2.67^{FE}	6.93^{DCE}	11.00^{F}
	DS_2	4.12^{DEF}	1010.05^{HG}	1020.21^{FE}	3.16^{C}	35.14^{H}	2.64^{F}	6.63^{FE}	10.82^{F}
	DS_1	4.20^{DE}	989.30^{HG}	1035.08^{FE}	3.33^{C}	21.10^{J}	2.64^{F}	6.63^{GF}	10.40^{F}
	UPS	7.67^{A}	9.75^{I}	8.96^{G}	6.64^{B}	5.44^{K}	0.66I	0.06^{K}	0.91^{I}
Bonke	EFF	4.15^{DEF}	1849.67^{E}	1451.67^{D}	1.23^{FE}	96.02^{A}	2.98^{D}	5.40^{H}	15.40^{D}
	INF	3.55^{HGI}	2201.63^{D}	1835.09^{CB}	0.14^{H}	72.47^{FE}	3.97^{A}	6.01^{G}	17.56^{C}
	ENP	4.96^{C}	1129.35^{G}	1163.20^{E}	2.15^{D}	77.62^{DE}	2.78^{FE}	6.14^{G}	13.79^{E}
	DS_2	5.69^{B}	1030.60^{HG}	928.69^{F}	3.40^{C}	13.06^{I}	1.35^{H}	3.83^{J}	8.28^{G}
	DS_1	5.57^{B}	992.55^{HG}	961.88^{F}	3.55^{C}	41.52^{H}	1.95^{G}	3.81^{J}	8.19^{G}
	UPS	7.52^{A}	4.34^{I}	3.99^{G}	6.14^{B}	4.47^{K}	0.66^{I}	0.07^{K}	0.13^{I}
Ketalenca	EFF	3.48^{HI}	1618.17^{F}	1488.03^{D}	0.73^{FG}	95.29^{A}	3.09^{D}	5.12^{H}	8.25^{G}
	INF	4.55^{DC}	1717.18^{FE}	1551.15^{D}	0.25^{HG}	66.36^{F}	3.60^{B}	6.29^{FE}	10.60^{F}
	ENP	4.60^{DC}	1109.83^{G}	1014.92^{FE}	2.19^{D}	50.09^{G}	2.64^{F}	4.62^{I}	8.84^{G}
	DS_2	5.90^{B}	902.88^{H}	874.23^{F}	3.64^{C}	19.88^{I}	1.51^{H}	3.83^{J}	4.86^{H}
	DS_1	5.66^{B}	1009.38^{HG}	912.93^{F}	3.27^{C}	35.31^{H}	1.96^{G}	4.26^{I}	5.84^{H}
	UPS	7.52^{A}	2.36^{I}	9.74^{G}	8.01^{A}	5.87^{K}	0.66^{I}	0.05^{K}	0.34^{I}
	Mean	4.80	1401	1268	2.40778	55.35	2.43	4.99	9.81
	Max	7.93	2993	2867	8.31	99.23	3.99	8.37	23.31
	Min	2.90	2.03	3.19	0.00	0.30	0.03	0.05	0.13
	WHO	65-8.5	10	40	6	-	10-45	0.2-5	5
	CV (%)	6.03	6.74	8.16	5.80	3.71	2.17	2.30	3.97
	MSD	0.56	165.01	165.57	0.52	6.48	0.17	0.36	1.23
	SEM(±)	0.10	30.27	30.37	0.10	1.19	0.03	0.07	0.23
	River	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
	River*Site	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001

CV, Coefficient of variation in percent; MSD, minimum significance difference at 5 and 1%, SEM, Standard error mean. Mean with different letters in the same column were significantly different (with Tukey's test at 5 and 1% level of probability) as established by MSD test. Except pH, the others parameters were expressed in mg/L. These six river sites were averages among each site. Awetu and Kebena river water from private and the other two were from the government refineries. Significant interactions and main effects were explored by Tukey's test, using the GLM procedure at P<0.05 and 0.01 as established by MSD test.

biological indicators illustrated a highly significant difference between rivers and all sites at p<0.05 and 0.01. These benthos assemblages would indicate the environmental effects of coffee refinery activities on the ecohydrological river water quality and its vicinity. The analysis of the average species diversity of benthos assemblages as biological indicators (Shannon, equitability and Simpson) was much reduced in the DS_2

Table 4. Summary of benthos assemblages diversity indices and taxa richness among ecohydrological rivers.

River	F		S		D		H'		E		Min		Max	
	UPS	DS_2	UPS	DS_2	UPS	DS_2	UPS	DS_2	UPS	DS_2	UPS	DS_2	UPS	DS_2
Awetu	9	11	266	157	0.88	0.85	2.15	2.09	0.98	0.89	0	0	42	42
Bonke	9	6	169	54	0.88	0.63	2.14	1.32	0.97	0.74	0	0	29	31
Ketalenca	15	8	266	127	0.92	0.58	2.62	1.31	0.97	0.63	0	0	41	80
Kebena	13	3	205	49	0.92	0.50	2.54	0.87	0.99	0.79	0	0	25	33
Total			906	387	0.90	0.64	2.36	1.40	0.98	0.76	0	0	35	47
Grand			1293		-	-	-	-	-	-	-	-	-	-
Average			647		0.77		1.88		0.87		0		41	

Table 5. Results of ANOVA for benthos assemblage composition, abundance and distribution among sites.

Site	Mean separation of diversity indices and taxa richness				
	F	S	H'	D	E
UPS	12^a	227^a	2.36^a	0.90^a	0.98^a
DS_2	7^b	97^b	1.40^b	0.64^b	0.76^b
CV (%)	29	7	19.35	11.42	6.17
MSD (0.05)	2.98	12.52	0.40	0.097	0.052
SEM (±)	0.95	4.02	0.13	0.03	0.02

F=Total number of Families, S=Total number of Richness, H'= Shannon- Wiener diversity index, D=, Simpson's diversity index E= Equitability or Evenness diversity indices. Means with different letters in the same column are significantly different (Tukey's test at $P<0.05$) as established by MSD test.

as against UPS, which was very high throughout the experimental period (Tables 4 and 5).

Pearson correlation matrix (r) among selected physicochemical parameters and benthos assemblages as biological indicators of river water quality

PH and DO exhibited that they are positively and highly significant correlated with benthos assemblages, while BOD and COD are negatively and highly correlated with benthos at $p<0.05$. Meanwhile, TN, NO_3-N and Orth-P had negative correlation with all diversity indices and taxa richness, except evenness at $p<0.05$. The richness and all diversity revealed that there is a highly significant dependence on pH and DO parameters. This suggests that a local increase in pH and DO was responsible for increase in the richness of benthos (Table 6).

DISCUSSION

The good ecohydrological status of sampling sites in the UPS of Limu Kosa District areas was indicated by high proportion of pollution sensitive benthos, whereas entry point segment received huge volume of effluents that acts as physical-chemical barrier, which restricts the movement of benthos from DS_2 to UPS and vice versa. The results showed that the physicochemical parameter of the effluent discharged from government coffee refineries into the river water (Bonke and Ketalenca river water) decreased slowly towards $DS_{2,}$ while physico-chemical parameter of the effluent discharged from private coffee refineries into the river water (Kebena and Awetu river water) alarmingly increased towards DS_2. Deterioration of the river water quality increases during the peak time of coffee refineries when rampant discharges are discharged into the river water. It could lead to reduction in volume of river water and also impede the free flow of the river water. The ecohydrological river water banks were disrupted by most processing.

High physicochemical and nutrient parameters concentration widely exceed assimilation capacity of ecohydrological integrity of river water quality and do not allow for aquatic life and complex effects on flowing river water and increased eutrophication concentration at $DS_{2.}$

Table 6. Spearman's median rank correlation among physico-chemical parameters with benthos assemblages as biological indicators of river water quality characteristics.

Parameter	pH	DO	BOD	COD	TN	NO_3.N	Orth-P	S	H'	D	E
pH	1.00										
DO	0.93**	1.00									
BOD	-0.94**	-0.97**	1.00								
COD	-0.95**	-0.95**	0.98**	1.00							
TN	-0.93**	-0.91**	0.91**	0.94*	1.00						
NO_3-N	-0.96**	-0.94**	0.89**	0.90**	0.90**	1.00					
Orth-P	-0.99**	-0.91**	0.94**	0.88**	0.81**	0.94**	1.00				
S	0.89**	0.86**	-0.86**	-0.80**	-0.65*	-0.65*	-0.78*	1.00			
H'	0.79**	0.91**	-0.88**	-0.85**	-0.72*	-0.69*	-0.72*	0.88**	1.00		
D	0.77**	0.87**	-0.88**	-0.85**	-0.71*	-0.65*	-0.69*	0.86**	0.97**	1.00	
E	0.86**	0.88**	-0.83**	-0.81**	-0.43	-0.53*	-0.60*	0.75**	0.84**	0.89**	1.00

**= Correlation are highly significant at $p < 0.05$ probability levels, *= Correlation are moderately significant at $p < 0.05$ probability levels and '-' indicate negative correlation. E = Equitability or evenness index, BOD = biological oxygen demand, COD = chemical oxygen demand, DO = dissolved oxygen, D= Simpson's diversity index, H' = Shannon-Wiener diversity index, Orth- P= orthophosphate, NO_3-N= nitrate nitrogen, S= specious richness taxa and TN= total nitrogen).

TN is not strongly adsorbed on effluent cation exchange complex. Low adsorption coefficients of waste stabilization pond lagoon and constructed wetlands effluent result in maintenance of high dissolved NH_3-Nconcentrations in the effluent river water quality (Akan et al., 2009; Akali et al., 2011).

This result indicates that the decline at an alarming and accelerating rate of ecohydrological river application benefits both watershed and their surrounding environment and society (health and welfare) deterioration. Due to drawdown river discharge (hypoxia or anoxia), increased temperatures and reduced water quality in peak time (mid-September to mid of December) of coffee refineries, the health of ecosystem is usually at stake in these months; so maintaining ecosystem health and improving biodiversity in such months is more important for water resources planners. This poses a health risk to several rural communities which rely on the receiving water bodies primarily as their sources of domestic water and for other purpose (Walakira and James, 2011). Biological indicators were strongly positive correlated with pH and DO while negative correlations were noticed in BOD and COD of river water quality. This showed that there was hypoxia or anoxia which affected taxa richness and all diversity indices (Aina, 2012a, 2012b).

Outfalls from private coffee refineries that are discharged into the river water column as well as into vicinity revealed highly significant variation of physico-chemical and nutrient characterization as compared to government site. Lagoons that were intended to serve as wastewater stabilization were neither properly constructed nor were they of the right dimension to accommodate the generated waste during peak time of refineries lead to overflow of raw effluents into natural river water column. There is need for the intervention of appropriate regulatory agencies to ensure production of high quality treated final effluents by wastewater treatment facilities in rural communities coffee refineries (Sharma and Samita 2011; Mary Joyce and Macrina, 2012).

Conclusion and recommendation

High proportion of pollution sensitive taxa of benthos assemblages (Ephemeroptera, Hemispheres, Trichoptera, Plecoptera and Coleoptera) in the UPS as against high pollution tolerant species of families Chironomidae, Simuliidae and Leeches DS_2 was recorded. Coffee refinery effluents having contaminants are intensively incorporated with river water regularly. This study clearly reveals that river water quality was found to be unfit for human consumption and other domestic purposes due to the exceeding level of physico-chemical parameters values recommended by WHO at DS_2 of Limu Kosa District. Thus the challenges to continuous physico-chemical parameters and biological indicators monitoring will be immense. Both planners, regulatory agencies & the scientific community should work together to establish sustainable coffee production that is economically viable, environmentally amendable and maintain ecological integrity of receiving water bodies. Therefore, urgent intervention in the area of coffee refinery for effluent management options should be dealt with top priority to avoid further needless damage to ecohydrological integrity, and the development of river water quality using well-designed treatment technologies (lagoons) for coffee waste treatment is highly recommended.

Conflict of Interest

The author has not declared any conflict of interests.

REFERENCE

Aina OA (2012a). Spatial and Temporal Variations in Water and Sediment Quality of Ona River, Ibadan, Southwest Nigeria. Eur. J. Sci. Res. 74(2):186-204.

Aina OA (2012b). Impact of Industrial Effluent on Water Quality and Gill Pathology of Clarias Gariepinus from Alaro Stream, Ibadan, Southwest, Nigeria. Eur. J. Sci. Res. 76(1):83-94.

Akali NM, Nyongesa ND, Neyole EM, Miima JB (2011). : An Empirical Investigation of the Pollution of River Nzoia. Sacha J. Environ. Stud. 1(1):1-30.

Akan JC, Abdulrahman FI, Ogugbuaja VO, Reuben KD (2009). Study of the Physicochemical Pollutants in Kano Industrial Areas, Kano State, Nigeria. J. Appl. Sci. Environ. Sanit. 4(2):89-102.

American Public Health Association (APHA) (2005). Standard methods for examination of water and wastewater (21st ed). Washington: American Public Health Association, American Water Works Association and the Water and Environment Federation board.

Beyene A, Kassahun Y, Addis T, Assefa F, Amsalu A, Legesse W, Helmut K, Ludwig T (2011). The Impact of Traditional Coffee Processing on River Water Quality in Ethiopia and the Urgency of Adopting Sound Environmental Practices. Environ. Monit Assess. DOI 10.1007/s10661-011-2479-7.

Bouchard RW (2004). Guide to aquatic macro-invertebrates of Effluent Discharge by Mumias Sugar Company in Kenya the upper Midwest. Water resources centre, University of Minnesota. St. Paul, MN. 208pp.

Dejen YT, Abebe BH, Taffere AW, Azeb GT (2015). Effect of coffee processing plant effluent on the physicochemical properties of receiving water bodies, Jimma zone Ethiopia. Am. J. Environ. Protect. 4(2):83-90.

Haddis A, Dev R (2008). Effect of effluent generated from coffee processing plant on the water bodies and human health in its vicinity. J. Hazard. Mater. 152: 259-262.

Kassahun Y, Kebede T, Assefa F, Amsalu A (2010). Environmental Impact of Coffee Processing Effluent on The Ecological Integrity of Rivers Found in Gomma Woreda of Jimma Zone, Ethiopia. Ecohydrology for Water Ecosystems and Society in Ethiopia. 10(2-4):259-270.

Kobingi N, Raburu PhO, Masese FO, Gichuki J (2009). Assessment of pollution impacts on the ecological integrity of the Kisian and Kisat rivers in Lake Victoria drainage basin, Kenya. Afr. J. Environ. Sci. Technol. 3 (4):097-107.

