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African Journal of Environmental Science and Technology

Table of Contents: Volume 11 Number 6 June 2017

ARTICLES

Review on heavy metal pollution in major lakes of India: Remediation through plants 255
Jamshed Zaidi and Amit Pal

Deforestation in power line construction in the Central African Region 266
Tchuidjan Roger, Biya Motto Fréderic, Tachago Raymond, Nguimbe Bernard, Mbinkar Edwin Nyuysever and Tabe Moses Ndem

Bioremediation of soil and water polluted by cyanide: A review 272
Lovasoa Christine Razanamahandry, Hela Karoui, Harinaivo Anderson Andrianisa and Hamma Yacouba

Seasonal macrophyte diversity and water quality in an urban wetland 292
Robert Calvin Okello, Alfonse Opio and Frank Kansiime

Using Citizen Science Approach to monitor water, sanitation and hygiene Related Risks in Karonga Town, Malawi 305
Elijah M. M. Wanda, Mtafu Manda, James Kushe, Orton Msiska, Chrispin Mphande, Dominic Kamlomo and Jean Kaunda

Geoelectrical logging for well screening in prolific aquifers in Ubima, Ikwerre Local Government Area, River State, Nigeria 324
G. I. Alaminiokuma, T. Warmate and J. E. Emudianughe

Impacts of bacterial pollution on hand-dug well water quality in Enugu, Enugu State, Nigeria 331
Onuigbo A. C., Onyia C. E., Nwosu I. G. and Oyeagu U.

Kinetic analysis of anaerobic sequencing batch reactor for the treatment of tannery wastewater 339
Andualem Mekonnen, Seyoum Leta and Karoli Nicholas Njau

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African Journal of Environmental Science and Technology

Review

Review on heavy metal pollution in major lakes of India: Remediation through plants

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The historical lakes of India are getting over burdened with the loads of pollution due to the rapid growth of small scale industries, nutrient enrichment and other anthropogenic activities. The high concentrations of metals in the water bodies, their entry into ecological food chain and the resulting health effects are of great concern to the researchers in the areas of ecology. Due to the problems associated with the conventional methods of pollutant removal, phytoremediation method is gaining more attention. The plants to be used in this technique, it must have high capacity of metal absorption, its accumulation and reduction in the time of decontamination in an ecosystem. It is an environment friendly and cost beneficial technique for the removal of toxicants from the environment. This article present over view on status of heavy metal pollution in lake and remediation through plants in Indian context may helpful to researcher not only in India but around the world also. This base line data can help governmental and non-governmental organization for the management of water pollution.

Key words: Lakes, heavy metals, toxicants, phytoremediation, India.

INTRODUCTION

A lake can be viewed as the most attractive and expressive characteristic of a landscape. In India there are some natural lakes that lies in the Himalayan region and in the flood plains of the Indus, Ganga and Brahmaputra. These lakes with various dimensions possess different names that are summarized in Table 1. However during the last 1000 years a large number of man-made water bodies were constructed in the western and peninsular India (Gopal et al., 2010).

Water qualities in such lakes have been studied by many researchers in different areas (states) of the country and are tabulated in Table 2 and the location of these lakes are shown in Figure 1. Accumulation of heavy metals in the freshwater ecosystem is a problem of global concern. On the recent time scale due to the continuous rise in the population, rapid industrialization, toxic chemicals used by agricultural industries and the technologies involved in waste disposals, there is increase in the rate of release of pollutants into the environment than the rates of their purification. Entry of these toxic metals to the ecosystem results in geo-accumulation, bio accumulation and bio magnifications

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Table 1. Various water bodies' dimensions have different names in India.

S/N	Name of lake In India	Features of lake
1	Bawri	Small, usually shallow pond
2	Beel	Oxbow lakes in Assam
3	Bheel	Oxbow lakes in Bengal
4	Bheri	Brackish water impoundments
5	Chaur	Oxbow lakes in Bihar
6	Jheel	Usually large and deep lakes
7	Johad	Shallow water bodies of Rajasthan
8	Kayal	Backwater lakes in Kerala
9	Kere	Tanks in Karnataka
10	Maun	Oxbow lakes in Uttar Pradesh and Bihar
11	Pat	Shallow floodplains in Manipur
12	Sagar	Very large and deep manmade lakes
13	Sar	Sanskrit word for lakes
14	Sarovar	Sanskrit Words for shallow lakes
15	Tal	Large spread of water (oxbow)
16	Talao	Usually a small pond natural/manmade
17	Talaiya	Shallow small pond
18	Tso	Tibetan word for lakes

Source: Conservation and management of lakes-an Indian perspective (2010).

(Lokeshwari and Chandrappa, 2006). The major sources of heavy metals are summarized in Table 3 and the most contaminated states of heavy metal in India are also depicted (Table 4 and Figure 2).

Heavy metals can be described as any metallic elements which have a relatively high density and are poisonous even at very low concentration in every organism (Lenntech, 2004). These groups of metals and metalloid have atomic density greater than 4 gm/cm^3 which is 5 times higher than water (Nriagu, 1989; Garbarino et al., 1995).