Mary Joyce LF, Macrina TZ (2012). Macro invertebrates Composition, Diversity and Richness in Relation to the Water Quality Status of Mananga River, Cebu, Philippines. Philipp. Sci. Lett. 5(2):1-11.

Sharma KA, Samita Ch (2011). Macro invertebrate Assemblages as Biological Indicators of Pollution in a Central Himalayan River, Tawi (J&K). Int. J. Biodivers. Conserv. 3(5):167-174.

Walakira P, James OO (2011). Impact of Industrial Effluents on Water Quality of Receiving Streams in Nakawa-Ntinda, Uganda. J. Appl. Sci. Environ. Manage. 15(2):289-296.

Appendix Table 1. Total number (n) of macro-invertebrates caught at four river water in Limu Kosa District. .

Taxa	Kebena				Awetu				Bonke				Ketalenca			
	UPS		DS		UPS		DS		UPS		DS		UPS		DS	
	N	%	N	%	N	%	n	%	N	%	N	%	n	%	N	%
Odonata	**37**	**18.05**	**0**	**0.00**	**91**	**34.21**	**41**	**26.11**	**67**	**39.64**	**6**	**11.11**	**74**	**27.82**	**6**	**4.72**
Coenagrionidae	10	4.88	0	0.00	37	13.91	9	5.73	23	13.61	4	7.41	22	8.27	0	0.00
Gonphidae	8	3.90	0	0.00	0	0.00	0	0.00	18	10.65	0	0.00	10	3.76	0	0.00
Libellulidae	19	9.27	0	0.00	27	10.15	13	8.28	26	15.38	2	3.70	11	4.14	6	4.72
Aeshnidae	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	31	11.65	0	0.00
Lestidae	0	0.00	0	0.00	0	0.00	11	7.01	0	0.00	0	0.00	0	0.00	0	0.00
Cordulegastridae	0	0.00	0	0.00	27	10.15	8	5.10	0	0.00	0	0.00	0	0.00	0	0.00
Hemiptera	**30**	**14.63**	**0**	**0.00**	**28**	**10.53**	**12**	**7.64**	**29**	**17.16**	**6**	**11.11**	**31**	**11.65**	**16**	**12.60**
Belostomatidae	14	6.83	0	0.00	28	10.53	12	7.64	0	0.00	0	0.00	13	4.89	5	3.94
Gerridae	16	7.80	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	18	6.77	0	0.00
Corixidae	0	0.00	0	0.00	0	0.00	0	0.00	29	17.16	6	11.11	0	0.00	11	8.66
Coleoptera	**42**	**20.49**	**0**	**0.00**	**36**	**13.53**	**0**	**0.00**	**30**	**17.75**	**0**	**0.00**	**9**	**3.38**	**1**	**0.79**
Gyrinidae	25	12.20	0	0.00	36	13.53	0	0.00	19	11.24	0	0.00	9	3.38	0	0.00
Dytiscidae	17	8.29	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	1	0.79
Elmidae	0	0.00	0	0.00	0	0.00	0	0.00	11	6.51	0	0.00	0	0.00	0	0.00
Trichoptera	**46**	**22.44**	**0**	**0.00**	**71**	**26.69**	**8**	**5.10**	**18**	**10.65**	**0**	**0.00**	**63**	**23.68**	**3**	**2.36**
Hydropsychidae	17	8.29	0	0.00	0	0.00	0	0.00	18	10.65	0	0.00	0	0.00	0	0.00
Hydroptilidae	11	5.37	0	0.00	29	10.90	0	0.00	0	0.00	0	0.00	16	6.02	0	0.00
Leptoceridae	18	8.78	0	0.00	42	15.79	0	0.00	0	0.00	0	0.00	17	6.39	0	0.00
Brachycentridae	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	12	4.51	0	0.00
Polycentropodae	0	0.00	0	0.00	0	0.00	8	5.10	0	0.00	0	0.00	0	0.00	3	2.36
Psychomyiidae	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	18	6.77	0	0.00
Diptera	**13**	**6.34**	**40**	**81.63**	**0**	**0.00**	**92**	**58.60**	**13**	**7.69**	**39**	**72.22**	**19**	**7.14**	**101**	**79.53**
Ceratopeganidae	13	6.34	0	0.00	0	0.00	0	0.00	13	7.69	0	0.00	0	0.00	9	7.09
Chironomidae	0	0.00	33	67.35	0	0.00	42	26.75	0	0.00	31	57.41	0	0.00	80	62.99
Pschodidae	0	0.00	0	0.00	0	0.00	6	3.82	0	0.00	0	0.00	0	0.00	0	0.00
Simuliidae	0	0.00	7	14.29	0	0.00	38	24.20	0	0.00	8	14.81	0	0.00	12	9.45
Tipulidae	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	19	7.14	0	0.00
Syrphidae	0	0.00	0	0.00	0	0.00	6	3.82	0	0.00	0	0.00	0	0.00	0	0.00

Appendix Table 1. Cont.

Ephemeroptera	**20**	**9.76**	**0**	**0.00**	**26**	**9.77**	**0**	**0.00**	**12**	**7.10**	**0**	**0.00**	**70**	**26.32**	**0**	**0.00**
Baetidae	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	41	15.41	0	0.00
Ephemeridae	20	9.76	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00	14	5.26	0	0.00
Heptageniidae	0	0.00	0	0.00	26	9.77	0	0.00	0	0.00	0	0.00	15	5.64	0	0.00
Caenidae	0	0.00	0	0.00	0	0.00	0	0.00	12	7.10	0	0.00	0	0.00	0	0.00
Plecoptera	**17**	**8.29**	**0**	**0.00**	**14**	**5.26**	**0**	**0.00**	**0**	**0.00**	**0**	**0.00**	**0**	**0.00**	**0**	**0.00**
Perlidae	17	8.29	0	0.00	14	5.26	0	0.00	0	0.00	0	0.00	0	0.00	0	0.00
Hirudinea	**0**	**0.00**	**9**	**18.37**	**0**	**0.00**	**4**	**2.55**	**0**	**0.00**	**3**	**5.56**	**0**	**0.00**	**0**	**0.00**
Leeches	0	0.00	9	18.37	0	0.00	4	2.55	0	0.00	3	5.56	0	0.00	0	0.00
Total	205		49		266		157		169		54		266		127	

Total # of Taxonomic order= 8 and Total # of individuals = 1293 UPS=906 DS=387

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The analysis of physicochemical characteristics of pig farm seepage and its possible impact on the receiving natural environment

Dikonketso S. Mofokeng[1]*, Rasheed Adeleke[2,3] and Olayinka A. Aiyegoro[1]

[1]Gastro intestinal Microbiology and Biotechnology, Agricultural Research Council - Animal Production Institute, Old Olifantsfontein Road, Irene, 0062, South Africa.
[2]Soil Health Unit, Agricultural Research Council - Institute of Soil Climate and Water, 600 Belvedere Street, Arcadia, Pretoria, 0001, South Africa.
[3]Unit for Environmental Science and Management, North-West University (Potchefstroom Campus) Potchefstroom 2520, South Africa.

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Pig farm seepage poses an environmental risk, considering that seepage can be generally applied on land without appropriate agronomic criteria or may accidentally spill on the natural environment. These environmental risks include increasing oxygen demand, nutrient loading of water-bodies, promoting toxic and algal blooms eutrophication, thus, leading to a destabilized environment. This research was conducted to determine the impact that the pig farm seepage may have the receiving environment based on the analyses of the physicochemical parameters of the adjacent environments. Wastewater and soil samples were collected between the periods of March 2013 to August 2013 and wastewater was analyzed for pH, temperature, Biological Oxygen Demand (BOD), Chemical Oxygen Demand (COD), Total Dissolved Solids (TDS), salinity, turbidity, Dissolved Oxygen (DO), NO_3, NO_2, and PO_4^{3-}. The results for wastewater samples for BOD (163 mg/L to 3350 mg/L), TDS (0.77 g/L to 6.48 mg/L), COD (210 mg/L to 9400 mg/L), and NO_3 (55 mg/L to 1680 mg/L), were higher than the maximum permissible limits. Results of soil samples for TDS (0.01g/L to 0.88 g/L), COD (40 mg/L to 304 mg/L), NO_3 (32.5 mg/L to 475 mg/L), and NO_2 (7.35 mg/L to 255 mg/L) were also higher than recommended limits. The results revealed that the seepage from pig farm degraded the natural environment by causing eutrophication, promote toxic and algal blooms, increase oxygen demand and thus destabilize the homeostatic balance of the receiving environment.

Key words: Physicochemical parameters, pollution, soil, wastewater, seepage, pig farm, environment.

INTRODUCTION

Agricultural activities in South Africa are advancing and increasing at an alarming rate and this may overburden the environment with organic substances from seepage mainly livestock droppings, heavy metals, fertilizers and

*Corresponding author. E-mail: mofokengd@arc.agric.za.

pesticides (Muhibbu-din et al., 2011). Mismanagement of seepage may pollute the environment with nitrogen, phosphorus, bacteriological pathogens, and parasites, which may impact negatively on the environment (Ramı´rez et al., 2005). Pollution is caused when a change in the physical, chemical, or biological condition in the environment harmfully affects the quality of the environment.

Pollution of the environment can have serious consequences, with negative impact on the aquatic life, from microorganisms to insects, birds, fish, and at the same time, the health of terrestrial animals and plants (Pachepsky et al., 2006). Land application of seepage may expose the receiving environment to pollutants, and it might become hazard and even toxic to the receiving environment (Obasi et al., 2008). Mass storage production of seepage of pig farm wastewater may also be a serious hazard for biological balance of the environment (Pachepsky et al., 2006). Most pig farms, store their seepage in lagoons for a long time and this may cause pollutants to leach through the soil and pollute ground water (Pachepsky et al., 2006).

Pig farms, also known as feedlots that house thousands of pigs, produce staggering amounts of animal seepage (Tymczyna et al., 2000). The way this seepage is stored in lagoons and used has profound effects on the natural environment (González et al., 2009). These cesspools often break, leak or overflow, sending dangerous microbes, nitrate pollution, organic and inorganic pollutants into the environment (Rufete et al., 2006). Environmental contamination by pig farm seepage can be associated with heavy disease burden and the assessment of disposal and management of this seepage is very important to safeguard the environment from pollution (Okoh et al., 2007). Monitoring the physicochemical parameters of soil and water systems is important to safely assess the environment for contamination (Singh et al., 2012). High seepage discharge or spillage is a major component of water pollution contributing to oxygen demand, nutrient loading, toxicity, eutrophication and algal blooms that destabilize the environment (González et al., 2009).

The physicochemical parameters of the receiving environment that may be affected by seepage includes pH, temperature, Electrical Conductivity (EC), salinity, turbidity, total dissolved solids (TDS), dissolved oxygen (DO), chemical oxygen demand (COD), concentrations of nitrate (NO_3), nitrite (NO_2), and orthophosphate (PO_4^3) (Muhibbu-din et al., 2011). Surrounding environments in the vicinity of pig farms may be contaminated due to fecal residues, seepage runoff and mismanagement of pig farm seepage. Thus, this may cause a threat to rivers, lakes and land surrounding the pig farms, with a significantly high contamination potential for groundwater (Villamar et al., 2011). The aim of this study is to assess the possible impacts of pig farm seepage on the natural environment by monitoring the physicochemical parameters of the seepage.

METHODOLOGY

Study area

The project was conducted at the Agricultural Research Council (Animal Production Institute). The ARC-Irene Campus is situated about 25 km south of Pretoria (25° 52′S, 28° 13′E/25.867°S 28.217°E/-25.867; 28.217) in Gauteng. The institution houses a dairy farm, pig farm, sheep farm and chicken farm.

Sampling

Wastewater samples and top soil (30 cm deep) samples were collected at the ARC-API pig farm. These samples were collected monthly from March to August 2013 between 07h00 and 09h00 A.m., on weekly basis. These samples were collected to determine their physicochemical parameters namely BOD, COD, Salinity, pH, Temperature, EC, TDS, Turbidity, DO, concentrations of NO_3, NO_2, and PO_4^{3-}.

Wastewater samples (1 L) were collected in triplicates in 1 L glass bottles cleaned with dilute nitric acid (HNO_3) and detergent, then followed by deionized water (Igbinosa and Okoh, 2009; Standard Methods, 2001). Wastewater samples were collected from 4 sites at the pig farm that is Pig farm enclosures (Enc W), pig farm influent 2 m from the constructed wetland (Influent), constructed wetland for wastewater treatment (CW), and final effluent 2 m from the constructed wetland (effluent). Before sampling from each site, sampling glass bottles were flushed three times before being filled with the sample. Sampling of wastewater was done by dipping each sample bottle at approximately 20-30 cm below surface, projecting the mouth of the container against the flow direction (Igbinosa and Okoh, 2009).

Soil samples about 2 kg were collected using soil auger in sterile polythene bags at depth of 30 cm (Bhat et al., 2011). Soil samples were collected from 5 sites at pig farm that is, pig farm enclosures (Enc S), soil 20 m (Enc S-20 m) and 100 m (Enc S-100 m) away from the pig farm enclosures, soil 20 m (CW S-20 m) and 100 m (CW S-100 m) from pig farm constructed wetland. Wastewater and soil samples were placed on ice in a cooler box immediately after sampling and transported to the lab to be analyzed.

Critical parameters such as BOD, Salinity, pH, Temperature, EC, TDS, Turbidity, DO, concentrations of NO_3, NO_2, and PO_4^{3-}, were tested on the same day of sampling while the COD parameter was tested within its time limit. Samples for analyses of COD were collected separately in 1 L bottles and preserved with 0.2 mL of concentrated sulphuric acid on point of sampling and were analyzed within 28 days.

Physicochemical analysis

Parameters such as pH, temperature, electrical conductivity (EC), total dissolved solids (TDS), salinity, dissolved oxygen (DO), turbidity and biological oxygen demand (BOD) for water samples were determined onsite using a multi-parameter ion specific meter (Hanna instruments, version HI9828). Analysis of each parameter for wastewater was performed in triplicates. Blank samples (deionized water) were passed between every three measurements of samples to check for any eventual contamination or abnormal response of equipment. Temperature and pH were measured for both water and soil samples (Singh et al., 2012).

First analyses of BOD and DO were performed onsite, then again in the laboratory. BOD and DO were measured using BOD LDO Probe (Model LBOD 10101). The BOD and DO determination of the wastewater samples was carried out using standard methods described by APHA (1998). A 300 ml BOD bottle was used to add 297 ml of BOD nutrient pillow and 3 ml of sample. The results for BOD were recorded when the BOD LDO probe had stabilized. The dissolved oxygen (BOD) content was determined before and after incubation. Sample incubation for BOD was for five days in the dark at 20°C in BOD bottle. The following formula was used to calculate BOD_5:

$$BOD_5 = (D_1 - D_2)/P$$

Where:
BOD_5 = BOD value from the five day test
D_1 = DO of diluted sample immediately after preparation
D_2 = DO of diluted sample after five days incubation at 20 ± 1°C, in mg/L
P = Decimal volumetric fraction of sample used.

For measuring DO in samples, 300 ml of sample was poured into 300 ml BOD bottles (Singh et al., 2012). The results for DO were recorded when the probe had stabilized.

Analyses of TDS, EC NO_3, NO_2, PO_4^{3-}, and salinity were also adopted from Singh et al. (2012) (with amendments) and Standard Methods (2001) were followed in determining the aforementioned variables. Salinity, TDS, and EC were measured using HACH CDC401 probe. About 250 ml of the sample was poured into a 300 mL beaker, the HACH CDC401 probe was placed in the sample and the results were taken in triplicates. The probe was rinsed in deionized water after each test.

Concentrations of NO_3, NO_2, PO_4^{3-}, and COD were read using spectrophotometer HACH DR 500. Blank determinations were performed for COD, PO_4^{3-}, NO_3, and NO_2. PO_4^{3-} and was determined using the Molybdovanadate method (HACH Method 8114) (HACH, 2008). PO_4^{3-} was measured by adding 20 ml of the sample into a 25 ml graduated mixing cylinder. 1 content of Molybdate, 1 Reagent Powder Pillow was added to sample. The cylinder was stoppered and shaken to dissolve reagents. Then 10 ml of prepared samples was added to a 10 mL square sample cell and 0.5 mL of molybdenum, 2 Reagent was added and the cell was swirled and left to stand for 2 min for reaction to complete and results were taken immediately upon completion.

NO_3 was analyzed using the cadmium reduction method (HACH Method 8039) (HACH, 2008). Nitrate was then measured by adding 10 ml of sample into a 10 mL square sample cell and NitraVer 5 Nitrate Reagent powder pillow (HACH) was added to the sample. The reaction was left standing for 1 min and then shaken vigorously and left for another 5 min for the reaction to complete. The results were read immediately.

NO_2 was analyzed using the ferrous sulphate method (HACH Method 8153) (HACH, 2008). Nitrite was measured by adding 10 mL of sample into a 10 mL square sample cell and 1 content of NitriVer 2 Nitrite Reagent Powder pillow (HACH). The cell was stoppered and shaken to dissolve the contents. When completely dissolved, the solution was left to stand for 10 min for the reaction to complete and the results were taken immediately.

Standard Methods (2001) was followed for analyses of COD, where 100 ml of sample was homogenized in a blender for 30 s and 250 ml of sample was poured into a beaker and gently stirred on a magnetic stir plate. About 2 mL of the homogenized sample was pipette from the beaker into a vial containing potassium dichromate. The vial was inverted several times and then placed into a 150°C preheated DRB200 reactor for 2 h. Results were read when vials were completely cooled.

Turbidity was measured using DR5000 spectrophotometer. About 1.5 ml of sample was pipetted into 2 mL cuvettes and placed in the DR 5000 spectrophotometer 1-inch cell adapter (Singh et al., 2012). The results were read at 860 nm wavelength.

For soil samples, 100 g of air dried soil sample (air dried at 65°C) was mixed with 1 L of deionized water in a 1 L bottle previously cleaned with dilute Nitric acid (HNO_3) and detergent, then followed by deionized water (Bhat et al., 2011). This soil solution was mixed for 5 h using a magnetic stir plate. The solution was then removed and placed on the bench and left for 30 min for the soil to settle completely at the bottom (Bhat et al., 2011). Soil samples were analyzed for pH, temperature, salinity, EC, DO, TDS, COD, PO_4^{3-}, NO_3, and NO_2. Similar procedure for analyzing the physicochemical parameter for water was also adopted for analyzing physicochemical parameters of soil.

Statistical analysis

Calculation of means and standard deviations were performed using Microsoft Excel office 2010 version. Correlations (paired T-test) and test of significance (two-way ANOVA) were performed using SPSS 17.0 version for Windows program (SPSS, Inc.). All tests of significance and correlations were considered statistically significant at P values of < 0.05.

RESULTS

Mean and standard deviation (SD) values for each of the physicochemical parameter analyses were done in triplicates of wastewater. Samples are given in Table 1, and sample for soil is given in Table 3. Their P-value and F-value along with their significance are given in Table 2 for wastewater samples and in Table 4 for soil samples.

The results for physicochemical parameters of wastewater samples (Table 1) ranged from 6.5 to 9 (pH), 1.25 mS/cm to 5.58 mS/cm (EC), 8 to 28°C (temperature), 163 to 3550 mg/L (BOD), 0.77 to 6.48 g/L (TDS). Table 1 also shows results for salinity, COD, turbidity, and DO for wastewater samples ranged from 0.83 to 6.35 psu, 210 to 9400 mg/L, 0.21 to 3.65 NTU and 4.14 to 7.64 mg/L, respectively. Concentrations of PO_4^{3-}, NO_3, and NO_2 for wastewater samples (Table 1) ranged from 55 to 1680 mg/L, 37.5 to 2730 mg/L and 50 to 1427 mg/L, respectively. Results for pH, BOD, COD, DO, salinity, temperature, nitrate, nitrite, orthophosphate varied significantly ($p<0.05$) and results variation for EC, TDS, turbidity were insignificant (Table 2).

Physicochemical parameters analyzed for soil samples were, pH, temperature, salinity, EC, DO, COD, TDS, and the concentrations of PO_4^{3-}, NO_3, and NO_2. Table 3 shows that results for the physiochemical parameters of soil samples ranged from 6.28 to 8.43 (pH), 0.11 mS/cm to 1.37 mS/cm (EC), 12 to 25.5°C (temperature). Results for TDS, salinity, COD, and DO (Table 3) for soil samples ranged from 0.01 to 0.88 g/L, 0.01 to 0.13 psu, 40 to 304 mg/L, and 5.31 to 8.45 mg/L, respectively. Results for the concentration of PO_4^{3-}, NO_3, and NO_2 (Table 3) also

Table 1. Physicochemical parameters of wastewater for pig farm.

Sampling period	Sampling point	parameters											
		pH	Temp. (^oC)	Salinity (psu)	EC (mS/cm)	BOD (mg/L)	TDS (g/L)	Turbidity (NTU)	COD (mg/L)	DO (mg/L)	PO_4^{3-}(mg/L)	NO_2 (mg/L)	NO_3 (mg/L)
March	Enc W	7.25±0.5	22.4±0.95	1.18±0.07	3.10±0.13	413.54±15.94	6.19±0.17	1.23±0.13	3122.8±22	7.35±0.66	246.89±7.60	308.78±12	430.51±6.6
	Influent	9±0.00	25±0.00	2.08±0.06	3.50±003	694.5±31.25	6.48±0.10	1.79±0.31	4050±78.25	5.18±0.09	324.5±0.45	498.5±0.05	517.50±2.3
	CW	8.5±0.71	28±1.41	0.99±0.06	2.11±0.31	289.2±95.05	6.10±0.23	1.01±0.20	1025.3±704	6.14±0.93	331.04±40	209.13±93	438.1±232
	Effluent	8±0.00	26±0.00	1.08±0.01	1.58±0.04	163±5.23	4.07±0.01	0.41±0.11	521±13.50	6.51±0.25	55.9±0.35	75±0.13	550±0.31
April	Enc W	7.5±0.8	15.8±1.05	2.90±0.49	2.64±0.19	767.25±5.91	2.28±0.48	2.57±0.20	5087.5±246	7.64±0.09	825.13±53	66.88±7.12	146±27.53
	Influent	8±0.00	20±0.00	3.74±0.10	4.17±0.05	770±49.35	3.66±0.11	3.10±0.71	9400±99.1	5.64±0.47	980.5±0.4	202.5±1.1	625±3.41
	CW	8.±0.00	19.5±0.71	1.21±0.27	2.02±0.39	645.5±160.5	1.16±0.02	1.41±0.45	843±357.80	7.25±0.70	833.96±42.2	109.9±55.3	169.5±85.6
	Effluent	8±0.00	18±0.00	0.83±0.04	1.25±0.06	623±11.31	1.03±0.04	0.58±0.52	512±96.07	6.19±0.05	992.32±0.08	52±0.71	70.83±0.72
May	Enc W	8.25±0.5	8.75±1.5	4.58±0.58	3.49±0.35	1247.95±292	3.73±1.14	2.43±0.17	6545±456.9	5.94±0.52	893.88±69.9	169.75±13	1306±134
	Influent	8±0.00	15±0.00	6.35±0.16	5.58±0.13	2562.5±25	6.32±0.26	2.85±0.27	7065±87	5.23±0.11	1427±0.2	233.5±0.4	1407±48.4
	CW	8±0.00	18±1.41	3.42±1.76	2.75±0.63	1758.95±320	3.91±1.05	1.93±0.74	3288±1403	5.39±0.20	1170.5±112.4	170.5±112	464.3±89.4
	Effluent	8±0.00	18±0.00	1.13±0.05	2.24±0.08	1263.2±83	2.12±0.05	0.41±0.36	760±99.6	5,43±0.02	853.19±0.1	37.5±1.37	637.5±2.0
June	Enc W	7.5±0.58	9.0±4.08	1.42±0.16	2.72±0.28	2168.5±244.3	1.61±0.29	2.26±0.18	5832.5±541	5.87±0.79	829±44.80	1425±132	471.3±22.9
	Influent	8±0.00	14±0.00	2.04±0.07	3.04±0.07	3350±209	2.03±0.05	2.60±0.39	7500±74.	4.14±0.05	925±0.28	2458±1.0	693±0.99
	CW	8±0.00	14±1.41	1.04±0.18	2.03±0.34	1745±625.80	1.02±0.17	1.33±0.34	6210±622.3	4.54±0.11	373.16±229	1131±149	325±176.8
	Effluent	8±0.00	15±0.00	0.93±0.07	2.24±0.04	1010±99.0	0.77±0.06	0.74±0.29	4560±94	4.87±0.08	99.31±0.3	653±0.17	145±0.93
July	Enc W	7.75±0.5	8.13±1.65	1.66±0.37	3.23±0.61	2066.38±607	1.67±0.33	2.26±0.25	6464±373.9	5.22±0.31	175.75±14	1625±64.5	581.3±41.5
	Influent	8±0.00	12,5±0.00	3.64±0.09	4.29±0.02	3152±68.3	2.64±0.09	3.65±0.46	7295±89.9	4.71±0.06	235±0.31	2730±1.21	1680±1.80
	CW	7.5±0.71	13.25±2.48	1.06±0.18	2.07±0.34	1020±38.89	1.05±0.18	1.01±0.57	1792.5±894	5.05±0.55	170±21.21	1125±460	1060±283
	Effluent	7±0.00	13±0.00	0.84±0.06	1.67±0.04	402.5±34.5	0.83±0.04	0.21±0.19	940±79.83	5.80±0.08	125±0.32	350±0.07	530±0.61
August	Enc W	7.5±0.58	8.0 ±0.82	1.46±0.43	2.20±0.82	1583.63±317	1.53±0.33	1.49±0.17	1718.5±132	5.73±0.45	173.75±14.9	1173.5±33.	178.8±11.8
	Influent	6±0.00	11±0.00	2.64±0.13	4.01±0.06	3550±480.8	3.35±0.06	2.24±0.51	3580±90.91	4.46±0.21	240±0.27	1850±0.86	490±1.31
	CW	8±0.00	11.5±1.41	1.15±0.15	2.54±0.71	1405±134.35	1.13±0.14	0.92±0.25	1100±608.1	4.97±0.35	107.5±38.89	425±35.36	105±35.36
	Effluent	8±0.00	16±0.00	0.93±0.07	1.84±0.06	1170±10.61	0.93±0.01	0.52±0.22	210±127.28	4.75±0,08	50±0.10	250±0.01	55±0.20
Standards Error		6-9	<25	33-35	70	<40	450	<5	≤1000	≥5	≤30	≤0.5	≤20

Temp: Temperature; EC: Electrical Conductivity; BOD: Biological Oxygen Demand; TDS: Total Dissolved Solids; COD: Chemical Oxygen Demand; DO: Dissolved Solids; PO_4^{3-}: Orthophosphate; NO_2: Nitrite; NO_3: Nitrate. The table shows results for physicochemical parameters of pig farm wastewater samples where the standards were adopted from DWARF (1996).

Table 2. The P-value and F-value for physicochemical parameters of wastewater for pig farm.

P and F values	Parameters											
	pH	**Temp.**	**Salinity**	**EC**	**BOD**	**TDS**	**Turbidity**	**COD**	**DO**	**Ortho-P**	**NO_2**	**NO_3**
F values[a]	2.91	71.59	32.21	1.01	28.62	0.86	0.74	3.79	32.58	13.95	39.53	7.22
P values[b]	0.02*	0.00*	0.00*	0.42	0.00*	0.51	0.60	0.00*	0.00*	0.00*	0.00*	0.00*
F values[c]	6.07	4.75	7.85	1.01	7.23	1.21	1.25	2.45	6.37	9.90	4.99	2.08
P values[d]	0.00*	0.00*	0.00*	0.43	0.00*	0.30	0.28	0.02*	0.00*	0.00*	0.00*	0.05*
F values[e]	3.55	10.22	11.52	0.93	10.28	0.98	0.98	2.87	4.95	9.14	12.58	4.00
P values[f]	0.00*	0.00*	0.00*	0.52	0.00*	0.47	0.47	0.00*	0.00*	0.00*	0.00*	0.00*

Temp: Temperature; EC: Electrical Conductivity; BOD: Biological Oxygen Demand; TDS: Total Dissolved Solids; COD: Chemical Oxygen Demand; DO: Dissolved Solids; Ortho-P: Orthophosphate; NO_2: Nitrite; NO_3: Nitrate. *P<0.05 significant variation. Values are expressed in milligrams per litre except in pH, temperature (in degrees Celsius), salinity (in practical salinity unit), and EC (in micro-Siemens per centimetre), TDS (grams per litre). [a]F values for parameters and month. [b]P values for parameters and month. [c]F values for parameters and sampling point. [d]P values for parameter and sampling point. [e]F values for combined effect of month and sampling point on parameters. [f]P values for combined effect of month and sampling point on parameter.

ranged from 32.5 to 475 mg/L, 9 to 142 mg/L and 7.35 to 255 mg/L. All the results for the physicochemical parameters of soil samples varied significantly (p<0.05), monthly (Table 4).

The correlation of the physicochemical parameter of wastewater samples from pig farm is shown in Table 5 and those of soil sample in Table 6. The highest significant correlations (p<0.05) for wastewater physicochemical parameters (Table 5) observed in this study were between Salinity and orthophosphate (positive correlation), and between BOD and temperature (negative correlation). The lowest insignificant correlation for pig farm wastewater physicochemical parameters (Table 5) observed in this study were between salinity and DO (positive correlation), and between TDS and nitrate (negative correlation). The highest significant correlations (p<0.05) for soil physicochemical parameters (Table 6) observed in this study were between orthophosphate and COD (positive correlation), and between orthophosphate and nitrate (positive correlation). The lowest insignificant correlation for soil physicochemical parameters (Table 6) observed in this study were between pH and nitrite (negative correlation), and between temperature and orthophosphate (positive correlation)

DISCUSSION

The pH values for wastewater samples (Table 1) and soil samples (Table 3) fell within the recommended limit of 6-9 (DWAF, 1996c; Government Gazette, 2012). The near neutral and alkaline nature observed for soil samples may be attributed to surface runoff or overflow of the observed alkaline wastewater. The pH values for wastewater and soil samples varied significantly (Table 2 and Table 4). High pH in soil and water systems altered the solubility of other chemical pollutants and caused the volatilization as well as microbial decomposition of organic acid. Thus, the subsequent release of ammonia through mineralization of organic nitrogen source (Singh and Agrawal, 2012) can be elevated due to high pH in soil and water systems. Low pH in soil can increase the availability of metals since hydrogen ions have the affinity for competing with metals ions and releasing them in soil solution for uptake by plants (Singh and Agrawal, 2012). Results were similar to those observed by Aguilar et al. (2011), where pH values of 6 to 8 was recorded for wastewater samples and 6.2 to 8.6 for soil samples from pig farm in this study.

The South African guideline for EC in wastewater and effluent that could be discharged into the receiving water system is 70 mS/cm and limit for soil EC is set at 2 mS/cm for the protection of plants and groundwater (Government Gazette, 2012). The variation of EC values for wastewater samples were insignificant (table 2) while EC values for soil samples varied significantly (p<0.05) at the monthly intervals (Table 4). This may be due to the low salinization and alkaline nature of soil and wastewater sample observed in all sampling sites.

High temperature affects the toxicity of some chemicals in the environment as well as the sensitivity of living organisms to toxic substances (Akan et al., 2008). Low temperature in soil slows the chemical and biological rate processes, while high temperature in soil affects seed germination, regenerates absorption and transport of water and nutrients (Roth et al., 2014). According to the South African standard for wastewater and effluent temperature, the limit was set at ≤25°C (Department of Water Affairs and Forestry (DWAF), Water Research Commission (WRC), 1995). Temperature for both soil and wastewater samples (Table 1 and Table 3) fell within

Table 3. Physicochemical parameters of pig farm soil samples.

Period	Sampling point	Parameters									
		pH	Temp. (°C)	Sal.(psu)	EC (mS/cm)	TDS (g/L)	COD (mg/L)	DO (mg/L)	PO_4^{3-} (mg/L)	NO_2 (mg/L)	NO_3(mg/L)
March	Enc-S	6.75±0.37	25.00±1.0	0.06±0.03	0.54±0.01	0.66±0.10	242.67±4.73	7.68±0.21	62.27±6.72	56.23±3.15	152.7±46.89
	Enc S-20m	7.2±0.03	13±0.0	0.03±0.01	0.48±0.0	0.49±0.02	258±2.15	7.91±0.1	53.61±0.07	46.37±0.67	79.5±0.57
	Enc S-100m	6.67±0.01	23±0.0	0.01±0.00	0.39±0.0	0.37±0.00	159±2.89	7.69±0.3	27.19±0.9	39.26±0.1	51±2.01
	CW S-20m	6.54±00	25.5±0.0	0.05±0.01	0.45±0.0	0.29±0.01	217±3.05	8.03±0.2	33.38±0.54	12.67±0.10	273.5±1.15
	CW S-100m	6.91±0.01	22±0.0	0.02±0.01	0.33±0.0	0.13±0.00	152±1.89	8.45±0.1	19.01±0.31	11.05±1.22	117.5±1.09
April	Enc-S	6.95±0.23	20.17±0.76	0.05±0.01	0.28±0.04	0.13±0.02	141.13±2.52	7.54±0.10	52.40±3.15	49.39±6.57	162.17±35.3
	Enc S-20m	7.4±0.03	13±0.0	0.02±0.00	0.22±0.0	0.16±0.01	108±3.75	7.61±0.1	40.50±0.1	29.77±0.2	87±2.06
	Enc S-100m	7±0.02	20±0.0	0.07±0.01	0.12±0.0	0.10±0.01	91±3.12	7.81±0.2	21.37±0.3	10.84±0.4	64.5±1.45
	CW S-20m	6.28±0.01	18.5±0	0.03±0.01	0.14±0.0	0.14±0.01	117±2.56	7.55±0.1	28.50±0.3	41.91±0.1	280.5±0.7
	CW S-100m	7.03±0.01	16±0.0	0.01±0.01	0.11±0.0	0.05±0.01	86±4.35	8.01±0.0	14.02±0.29	12.35±1.31	110.5±0.37
May	Enc-S	7.14±0.48	17.67±1.53	0.07±0.03	0.75±0.48	0.24±0.02	169.33±8.50	6.69±0.60	37.83±2.70	48.33±5.75	135±28.83
	Enc S-20m	7.07±0.01	13±00	0.06±0.00	0.43±0.0	0.16±0.00	118±4.32	7.34±0.2	26.8±0.08	31.00±0.1	77.5±0.72
	Enc S-100m	6.98±0.01	17±00	0.09±0.02	0.32±0.0	0.09±0.02	82±3.65	7.56±0.1	12.05±0.29	15.50±0.03	32.5±1.74
	CW S-20m	7.98±0.02	15±00	0.01±0.00	0.34±0.0	0.07±0.00	152±2.50	7.36±0.3	21.7±0.35	52.50±0.1	217.5±1.3
	CW S-100m	7.11±0.03	14.5±00	0.02±0.01	0.18±00	0.05±0.0	92±1.55	7.71±0.3	10.17±0.49	10.25±0.07	122.5±1.75
June	Enc-S	6.91±0.52	14.00±1.00	0.04±0.02	0.65±0.41	0.22±0.04	122.67±4.73	5.64±0.55	24.37±0.96	29±7.72	272.83±19.8
	Enc S-20m	6.97±0.01	13±00	0.01±0.00	0.33±0.0	0.14±0.01	98±2.56	7.14±0.3	13.1±0.36	13.9±0.10	135±1.10
	Enc S-100m	6.87±0.02	12±00	0.01±0.00	0.28±0.0	0.08±0.01	41±1.25	7.48±0.4	9.2±0.30	9.6±0.37	50±0.42
	CW S-20m	7.73±0.02	16±00	0.03±0.01	0.30±0.0	0.15±0.02	107±2.55	7.44±0.1	15±0.14	21±0.29	225±1.81
	CW S-100m	6.93±0.01	14±00	0.00±0.00	0.19±0.0	0.06±0.00	72±3.01	7.75±0.1	7.35±0.27	9.23±0.34	112.5±2.00
July	Enc-S	8.01±0.60	13.33±2.08	0.05±0.01	0.73±0.08	0.29±0.15	260.17±9.57	5.31±0.40	41.82±2.46	79.67±95.62	241.67±27.5
	Enc S-20m	7.41±0.02	13±00	0.01±0.00	0.42±0.0	0.11±0.01	140±2.43	6.02±0.1	21.7±0.58	17±0.13	130±1.15
	Enc S-100m	7.67±0.02	12±00	0.01±0.00	0.31±0.0	0.03±0.01	124±1.46	6.25±0.0	10±0.26	9.01±0.39	52.2±0.96
	CW S-20m	8.15±0.01	15±00	0.04±0.02	0.48±0.0	0.54±0.01	260±2.03	7.10±0.3	64±0.17	71±0.32	301.5±2.78
	CW S-100m	7.99±0.02	13±00	0.01±0.00	0.33±00	0.01±0.01	108±2.99	7.89±0.2	24±0.20	11±0.19	90±1.42
August	Enc-S	7.82±0.34	13.17±0.29	0.02±0.01	0.28±0.02	0.15±0.03	266±4.58	6.12±0.13	137.67±6.43	17.33±2.52	146.67±12.6
	Enc S-20m	7.76±0.01	13±00	0.01±0.00	0.21±0.0	0.09±0.01	140±3.45	6.69±0.2	45±0.09	19±0.51	80±1.66
	Enc S-100m	6.95±0.01	12.5±00	0.01±00	0.23±0.4	0.011±00	40±2.18	7.01±0.1	29±0.21	8±0.21	75±1.38
	CW S-20m	8.43±0.01	16±00	0.13±0.02	1.37±0.0	0.88±0.01	304±2.79	6.52±0.4	255±0.68	142±0.15	475±1.19

Table 3. Cont.

	CW S-100m	7.75±0.02	14±00	0.02±0.01	0.54±0.0	0.03±0.01	152±4.04	6.85±0.2	48±0.31	12±0.31	100±1.74
Standards		6.5-8	<40	≤0.2	≤2	≤500	≤ 200	≥ 5	≤ 5	≤ 13	≤ 120

Temp: Temperature; EC: Electrical Conductivity; TDS: Total Dissolved Solids; COD: Chemical Oxygen Demand; DO: Dissolved Solids; Ortho-P: Orthophosphate; NO_2: Nitrite; NO_3: Nitrate. All parameters are expressed in mg/L except for Temperature (^{O}C), Electrical Conductivity (mS/cm), Salinity (psu). Standards were adopted from FME and Government Gazette.

Table 4. The P-value and F-value for physicochemical parameters of pig farm soil samples.

P and F values	Parameters									
	pH	Temp.	Salinity	EC	TDS	COD	DO	Ortho-P	NO_2	NO_3
F values [a]	26.03	20.68	15.96	7.62	17.69	20.58	17.37	20.57	48.21	2.80
P values [b]	0.00*	0.00*	0.00*	0.00*	0.00*	0.00*	0.05*	0.00*	0.00*	0.02*
F values [c]	4.88	0.86	8.80	10.32	5.88	9.74	7.64	4.74	13.63	38.77
P values [d]	0.00*	0.53	0.00*	0.05*	0.00*	0.00*	0.00*	0.00*	0.00*	0.00*
F values [e]	2.09	6.88	8.50	6.56	9.34	8.62	3.32	9.13	5.38	8.36
P values [f]	0.03*	0.00*	0.00*	0.00*	0.00*	0.00*	0.00*	0.00*	0.00*	0.00*

Temp: Temperature; EC: Electrical Conductivity; BOD: Biological Oxygen Demand; TDS: Total Dissolved Solids; COD: Chemical Oxygen Demand; DO: Dissolved Solids; Ortho-P: Orthophosphate; NO_2: Nitrite; NO_3: Nitrate. *P<0.05 significant variation. Values are expressed in milligrams per litre except in pH, temperature (in degrees Celsius), salinity (in practical salinity unit), and EC (in micro-Siemens per centimetre), TDS (grams per litre). [a] F values for parameters and month. [b] P values for parameters and month. [c] F values for parameters and sampling point. [d] P values for parameter and sampling point. [e] F values for combined effect of month and sampling point on parameters. [f] P values for combined effect of month and sampling point on parameter.

the recommended limits of ≤25°C for wastewater to be discharged to water systems (DWAF, WRC, 1995) and for soil at ≤40°C for the protection of plants and groundwater (FME, 2011). This may be because samples were collected in the morning and atmospheric temperature (differed monthly due to seasons) never reached as high as 25°C during sampling periods. This explains the significant interaction effect of month and sampling point on temperature (Table 2 and Table 4) and indicates that temperature was not only a function of season but also dependent on sampling point. Thus, the observation of temperature values in this study implies that seepage temperature may not offset the homeostatic balance of the receiving environment.

The levels of BOD for wastewater samples (Table 1) exceeded the recommended limit of 40 mg/L set by FAO (1992) for agricultural purposes and varied significantly (Table 2). This may be attributed to the high use of chemicals at the pig farm that are organic or inorganic in nature and this can promote an increase in microbial growth and microbial degradation of organic or inorganic matter. High BOD values can cause greater oxygen demand in the receiving environment and thus leading to depletion of available oxygen to critical levels (Roth et al., 2014). BOD for wastewater samples observed in this study were lower than those reported by Vanotti et al., (2002) and were also higher than those reported by González et al., (2009).

TDS results for wastewater samples (Table 1)

Table 5. Correlation matrix of physicochemical parameters of pig farm wastewater samples.

	pH	***BOD***	***Temperature***	***EC***	***Salinity***	***TDS***	***DO***	***Turbidity***	***Ortho-P***	***COD***	***Nitrate***	***Nitrite***
pH	1											
BOD	-0.0503	1										
Temperature	0.2075*	-0.5933*	1									
Conductivity	0.0415	0.1173*	0.0064	1								
Salinity	0.0379	0.2685*	-0.1843*	0.2731*	1							
TDS	-0.1011*	0.0184	-0.0418	0.021	0.0359	1						
DO	-0.2956*	-0.582*	0.3268*	-0.0486	0.0017	-0.0219	1					
Turbidity	0.0266	0.0786	-0.1212*	0.0178	0.0687	0.0213	-0.088	1				
Orthophosphate	-0.0914	0.1827*	-0.1343*	0.2563*	0.6206*	-0.0091	0.1874*	0.0561	1			
COD	0.0512	0.3591*	-0.1856*	0.0269	0.3199*	0.0434	-0.1467*	0.0335	0.1472*	1		
Nitrate	-0.1561*	0.3184*	-0.218*	0.0423	0.176*	-0.0003	-0.2076*	0.0468	-0.0085	-0.0247	1	
Nitrite	-0.0543	0.171*	0.0905	-0.0489	-0.0961	0.055	-0.1921*	-0.012	-0.0475	0.0922	0.0647	1

COD: Chemical Oxygen Demand; BOD: Biological Oxygen Demand; TDS: Total Dissolved Oxygen; DO: Dissolved Oxygen; EC: Electrical Conductivity; Ortho-P: orthophosphate. -: negative correlation
*= P<0.05 significant variation

Table 6. Correlation matrix of physicochemical parameters of pig farm soil samples.

	pH	***Temperature***	***EC***	***Salinity***	***TDS***	***DO***	***0rtho-P***	***COD***	***Nitrate***	***Nitrite***
pH	1									
Temperature	-0.4*	1								
EC	0.1404*	0.1068*	1							
Salinity	-0.007	0.2909*	0.6358*	1						
TDS	-0.113*	0.5796*	0.3488*	0.5266*	1					
DO	-0.453*	0.4721*	-0.5026*	-0.2149*	-0.0658	1				
0rtho-P	0.2924*	-0.0650	-0.0889	0.0661	0.3472*	-0.1805*	1			
COD	0.3655*	0.1083*	0.3338*	0.2920*	0.5927*	-0.3851*	0.6897*	1		
Nitrate	0.1551*	-0.0994	0.1666*	0.5571*	0.1975*	-0.3276*	0.4373*	0.3679*	1	
Nitrite	-0.004	0.3241*	0.1625*	0.4737*	0.5120*	0.0577	0.6111*	0.4238*	0.5935*	1

COD: Chemical Oxygen Demand; BOD: Biological Oxygen Demand; TDS: Total Dissolved Oxygen; DO: Dissolved Oxygen; EC: Electrical Conductivity; Ortho-P: orthophosphate. -: negative correlation
*= P<0.05 significant variation

were higher than the recommended standards set by DWAF, (1998) which set the limit of ≤ 450 mg/L of no risk to aquatic life for seepage released into the receiving environment. TDS values for wastewater samples did not vary significantly (Table 2). This may be attributed to the possible presence of potassium chloride and sodium which are known to elevate TDS concentrations. High TDS can be toxic to freshwater animals by causing osmotic stress and affecting the osmoregulatory capability of organisms (Akan et al., 2008). The results obtained for soil TDS

(Table 3) were within the limits of ≤500 mg/L for the protection of ground water as set by FME and varied significantly (Table 4). Soil TDS was observed to be higher in March in enclosure soil (Enc S), and in July and August Soil 20 m from constructed wetland (CW S-20 m) as shown in Table 1, where TDS was recorded to be 0.66±0.10 g/L (March), 0.54±0.01 g/L (July), and 0.88±0.01 g/L (August). The significant difference ($P<0.05$) in soil TDS values observed in March (Enc S), July and August (CW S-20 m) may be responsible for the observed monthly variation.

The salinity results for wastewater fell within the acceptable limit of 33 psu to 35 psu of no risk for all biological activities in the marine ecosystem (Whitefield and Bate, 2007). Salinity for soil must not exceed 0.2 psu for the protection of plants and ground water (Government Gazette, 2012). Salinity levels in soil samples (Table 3) were within the recommended limits set by Government Gazette (2012). This may be due to the low EC observed in soil samples (Table 3). High salinity levels in water resources increases requirements for pre-treatment of water for selected seepage. This can cause serious ecological disturbance that may result in adverse effects on the aquatic biota (Oluyemi et al., 2006). High salinity in soil hinders plant growth by affecting the soil-water balance in soil. Salinity values for both soil and wastewater samples varied significantly (Tables 2 and 4). This significant variation may be due to salinity level at effluent, Enc S- 100 m and CW S-100 m which consistently remained lower as compared to other sampling points (Table 1 and Table 3). This may be caused by low EC observed at these sampling points as EC is a measure of salinity in the environment.

The COD results for wastewater samples (Table 1) fell short of the acceptable limit of < 30 mg/L as recommended by the South African government (Government Gazette, 1984). COD values for wastewater samples varied significantly ($P<0.05$) (Table 2). An elevated level of COD in water system leads to drastic oxygen depletion which adversely affects the aquatic life (Fatoki et al., 2003). High COD values observed in wastewater samples in June and July compared to other sampling months could be attributed to low degradation rate of organic matter due to low microbial activity due to cold temperatures. This is shown by the observed positive correlation between COD and temperature (Table 5). The results obtained for soil COD met the required limit of ≤ 200 mg/L for protection of ground water as set by Government Gazette (1984) except for CW S-20 m and Enc S in March, July, and August (Table 3). This may be attributed to surface run off or leaching of pig farm wastewater with high COD levels (Table 1). The COD values for soil samples varied significantly (Table 4). The significant variation may be attributed to COD values at Enc S and CW S-20 m and Enc S- 20 m were consistently high. This may be due to an increase in the addition of both organic and inorganic substrate leaching from wastewater emanating from Enc W and CW. High COD in soil causes soil fixation, resulting in lower availability of nutrients for plants (Chukwu, 2005). Similar results were also reported by Aguilar et al. (2011) where COD from pig farm seepage was recorded to be as high as 9960.83 mg/L due to low microbial activity caused by cold temperature.

Turbidity values for wastewater (Table 1) fell within acceptable limits of ≤5 NTU by DWAF (DWAF, 1996c). The variation in turbidity values in wastewater samples was insignificant (Table 2). Excessive turbidity in seepage can cause problem with water purification processes such as flocculation and filtration, which may increase treatment cost (DWAF, 1998). High turbid waters are often associated with the possibility of microbiological contamination and high load of organic and inorganic nutrients, as high turbidity makes it difficult to disinfect water properly (DWAF, 1998).

The results of DO for wastewater samples (Table 1) and soil samples (Table 3) fell within the acceptable limit (≥5 mg/L) of no risk for the support of aquatic life and protection of ground water (DWAF, 1998) except for Influent, CW, and Effluent in June and August (Table 1). This may be attributed to the high nutrient load in seepage which can be a contributing source of nutrient to the receiving environment (Akan et al., 2008). DO values vary significantly ($p< 0.05$) for both soil and wastewater samples (Tables 2 and 4). Dissolved oxygen is essential in maintaining the oxygen balance in the environment especially aquatic ecosystem (Fatoki et al., 2003). Low DO can negatively impact an aquatic life by increasing their feeding migration and thus, leading to loss of life (Environment Canada, 2001).

Nitrate concentration for seepage must not be exceeded by the acceptable limit of 20 mg/L set by (DWAF, 1996c) and FME has set the limit for soil nitrate at ≤ 13 mg/L for the protection of ground water. As observed in this study, both soil and wastewater samples (Tables 1 and 3) did not meet the required limit set by DWAF and FME. This may be attributed to high nutrient load due to surface run-off and leaching of wastewater with high nitrate concentration from pig farm on the surrounding natural environment. Nitrate values for both wastewater and soil samples varied significantly ($p<0.05$) (Tables 2 and 4). High nitrate levels may result in excessive nutrient enrichment in water systems (eutrophication) leading to loss of diversity in the aquatic biota, environmental degradation through algal blooms, oxygen depletion and reduced sunlight penetration (Canadian Environmental Quality Guidelines (CCME), 2006). Nitrite like nitrate is also a source of nutrition that could have negative impacts on aquatic ecosystems at elevated concentrations. The nitrite levels for wastewater samples (Table 1) fell short of the South African standard (<0.5 mg/L NO_2) for preservation of aquatic ecosystem

(DWAF, 1996c). Nitrite levels for soil samples (Table3) also did not fall within the limit of ≤ 13 mg/L as set by FME for protection of ground water. This nitrite levels for wastewater and soil observed in this study can put the aquatic ecosystems and ground water at risk of eutrophication. Nitrite values for both soil and wastewater samples varied significantly ($p<0.05$) (Table 2 and Table 4). Soil nitrite concentration at Enc S, Enc S-20m, CW S-20m exceeded the required limit and this may be caused by surface run-off or leaching from wastewater with high nitrate concentration of surrounding environment. Results for wastewater samples in this study were higher than those observed by knight et al., (2000). Results obtained in this study for soil nitrite were higher than those obtained by Roth et al., (2014).

Orthophosphate levels for wastewater samples (Table 1) observed in this study exceeded the standard of 30 mg/L (DWAF, 1996c) and orthophosphate levels for soil samples (Table 3) also exceeded the FME standard of ≤ 5 mg/L. This observed PO_4^{3-} level will promote growth of algae and suggest that seepage from pig farm is polluted and poses a serious threat to the aquatic biota and the ecosystem of the receiving environment in general. PO_4^{3-} values for both soil and wastewater samples in this study varied significantly ($p<0.05$) (Tables 2 and 4). High PO_4^{3-} concentrations in wastewater samples (Table 1) in April, May, and June as compared to other sampling months may be the cause of PO_4^{3-} concentration (44 to 88 mg /L) variation. High orthophosphate concentration increase algae and plant growth in aquatic systems. Orthophosphate concentration for soil samples ranged from 3.69 to 9.5 mg/g near pig farm's enclosure.

The correlation of the physicochemical parameter of water samples from pig farm is shown in Table 5 and those of soil sample in Table 6. In Table 5, the insignificant correlation between pH and salinity (r = 0.038) was caused by the almost neutral pH concentration observed in samples. The insignificant negative correlation between DO and salinity (r = -0.121) in soil and the positive insignificant correlation in water samples (r = 0.002) as shown in Table 5 and Table 6 indicates that DO concentration decrease with an increase in salinity levels as observed in this study which may be due to a high nutrient load in pig farm seepage.

Several studies have reported that EC and TDS are good indicators of salinity (Akan et al., 2008; Oluyemi et al., 2006). Correlation of EC and TDS (r = 0.0211) in water samples (Table 5) was insignificant in this study, while the correlation of EC and TDS (r = 0.349) in soil samples (Table 6) were significantly higher as compared to salinity. It is expected that seepage should be high in EC and TDS levels to promote microbial growth. The significant correlation of salinity and nitrates (r=0.176) in water samples (Table 5) indicates that the less saline seepage of nitrates can be attributed to the consistent high concentration of nitrate in pig enclosures and influent, as compared to other sampling points.

The insignificant correlation between salinity and turbidity (r = 0.068), TDS and turbidity (r = 0.021) in water samples (Table 5) shows that effluents released in the pig farm may be a source of turbidity in the receiving environment. However as observed in this study, there was no correlation between salinity and TDS in both soil (r = 0.11872) and water samples (r = 0.035976) (Tables 5 and 6). This may be due to the high concentrations in organic and inorganic nutrients in the pig farm seepage. The insignificant correlation of COD with EC (r = 0.0270), TDS (r = 0.0434), pH (r = 0.0513), DO (r = 0.147), in waters samples (Table 5) and the insignificant correlation of COD with salinity (r = 0.086) in soil samples (Table 6) was due to the high COD levels caused by high rate of organic breakdown of organic and inorganic nutrients in seepage. This study is still ongoing, and efforts to further determine the impact on the microbial diversity of natural environment in the vicinity of pig farm due to impacts of pig farm seepage on the physicochemical parameters of the natural environment are still in progress.

Conclusion

High BOD, COD, and TDS levels as observed in wastewater and soil samples in this study could constitute potential pollution problems to the natural environment since they contain organic compounds that will require large quantities of oxygen for degradation. High levels of PO_4^{3-}, NO_3 and NO_2 leads to the eutrophication of the natural environment, which was evident of organic matter infiltration occurring at pig farm. It is therefore concluded that the pig farm seepage caused negative impact on the receiving site and its environment due to depletion in available oxygen, increase solubility of heavy metals and increase toxicity of other chemicals. Pig farm seepage also caused an increase in the sensitivity of living organisms to other toxic substances in soil and water systems. Furthermore, pig farm seepage may cause osmotic stress to the natural environment and affects osmoregulatory capability of organisms, and may cause eutrophication of water systems and soil in the natural environment in the vicinity of pig farm. Thus, efforts are still ongoing to further determine the impacts of pig farm seepage on the microbial diversity in the natural environment in the vicinity of pig farm in ARC-API. If microbial diversity is determined, mitigation for preventing microbial contamination of natural environment in vicinity of pig farm can be effected to reduce degradation of the environment.

Conflict of interest

The authors have not declared any conflict of interest.

REFERENCES

Aguilar Y, Bautista F, Diaz-Pereira E (2011). Soils as natural reactors of swine wastewater treatment. J. Trop. Subtrop. Agroecosys. 13(6):199-210.

Akan JC, Abdulrahman FI, Dimari GA, Ogugbuaja VO (2008). Physicochemical determination of pollutants in wastewater and vegetable samples along the Jakara wastewater channel in Kano Metropolis, Kano State, Nigeria. Eur. J. Sci. Res. 23(1):122-133.

APHA (1998). Standards methods for examination of wastewater.18th edition. American public health association Washington, DC. pp. 45-60.

Bhat SH, Darzi AB, Dar MS, Ganaie MM, Bakhshi SH (2011). Correlation of soil physico-chemical factors with vam fungi distribution under different agroecological conditions. Int. J. Pharm. Bio. Sci. 2(2):98-107.

Canadian Council of Ministers of the Environment [CCME] (2006). Municipal wastewater effluent in Canada: a report of the municipal wastewater effluent development committee.

Chukwu O (2005). Development of predictive models for evaluating environmental impact of the food processing industry: Case studies of Nasco Foods Nigeria Limited and Cadbury Nigeria PLC. Unpublished PhD. Thesis, DEA, FUT, Minna Niger State, Nigeria.

DWAF (1998). Quality of Domestic Water Supplies. Assessment Guide. 1 (2nd. Ed.) Department of Water Affairs and Forestry, Department of Health and Water Research Commission.

DWAF (1996c). South African Water Quality Guidelines, Aquatic ecosystems (1st. Ed.). Department of Water Affairs and Forestry, Pretoria. Vol. 7.

DWAF, WRC (1995). Procedures to assess effluent discharge impacts. WRC Report No. TT 64/94. South African water quality management series; Department of water Affairs and Forestry and Water Research Commission, Pretoria.

Environment Canada (2001). The state of municipal wastewater effluent in Canada. Minister of Public Works and Government Services Canada, Ottawa, Ontario K1A 0H3.

FAO (1992). Wastewater treatment and use in agriculture Food and Agricultural Organization irrigation and drainage paper 47. FAO corporate document repository, Available at: http://www.fao.org/docrep/T0551E/t0551e00.htm.

Fatoki SO, Gogwana P, Ogunfowokan AO (2003). Pollution assessment in the Keiskamma River and in the impoundment downstream. Water SA 29(3):183-187.

Federal Ministry for the Environment, FME (2011). Soil, groundwater and sediments standards for use under part xv.1 of the environment protection act. Queens Printer Ltd (Pty).

González FT, Vallejos GG, Silveira1 JH, Franco CQ, García J, Puigagu J (2009). Treatment of swine wastewater with subsurface-flow constructed wetlands in Yucatán, Mexico: Influence of plant species and contact time. Water SA 35(3):335-342.

Government Gazette (1984). Requirements for the purification of wastewater or effluent. Gazette No. 9225, Regulation. P 991.

Government Gazette (2012). Draft national norms and standards for the remediation of contaminated land and soil quality. Volume 561, Gazette No.35160.

HACH, DR500 user manual (2008). 2nd edition, HACH Company. Germany.

Igbinosa EO, Okoh, AI (2009). Impact of discharge wastewater effluents on the physicochemical qualities of a receiving watershed in a typical rural community. Int. J. Environ. Sci. Tech. 6(2): 175-182.

Knight RL, Pyne VWE, Borer RE, Clarke RA, Pries JH (2000). Constructed wetlands for livestock wastewater management. J. Ecol. Eng. 15(1-2):41-55.

Muhibbu-din OI, Aduwo AO, Adedeji AA (2011). Study of Physiochemical Parameter of Effluent Impacted Stream in Obafemi Awolowo University, Ile-Ife, Nigeria. A Dissertation Submitted to the Department of Zoology, Obafemi Awolowo University, Ile-Ife.

Obasi LN, Nwadinigwe CA, Asegbeke JN (2008). Study of Trace Heavy Metal in Fluted Pumpkin leaves grown on Soil Treated with Sewage Sludge and Effluents. Proceedings 31st International Conference of C.S.N Petroleum Training Institute (PTI) Conference Centre Complex Warri. pp. 241-244.

Okoh AI, Odjadjare EE, Igbinosa EO, Osode AN (2007). Wastewater treatment plants as a source of pathogens in receiving watersheds. Afr. J. Biotechnol. 6(25):2932-2944.

Oluyemi EA, Adekunle AS, Makinde WO, Kaisam JP, Adenuga AA, Oladipo AA (2006). Quality evaluation of water sources in Ife North Local Government Area of Osun State, Nigeria. Eur. J. Sci. Res. 15(3):319-326.

Pachepsky YA, Sadeghi AM, Bradford SA, Shelton DR, Guber AK, Dao T (2006). Transport and fate of manure-borne pathogens: Modeling perspective. J. Agric. Manage. 86(1-2):81-92.

Ramı´rez G, Martı´nez R, Herradora M, Castrejo´n F, Galvan E (2005). Isolation of Salmonella spp. from liquid and solid excreta prior to and following ensilage in ten swine farms located in central Mexico. J. Bioresour. Technol. 96(5):587–595.

Roth E, Gunkel-Grillon P, Joly L, Thomas X, Decarpenterie T, Mappe-Fogaing I, Lapoerte-Magoni C, Dumelié N, Durry G (2014). Impact of raw pig slurry and pig farming practices on physicochemical parameters and on atmospheric NO2 and CH4 emission of tropical soils, Uvéa Island (South Pacific). Environ. Sci. Pollut. Res. 21(17):10022-10035.

Rufete B, Perez-Murcia MD, Perez-Espinosa A, Moral R, Moreno-Casseles J, Paredes C (2006). Total and faecal coliform bacteria persistence in a pig slurry amended soil. J. Biosec. Livest. Efflu. 102(3): 211-215. .

Singh A, Agrawal M (2012). Effects of wastewater irrigation on the physicochemical characteristics of soil and metal partitioning in Beta vulgaris L. J. Agric. Res. 1(4):379-391.

Singh SN, Srivastav G, Bhatt A (2012). Physicochemical determination of pollutants in wastewater in Dheradun. J. Curr. World Environ. 7(1):133-138.

Standard Methods (2001). Standard methods for the examination of water and wastewater (20th Ed.). APHA-AWWA-WEF, American Public Health Association, Washington DC.

Tymczyna L, Chmielowiec-Korzeniowska A, Saba L (2000). Bacteriological and Parasitological pollution of the natural environment in the vicinity of pig farm. Pol. J. Environ. Stud. 9(3):209-214

Vanotti MB, Rashash DM, Hunt PG (2002). Solid-Liquid separation of flushed swine manure with PAM: Effect of wastewater strength. J. Am. Soc. Agric. Eng. 45(6):1959-1969.

Villamar CA, Cañuta T, Belmonte M, Vidal G (2011). Characteristics of swine wastewater by toxicity identification evaluation methodology (TIE). Water Air Soil Pollut. 223(1):363-369.

Whitefield A, Bate G (2007). A review of information on temporarily open/closed estuaries in the warm and cool temperate biogeographic regions of South Africa, with particular emphasis on the influence of river flow on These Systems. WRC Report No. 1581/1/07.

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Modeling sludge accumulation rates in lined pit latrines in slum areas of Kampala City, Uganda

Yvonne Lugali, Ahamada Zziwa*, Noble Banadda, Joshua Wanyama, Isa Kabenge, Robert Kambugu and Peter Tumutegyereize

Department of Agricultural and Biosystems Engineering, Makerere University, P.O. Box 7062, Kampala, Uganda.

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Disposal of faecal sludge particularly in slum areas is a difficult undertaking given the lack of space and resources. Inaccurate prediction of sludge accumulation rates (SAR) in pit latrines leads to unplanned pit latrine emptying. Given that the users and owners cannot afford the conventional emptying techniques frequently, inappropriate methods such as open defecation and emptying into storm drainages are employed which consequently contribute to environmental and health-related challenges. The main objective of this study was to develop a predictive model for sludge accumulation rates in lined pit latrines in slum areas of Kampala so as to guide routine management of pit latrines. This mathematical model was developed using a mass balance approach with a sample space of 55 lined pits. The developed model gave an average sludge accumulation rate of 81±25 litres/person/year with an efficiency of 0.52 and adjusted R^2 value of 0.50. The model was found to be sufficient and most suited for rental and public pit latrines given their bigger percentage in the slums. Further studies should include geo-physical characterization of soil and drainage of pit latrine sites so as to improve model accuracy.

Key words: Faecal, sludge accumulation rates, slum areas, lined pit latrines.

INTRODUCTION

Like many developing countries, the rural-urban migration has constrained local council authorities in Kampala City of Uganda to a level that they cannot cope with service delivery. The lack of proper urban housing has forced millions into informal settlements such as slums, where basic services including sanitation and hygiene are appalling. Slums are mainly located in areas of high ground water table (Fogg, 2008; Katukiza et al., 2014) that necessitate frequent pit emptying. The common emptying methods include use of vacuum tankers, manual emptying and the newer use of gulpers and nibblers. Most of the informal settlements are temporary

*Corresponding author. E-mail: engzziwa@gmail.com, zziwa@caes.mak.ac.ug.

and illegal (UN-HABITAT, 2007; Ministry of Lands, Housing & Urban Development (MLHUD), 2008) and based on the sanitation policy, on-site sanitation is the responsibility of the user (Kariuki et al., 2003). The business of pit emptying is mainly carried out by private pit emptiers using vacuum tankers. Emptying charges are mainly based on distance and the capacity of the truck. The charges also depend on the pit latrine characteristics such as depth and accessibility, faecal sludge characteristics, disposal site and geography of the site (Thye et al., 2011; Murungi and Van Dijk, 2014; Mikhael et al., 2014). As a result, pit emptiers charge a fee that ranges from 25 to 50 US dollars for a trip within a distance of 5 km. In cases where there is need to remove non faecal matter such as polythene bags, sanitary towels, clothes and in congested areas requiring disposal trips within a distance of 5 km where there is need for an extra vacuum pipe to be added, the price goes up by 3 to 10 US dollars (Murungi and Van Dijk, 2014).

Most of the residents in slum areas are low-income earners (Morella et al., 2008) thus, the cost of conventional pit emptying is high. It requires pit owners to actively save and plan for pit emptying. For pit latrines that cannot be emptied by tankers due to poor accessibility and cost, manual emptying is carried out (Kone and Chowdhry, 2012; WUP, 2003). This involves accessing the pit by inserting a hole on the side, and removing the sludge usually with simple tools such as spades, shovels and buckets (WSP, 2014; Eales, 2005). This practice is risky due to the pathogenic content of the sludge with the presence of dangerous micro-organisms such as Ascaris, *Salmonella species* (Parkinson and Quader, 2008; Murungi and Van Dijk, 2014). Besides, sludge is often dumped into the environment (Klingel et al., 2002) by simply disposing it off in the nearest streams and drainage channels (Schaub-Jones et al., 2006; Samuel, 2008). Given that the pit latrines are located in high water table areas, they are usually shallow. The pit latrines were not meant for solid waste disposal but given the poor management practices in the slum areas (Musiige, 2010) they fill up when the owners and users are not well-prepared for their emptying (Still et al., 2013). Desperate times call for desperate measures and so the pit latrines are either used when full or pit users seek alternative methods such as use of plastic bags and emptying into streams during the rainy season leading to a deplorable sanitation in the areas (Kulabako et al., 2007; Kimuli et al., 2016). This affects the environment and health of the residents in these areas with frequent opportunistic disease (e.g. cholera and typhoid) outbreaks among the slum dwellers in Kampala (Kulabako et al., 2010). In addition, the pit latrines in the slum areas are few compared to the population, so it is not an unusual sight to have a pit latrine with many users (Isunju et al., 2013) and there is always vandalism of the locks on the pit latrines and so the number of people using the pit latrines is usually higher than that reported (personal observation in the field data collection).

The responsibility of pit emptying and maintenance is still carried out by the pit owners or landlords for the case of rentals. Given that most of the landlords do not stay near their tenants or the pit latrine, pit latrines are usually emptied past the time they are full. There has been attempts by earlier researchers (Runyoro, 1981; Brouckaert et al., 2013) to address the issue of inaccurate prediction of sludge fill-up rates but this information was generalized for a wide range of pit latrines and it was not very applicable to the slum areas and it was necessary to determine the pit filling rates specifically for these areas (Bakare, 2014). It is against this background that the overarching objective of this study is to develop a predictive mathematical model capable of simulating sludge accumulation rates in lined pit latrines in the slum areas, and this model can be used to develop an algorithmic tool that would aid in the planning for emptying of the pit latrines.

MATERIALS AND METHODS

Study area

This study was carried out in the slum areas of Kamwokya, Luzira, Bwaise, Ndeeba, Banda, Nakulabye, Naguru, Kibuye and Kabalagala, all located within the five divisions of Kampala city (Figure 1). A total of fifty five pit latrines were studied from August, 2014 to July, 2015 and these were purposively chosen basing on the pit history available and willingness of the owner/ user to engage with the team carrying out the study. Pit latrines in the slum areas of Kampala city have unique characteristics unlike those study observed elsewhere in Africa (Bakare, 2014; Still and Foxon, 2012; Buckley, 2008). The majority of slum areas in Kampala are located in low lying areas (altitudes between 650-850 m above sea level) with high water tables. This means that the majority of pit latrines are shallow (not more than eight feet in depth). There is also frequent flooding especially during the rainy season and this is the reason the practice of emptying into streams is very common (Kulabako et al., 2007). The slum areas are unplanned informal settlements and with the exception of the public pit latrines that are built by the city authorities, the other pits are built with different construction designs and styles. The pits have very many users and most of them take on the solid waste disposal role as well (Isunju et al., 2013; MLHUD, 2008).

The pit latrines in the slum areas were classified into lined and unlined. Majority of the pit latrines were lined although areas such as Ndeeba, Luzira and Nakulabye that were not originally slum areas had most of the pit latrines as unlined. This study specifically focused on the lined pit latrines and these were divided into public pit latrines (more than 82 users); rental pit latrines (single pit used by several households but limited to only those households) and private pit latrines (used by single household).

Field data collection

Pit sampling

The type of material deposited in the pit was assessed basing on

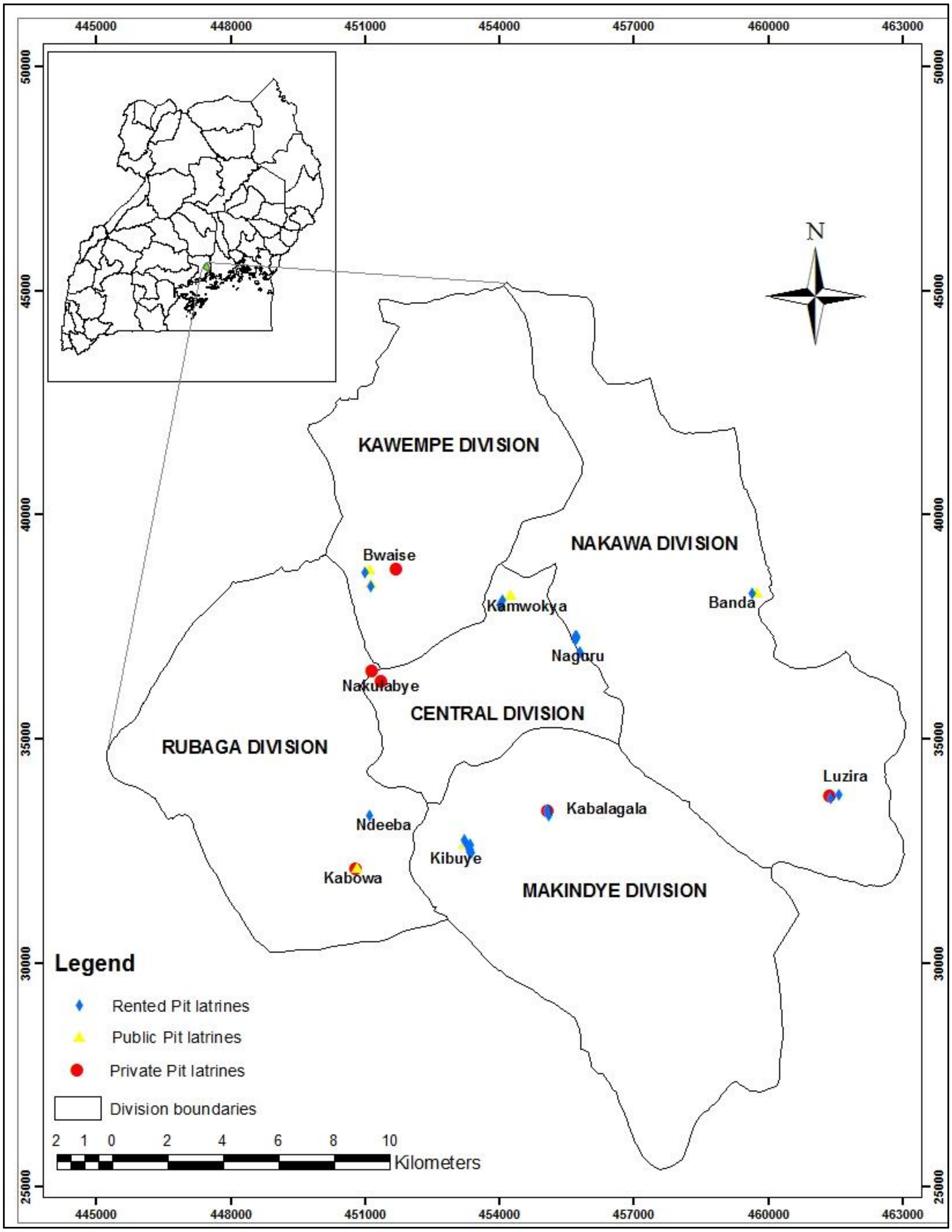

Figure 1. Map of Kampala showing selected pit latrines.

observation of the pit latrine contents in the pit and as the pit latrines were being emptied. Samples were collected using the pit sludge sampler (Figure 2). This tool was developed specifically for sampling faecal sludge. It was lowered into the pit latrine, adjusted by pushing the inner handle to ensure that the bottom can is open. The piston was then pulled to suck a reasonable quantity of sample. The sampling tool was removed from the pit latrine and the contents emptied into a sample container that was clearly marked

Figure 2. Sampling of faecal sludge: Pit sludge sampler (Left)-; Pit sludge sampler in action (Right).

Figure 3. Depth tool with sludge markings.

with the sample location, date and description of the pit latrine.

Pit size and depth measurements

The size of the pit latrine was measured with a tape measure for the length and width. The sludge depth was measured using a sludge depth measuring tool that was purposely developed for monitoring sludge depth changes (Figure 3). The tool was dipped into the pit latrine and the top layer of the sludge registered a mark on the tool which had a metric ruler attached to it. This reading gave the depth of sludge in the pit latrine.

Rate of degradation test

The rate of degradation on faecal samples measured the long term effect of aerobic and anaerobic degradation. The rate of degradation was measured in an experiment that was set up to measure mass loss rates at different moisture content levels; that is 80-90% and 90-100% (Buckley et al., 2008) which were the moisture content ranges found in the sampled pit latrines. Six samples were randomly selected for this experiment because of the time duration of the test (three months) and space requirements for the set-up. A small quantity (15 grams) of each of the selected samples was placed in sealed containers at each of the moisture levels and each sample was replicated three times. A control with only water was also set up to cater for the loss due to evaporation. The test was set up for three months with mass loss measurements taken weekly to determine the average mass loss over a predetermined period of time. After the three months, the percentage mass loss was computed and used to estimate the rate of degradation in the pit latrines. The rate of degradation was determined using the first-order kinetic equations:

$$dM/dt = -kt$$

With separation and integrating;

$$M_2 = M_1 e^{-kt} \tag{1}$$

Where M_1 is initial mass at start of experiment; M_2 is mass after time, t= three months. Given that all these values are known, k, first order reaction rate constant can be calculated.

Modelling sludge accumulation rates

Model selection

Earlier model approaches to sludge accumulation rates considered the amount of faecal matter that would go into a pit latrine and accumulated over a period of time as contributed to by the number of users (Runyoro, 1981; Wagner and Lanoix, 1958). For situations where this data was not available, sludge accumulation rates were assumed based on relative location to ground water and type of

Table 1. Model parameter

Parameter	Value and its unit	Source
Faecal excretion rate	260g/person/day	Niwagaba (2009)
Fractional content of non-faecal matter	25.8%	Zziwa et al. (2016)
Density of faecal excreta	1000g/l	Murphy, 2015
Yield of un-biodegradable material from degradation of biodegradable material, z	0.1 m^3/m^3	Brouckaert et al., (2013)
First-order kinetic constant, k	0.002	Rate of degradation test and comparable with Brouckaert *et al*, 2013

anal cleansing material used (Franceys et al., 1992; Mihelcic et al., 2009). This had some shortcomings for areas where this data was not available or easily accessible and hence, a new model approach was developed that considered the inflow of faecal matter into the pit latrine, the degradation process that takes place and the new solid material formed plus the outflow. In addition, non-biodegradable material that is thrown into the pit latrines was later included in the inflow (Still and Foxon, 2012; Bakare, 2014; Murphy, 2015). This more comprehensive modeling approach was adapted and modified for this study. As a consequence, lined pit latrines were considered only to have input of faecal matter in the pit latrine; degradation of the faecal matter and addition of new material from the degradation process. This was assumed so because lined pit latrines are closed up to the environment so there is negligible outflow or inflow of any material through the pit surface apart from through the pit drop hole. Thus, given the nature of the lined pit latrines in Kampala slums, a simple mass balance was considered for sludge accumulation as shown in Equation (2).

$$Inflow - \mathrm{Re}\,action - Outflow = Accumulation \quad (2)$$

The inflow included; urine, faeces, anal cleansing materials, detergents, rubbish and water used for cleaning; the reaction conversion was considered to be due to anaerobic processes though some aerobic processes could take place especially at the top of the pit latrine and the outflow included drainage from the pit and evaporation to the atmosphere (WINSA and WRC, 2011).

Model development

Model development included using a set of equations that took into account the input of faecal matter in the pit latrines, the degradation and finally the accumulation in the pit latrine. The rate at which a pit fills depends on the rate of addition of material in the pit and the rate of degradation. The process of model development followed the series of equations 3 to 6.

$$Faecal\ \ Inflow = N \times v \quad (3)$$

Where:
N is the number of users and v is the average volume of faecal excreta per person per year.
For a first-order reaction, the volume of initial faecal sludge reacting depends on the reaction rate and is expressed in the form of differential equation, that is

$$-r = dV/dt = kV$$

Which separating and integrating will give;

$$V = V_o e^{-kt} \quad (4)$$

Where:
r is the reaction rate, and k is the first order reaction rate constant; V_o is the initial volume of faecal sludge in the pit latrine; V is the volume of sludge in pit after degradation during time, t. For every 1 m^3 of biodegradable material, z m^3 of unbiodegradable residue is formed and this also contributes to the final volume of sludge in the pit latrine after an accumulated time as given in equation (5). The mass balance equation;

$$V_R = V + z(V_o - V) + V_n \quad (5)$$

Where: V_R is the total volume accumulated in a pit after time, t in years; z is the fraction of un-biodegradable residue; and V_n is the volume of non-degradable products in the pit latrine in litres. Sludge accumulation rates are in units of litres per person per year and so the volume of accumulated sludge in the pit latrine is converted into this format.

$$SAR = \frac{V_R}{n \times t} \quad (6)$$

Where:
n is the number of users for a particular pit latrine.

Table 1 is a summary of the model parameters as adopted from various researchers. The percentage of non-faecal matter, in the pit latrines was adopted from Zziwa et al. (2016). The faecal excretion (Table 1) rate used was based on the assumption of one stool per person per day on average. This was a realistic assumption in the slum areas as most adults were at work most of the day and used the pit latrines either in the morning or at night after work.

Model calibration

Model calibration was carried out using fifteen (15) pit latrines (27% of the total pit latrines) that were selected from the same slum and whose sludge accumulation rates were calculated using equations 4 and 5. The pit latrines were randomly chosen from the same slum to ensure that there was no variability caused by geo-physical factors and soil characteristics in the collected data. In the first phase, the sludge depth in the pit latrine was measured. In the subsequent phases, the sludge depth was measured and the change in the depth was used to calculate the sludge accumulation rate after a pre-defined period of time of five months followed by measurements after one month. The model was calibrated by correcting the model results with an addition factor that varied depending on the pit latrine and its calculated sludge accumulation rate (a form of correction factor) to ensure that they were similar to the field results.

Model validation

While the performance of the identified models is promising, the overall quality of the models had to be assessed by validation on separate data sets; thirty five (35) pit latrines from different slum areas in Kampala. The software used to run the model validation was GenStat discovery edition 4 and Microsoft excel 2010 to carry out the paired t-tests to determine a significant difference in the predicted and observed values.

Optimization criteria

In order to select the correct model structure, it was important to have a performance measure which captures the essential features of the model, so that the question of how good a model really is can be answered in a satisfying way. After all, the first and most straightforward test of appropriateness for any model is its ability to reproduce observed dynamics given relevant inputs. The criterion to be maximized in this paper was the R^2 value from regression model, which is often expressed in percentage form (Neter et al., 1990). The values obtained from the criterion reflect the percentage of output variation explained by the model (i.e., $y_h(t)$). Moreover, the R^2 as given by equation (7) does not address the tradeoff between the model accuracy and number of parameters.

$$R^2 = 100\left(1- \frac{\sum_{t=1}^{N}(y(t)-y_h(t))^2}{\sum_{t=1}^{N}(y(t)-mean(y(t)))^2}\right) \tag{7}$$

With y (t) the measured output at discrete time t and y_h (t) the model output at discrete time t, the performance index was used to evaluate the adequacy of the model.

It should be noted that R^2 with value 100% means a perfect fit between model and data over the entire data set; that is., y_h (t) equal to mean (y (t)) over the entire interval which is not satisfactory at all for the highly dynamic system in this study; R^2 with value 0% means that the model explains none of the variability of the response around its mean and R^2 with negative value means that the model predictions are even worse than the mean value. Another criterion used in this paper to assess the performance of the identified model was the Nash Sutcliffe value. The approach followed by Nash and Sutcliffe (1970) was to build a relative index of agreement or disagreement between the observed and computed values of the model and this can be used to compare model performance between periods. It basically measures the improvement made by the model in predicting sludge accumulation rates in comparison to the average value of the observed values. It starts from the sum of square errors given by equation 8;

$$F_o = \sum_{i=1}^{n}(Q_{obs,i} - \overline{Q_{obs}})^2 \tag{8}$$

Where *F* is the index of disagreement, $Q_{obs,i}$ and $Q_{sim,i}$ are the observed and predicted values at time step i, the sum being taken over n times steps of a pre-selected period. F is analogous to the residual variance of a regression analysis. The initial variance F_o is given by equation 9:

$$F_o = \sum_{i=1}^{n}(Q_{obs,i} - \overline{Q_{obs}})^2 \tag{9}$$

Where $\overline{Q_{obs}}$ is the mean of the observed values over the pre-selected period. Nash and Sutcliffe (1970) defined the efficiency of the model E as the proportion of the initial variance accounted for by the model as given by equation 10:

$$NS = 1 - \frac{F}{F_o} \tag{10}$$

The range *of NS* is from negative infinity to 1. A value of 1 indicates a perfect agreement and a value of 0 indicates that the model predictions are as accurate as the mean of the observed data. A negative value indicates that the model performs worse than the mean of the observed data.

RESULTS AND DISCUSSION

Predictive mathematical modeling

The percentage of non-faecal matter in the pit latrines as adopted from Zziwa et al. (2016) was taken to be 25.8% (Table 1), a value close to what was reported in earlier studies by Bakare (2014) and Still and Foxon (2012). The simulated results from the Equation (6) are shown in Figure 4 and Figure 5. The average sludge accumulation rate according to the developed model was 81 ± 25 litres/ person/ year. The model was calibrated by first removing outliers, that is, values with very high or very low sludge accumulation rates (greater than 350 litres/ person /year or lower than 30 litres/ person/ year). For pits that had values lower than 30 litres/person/year and those greater than 350 litres/per/year, the stated field emptying time

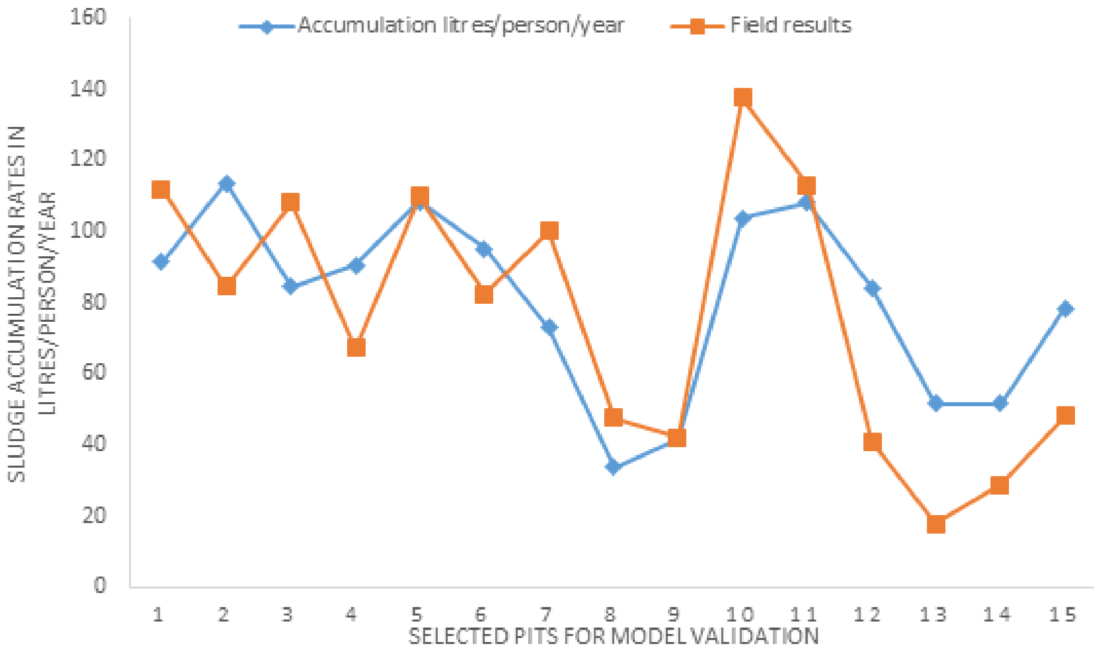

Figure 4. Comparison of simulated and experimental data.

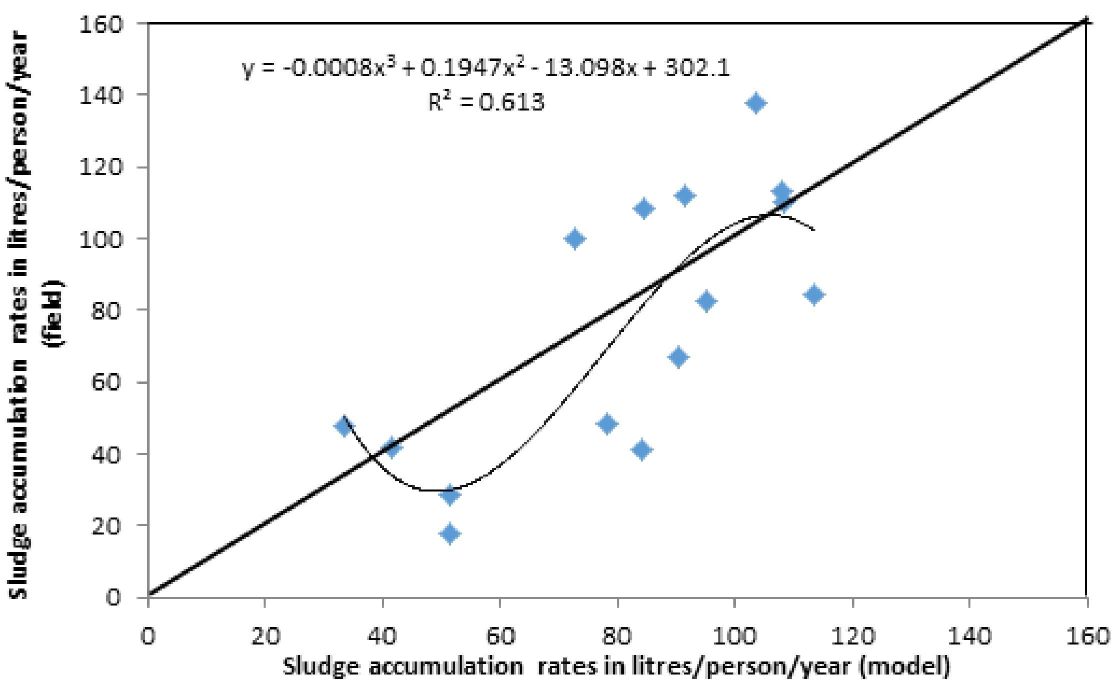

Figure 5. Simulation of model and field results showing R2 value.

was averagely twice a year while that which was calculated was much less (less than 3 months) which meant that some of the information given by the pit users might have been inaccurate. The model parameters were adjusted to ensure that the model fits the data used for its development.

Table 2. Paired t-test for sample means between modelled results and field measurements.

Parameter	Modeled results	Field measurements
Mean	80.64	75.98
Variance	647.37	1354.85
Observations	15	15
Pearson Correlation	0.73	-
Hypothesized Mean Difference	0	-
df	14	-
t Stat	0.72	-
P(T<=t) one-tail	0.24	-
t Critical one-tail	1.76	-
P(T<=t) two-tail	0.48	-
t Critical two-tail	2.14	-

The model value for sludge accumulation rate was almost twice that which was recorded in previous literature (Bakare, 2014; Brouckaert, 2013; Mara, 1984). This was unlike areas were previous studies focused, the pit latrines in the slums selected for this study were designed differently having varying dimensions, sizes, drop holes and found in different geographical locations. The higher value of sludge accumulation rate was mostly contributed by the non-biodegradable content of solid waste deposited in the pits along with faecal matter (Zziwa et al., 2016). This is particularly so because slum areas in Kampala city have a challenge with solid waste management and given that most of the plots of land are small, pit latrines double as rubbish pits as well (Niwagaba et al., 2014; Hoornweg and Bhada-Tata, 2012; Kulabako et al., 2004; Still et al., 2005).

Specific notes for model calibration

The sludge accumulation rates of fifteen out of the thirty five pit latrines simulated by the model were accurately predicted to within 70% - 90%. These pit latrines for which the model performed very well were considered to be ‘good’ pits and had common characteristics of having more than fifteen users (mainly public and rental pit latrines) and the non-faecal material accounted for 25.8% of the total matter in the pit latrine. Outliers (pits whose observed values were higher than 350 litres/person/year and lower than 30 litres/person/year) had to be discarded from the model since these results were not realistic in nature given the parameters involved. For instance, it was unlikely that a pit latrine with less than 10 people could have a sludge accumulation rate of close to 500 litres/ person/ year. This is because with such a value of SAR and given the size of the pit latrines, there would be the need to empty the latrines every week which is not the case in reality. It was suspected that some inaccurate information about the pit characteristics was given during sampling.

Optimization criteria

The developed model may be considered efficient for the predicted model results of the fifteen pit latrines given the Nash Sutcliffe value of 0.52 and the adjusted R^2 value of 0.50. Values of R^2 in ranges of 0.8 and above are considered to be acceptable model accuracy values. However, models that try to predict human behaviour generally have low R^2 values of less than 0.5 (Frost, 2013). The model developed accounted for half the variation in sludge accumulation rates in pit latrines in slum areas. This low value could be attributed to poor pit maintenance and not ensuring that the pit bottoms are not fully sealed. Hence, the observed values could have been impacted upon by geo-physical conditions of the soil and drainage of pit latrine sites (Kulabako, 2005; Kulabako et al., 2007).

Comparison of predicted and experimental data

The model results were compared with the experimental data. Results from a paired t-test showed that the Pearson’s correlation to be 0.73 which indicated a strong relationship between the model and the field results (Table 2). The mean value of the sludge accumulation rates given by the model and that of the field results were comparable as there was no significant difference between them ($p>0.05$) and this means that the model could be used to estimate the sludge accumulation rates in the slum areas. An equality line (1:1 line) was drawn to indicate a measure of agreement between the model and field results. The equality line (Figure 5) showed that the model was a good approximation since it showed an even distribution between the points. The 1:1 line in Figure 5 shows that the model was efficient for values between 40 and 110 litres/person/year. For values below this range, the model is overestimated the sludge accumulation rates while for those above the range, the model is underestimated. This is because the model considered a constant value for the non-faecal matter (Zziwa et al., 2016) which in reality is not the case. Hence, for pits with better use and less non-faecal matter; the model did not capture this and so overestimated the SAR while it underestimated the same for pit latrines that had more non-faecal matter. The developed model was however, found to be a better approximation of sludge accumulation rates in slum areas since it considered solid waste deposited in the pit latrines and was able to cover a range of pit latrines with different designs and user behaviour unlike previous studies that had been carried out (Brouckaert, 2013; Bakare, 2014; Murphy, 2015). The model was found to be a better approximation for rentals and public pit latrines compared to the private pit latrines, given their numbers were more in the study.

CONCLUSION AND RECOMMENDATIONS

The average sludge accumulation rate determined by model was 81 ± 25 litres/person/year. Model validation showed that the developed model was 52% efficient and accounted for 50% of the variation in the sludge accumulation rates. The model is sufficient for prediction of filling rates in the public and rental pit latrines within the studied slums given the variation in pit latrine designs, user behaviour, pit dimensions, location and solid waste deposal patterns. The model can therefore be adequately used for prediction of sludge accumulation rates of lined pit latrines in slum areas. The model was found to have limitations for determining sludge accumulation rates for private pit latrines and those pits that are managed properly. This study did not provide a specific emptying plan for each pit latrine but with the information provided on the sludge accumulation rates and the estimates provided, each pit latrine owner is able to adequately plan for emptying, given the different sizes of the pit latrine. Further studies can be taken on the effect of geo-physical factors such as soil characteristics and drainage patterns on sludge accumulation rates and a study to model sludge accumulation rates in unlined pit latrines in the slum areas. .

Conflict of interest

The authors have not declared any conflict of interest.

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REFERENCES

Bakare BF (2014). Scientific and Management Support for Ventilated Improved Pit Latrines (VIP) Sludge Content. School of chemical engineering. University of KwaZulu-Natal Durban (Unpublished).

Brouckaert CJ, Foxon KM, Wood K (2013). Modelling the filling rates of pit latrines. Pollution Research Group, School of Engineering, University of KwaZulu-Natal, Durban.

Buckley CA, Foxon KM, Brouckaert CJ, Nwaneri C, Balboni E, Couderc AMD (2008). Scientific support for the design and operation of ventilated improved pit latrines (vips). (S. of C. E. U. of K.-N. Pollution Research Group, Ed.). Durban: Water Research Commission. doi:TT 357/08.

Eales K (2005). Sanitation partnership series: Bringing pit emptying out of the darkness: A comparison of approaches in Durban, South Africa, and Kibera, Kenya. Building Partnerships for Development. Available at: http://www.bpdws.org/bpd/web/d/doc_131.pdf?statsHandlerDone=1

Fogg NA (2008). The Blame Game: Water and Sanitation in Kampala's Urban Slums. Independent Study Project (ISP) Collection. Paper 98.

Franceys R, Pickford J, Reed R (1992). A guide to the development of onsite sanitation. World Health Organization: Geneva.

Frost J (2013). Regression Analysis: How Do I Interpret R-squared and Assess the Goodness-of-Fit? Available at: http://blog.minitab.com/adventures-in-statistics/regression-analysis-how-do-i-interpret-r-squared-and-assess-the-goodness-of-fit.

Hoornweg D, Bhada-Tata P (2012). What a waste; A global review of solid waste management. Urban Development Series Knowledge Papers, World Bank.

Isunju JB, Etajak S, Mwalwega B, Kimwaga R, Atekyereza P, Bazeyo W, Ssempebwa JC (2013). Financing of sanitation services in the slums of Kampala and Dar-es-Salaam. Health 5(4):783-791.

Kariuki M, Collignon B, Taisne R, Valfrey B, Plummer J (2003). Better Water and Sanitation for the Urban Poor: Good Practice from Sub-Saharan Africa. Kenya: Water utility partnership for capacity building (WUP) Africa.

Katukiza AY, Ronteltap M, Oleja A, Niwagaba CB, Kansiime F, Lens PNL (2014). Identification of Sustainable Sanitation Options for Urban Slums- A Case of Bwaise III in Kampala Uganda. Department of Environmental Resources, UNESCO-IHE Institute for Water Education, Netherlands.

Kimuli D, Zziwa A, Banadda N, Kabenge I, Kiggundu N, Kambugu RK, Wanyama J, Tumutegyereize P, Kigozi J (2016). Quantification of Physico-Chemical Characteristics and Modeling Faecal Sludge Nutrients from Kampala City Slum Pit Latrines. Int. J. Res. Eng. Adv. Tech. 3(6):129-141.

Klingel F, Montenegro A, Koné D, Strauss M (2002). Fecal Sludge Management in Developing Countries: A planning manual, (April), 55.

Kone D, Chowdry S (2012). Business analysis of faecal sludge management: Emptying and transportation services in Africa and Asia. Draft Final Report.

Kulabako NR, Nalubega M, Roger T (2004). Characterization of Peri-Urban Anthropogenic Pollution in Kampala, Uganda. 30th WEDC International Conference, Vientiane, Lao PDR.

Kulabako NR, Nalubega M, Thunvik R (2007). Study of the impact of land use and hydrogeological settings on the shallow groundwater quality in a peri-urban area of Kampala, Uganda. Sci. Tot. Environ. 381(1):180-199.

Kulabako NR, Nalubega M, Wozei E, Thunvik R (2010). Environmental health practices, constraints and possible interventions in peri-urban settlements in developing countries–a review of Kampala, Uganda. Int. J. Environ. Health Res. 20(4):231-257.

Mara DD (1984). The Design of Ventilated Improved Pit Latrines. TAG Technical Note 13:83.

Mihelcic JR, Fry LM, Myre EA, Phillips LD, Barkdoll BD (2009). Field Guide to Environmental Engineering for Development Workers: Water, Sanitation and Indoor Air. Reston: ASCE Press.

Mikhael G, Robbins, DM, Ramsay JE, Mbéguéré M (2014). Methods and means for collection and transport of faecal sludge. In: Strande L, Ronteltap M, Brdjanovic D (Eds.), Faecal Sludge Management – Systems Approach Implementation and Operation. IWA Publishing, London, UK: pp. 67-96.

Ministry of Lands, Housing and Urban development, MLHUD (2008). A situation analysis of slums in Uganda and national slum upgrading strategy and action plan. Department of Human Settlements. UNDP program.

Morella E, Foster V, Banerjee GS (2008). Climbing the Ladder: The State of Sanitation in Sub-Saharan Africa. Africa Infrastructure Country Diagnostic (AICD) Background Paper 13 (Phase 1). The World Bank, Washington, DC, USA.

Murphy CM (2015). Solids Accumulation rates of Pit latrines at Rural Schools in Nimba County, Liberia. Graduate Theses and Dissertations, University of South Florida, Scholar Commons.

Murungi C, Van Dijk MP (2014). Emptying, transportation and disposal of faecal sludge in informal settlements of Kampala Uganda: the economics of sanitation. Habitat Int. 42:69-75.

Musiige R (2010). Pollution of groundwater in shallow aquifers-a critical moment in Uganda. Groundwater Quality Sustainability. XXXVIII IAH Congress, Krakow.

Nash JE, Sutcliffe JV (1970). River flow forecasting through conceptual models. Part 1-A discussion of principles. J. Hydrol. 27(3):282-290.

Neter J, Wasserman W, Kutner M (1990). Applied linear statistical models, 2nd edn. Richard E. Irwin, Inc. Home-wood, IL.

Niwagaba CB (2009). Treatment technologies for human faeces and urine. Doctoral thesis, Swedish University of Agricultural Sciences, Uppsala.

Niwagaba CB, Mbéguéré M, Strande L (2014). Faecal Sludge Quantification, Characterisation and treatment objectives. Faecal Sludge Management; Systems Approach for Implementation and Operation. IWA Publishing.

Nwaneri CF (2009). Physico-Chemical characteristics and biodegradability of contents of Ventilated Improved Pits in eThekwini Municipality. Master of Science dissertation. School of Biological and Conservation Science. University of KwaZulu-Natal. South Africa.

Parkinson J, Quader M (2008). The challenge of servicing on-site sanitation in dense urban areas: Experiences from a pilot project in Dhaka. Waterlines 27(2):149-163.

Runyoro SK (1981). Pit Latrine Emptying. Department of Civil Engineering, Loughborough University of Technology (Unpublished Thesis).

Samuel E (2008). The financing of sanitary facilities in urban slums of Kampala. A case study of Bwaise slum. Master's degree thesis, Makerere University. Unpublished work.

Schaub-Jones D, Eales K, Tyers L (2006). Sanitation partnerships: harnessing their potential for urban on-site sanitation. BPD Sanitation series.

Still D, Foxon K (2012). Understanding Sludge Accumulation in VIPs and Strategies for Emptying Full Pits. Water research Commission. 1. DOI: 978-1-4312-0291-1

Still D, Riordan MO, McBride A, Louton B (2013). Adventures in search of the ideal portable pit-emptying machine. Waterlines 32:187-199.

Still DA, Salisbury RH, Foxon KM, Buckley CA, Bhangwan JN (2005). The challenges of dealing with full VIP latrines.

Thye YP, Templeton MR, Ali M (2011). A critical review of technologies for pit latrine emptying in developing countries. Crit. Rev. Environ. Sci. Technol. 41:1793-1819.

UN-HABITAT (2007). Situational Analysis of Informal Settlements in Kampala. Cities without slums sub-regional programme for Eastern and Southern; Kivulu (Kagugube) and Kinawataka (Mbuya 1) Parishes.

Wagner EG, Lanoix JN (1958). Excreta disposal for rural areas and small communities. Geneva Switzerland: World Health Organisation (WHO).

Water and Sanitation Program, WSP (2014). The Missing Link in Sanitation Service Delivery: A review of faecal sludge management in 12 cities. Targeting the urban poor and Improving Services in small towns, a research brief.

WINSA, WRC (2011). What Happens When the Pit is Full? Developments in on-site Faecal

Wood K (2013). Modeling fill rate with an unsteady-state mass balance. Transformation of feacal sludge in VIP.

WUP (2003). Better Water and Sanitation for the Urban Poor. Kenya, European Communities and Water Utility Partnership.

Zziwa A, Kabenge I, Kayondo H, Lugali Y, Kambugu RK, Wanyama, J (2016). Fractional content of non-feacal matter and its contribution to filling rates of pit latrines in Kampala slums. Glob. J. Eng. Sci. Res. Manage. 3(2):36-45.

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