In counties like ours, with developing economies, the sustainable development, efficient utilization and effective management of their water bodies should be the acceptable strategy for economic growth. But in the recent past improper management and the ineffective utilization of the natural resources for various purposes has resulted in various problems such as water logging and salinity in the field of agriculture and heavy metals contamination due to mining industries and municipal uses (Rai and Pal, 2001; Kumar et al., 2008). According to the Indian standards, the maximum permissible level of these heavy metals is shown in Table 5. Status of heavy metal pollution in lakes of different states of the country is summarized in Table 6.

Phytoremediation as an emerging clean up technology used for pollutes groundwater and wastewater treatment is described as the engineered use of green plants (including grasses, forbs, and woody species) to remove or eliminate environmental contaminants such as heavy metals, trace elements, organic compounds and radioactive compounds in many aquatic ecosystems. Macrophytes is one of the important component of the aquatic ecosystems, these can be used as an effective accumulator of heavy metals and is also a food source for many aquatic invertebrates (Preetha and Kaladevi, 2014).

ROLE OF PLANTS IN REMEDIATION

Plants can be used to cleanup or remediate contaminated sites by several ways in order to remove contaminants from the soil, sediment, or water. Such plants can breakdown or decompose organic pollutants or may stabilize metal pollutants by acting as filters or traps. Plants usually take contaminants through their root system in which the main mechanism for controlling the contaminant's toxicity lies. The root system of plants provides wide surface area to absorb and accumulate the nutrients and water that is required for growth and other non-essential pollutants. Research is still going on finding the use of trees rather than smaller plants for affective treatment in deeper contamination because tree roots can penetrate more deeply into the soil. Further polluted ground water can undergo treatment by pumping out the water from the ground and using plants to treat the contamination.

Table 2. Status of water quality of different lakes of India.

States	Lakes	Parameter									References
		pH	EC	Turb.	DO	TDS	BOD	TN	TP	Temp.	
Andhra Pradesh	Hussain Saga	7.78	-	7.2	2.26	680	2.69	5.4	7.2	30	Sailaja and Reddy (2015)
Chandigarh	Sukhna Lake	8.1	-	42	6.8	-	3.0	0.16	0.6	25	Chaudhry et al. (2013)
Chattisgarh	Bhilai	6.5	807.6	-	5.14	452.3	-	0.26	-	25	Jena et al. (2013)
Haryana	Sanhit Sarovar	8.8	-	18.5	-	-	14.5	1.5	0.6	31	Kazmi et al. (2013)
Gujarat	Sarkhej Lake	8.7	489	18	3.36	184.2	1.21	8.22	0.77	20	Umerfaruq et al. (2015)
Himachal Pradesh	Renuka Lake	7.3	-	-	6.66	363.8	1.81	4.29	0.16	-	Singh and Sharma(2012)
Jammu and Kashmir	Wular Lake	7.8	232.3	-	9.3	143.5	-	0.80	-	14.1	Yaseen et al. (2015)
Karnataka	Bhattrahalli Lake	8.51	1707	3.25	4.7	584.00	12.00	-	-	-	Veena et al. (2016)
Madhya P.	RoopSagar	7.4	_	6.5	4.2	-	4.2	0.19	0.24	26	Vaheedunnisha and Shukla (2013)
Maharashtra	Futala Lake	7.8	-	1.1	7.8	263	2.4	3.5	1.4	26	Kazmi et al. (2013)
Manipur	Loktak Lake	7.31	-	-	8.58	71.33	5.07	-	-	-	Laishram and Dey (2014)
Mizoram	PalakDil	8.0	70	-	5.8	51.03	3.4	47	0.42	32	Lalmuansangi and Lalramnghinglova (2014)
Odisha	Chillka Lake	8.03	-	-	7.56	26.66	3.5	16.74	0.17	33.5	Patra et al. (2010)
Punjab	Harike Lake	8.37	-	-	9.35	445.6	4.5	4.74	-	-	Parmar and Bhardwaj (2013)
Rajasthan	Mansagar Lake	8.0	-	35.8	15.4	1840	2.1	11.3	0.2	21.8	Kazmi et al. (2013)
Tamilnadu	Kolavai Lake	7.38	811	1.1	7.8	576	11.2	-	0.08	30.8	Babu and Selvanayagam (2013)
Telangana	Kargil Lake	7.8	258	4.6	15.8	213	3.9	9.2	4.8	24.5	Bondugula and Rao(2015)
Tripura	Rudrasagar Lake	9	115	25	8.6	-	4.1	7.0	-	-	Pal et al. (2016)
Uttarakhand	Naini Lake	8	-	0.34	9.9	440	2	4.6	0.2	18.8	Kazmi et al. (2013)
Uttar Pradesh	KeeratSagar	8.3	778.2	8.2	6.34	502	3.48	10	_	25.4	Pal et al. (2013)

EC, Electrical conductivity; Turb., Turbidity; DO, Dissolved oxygen; TDS, Total dissolved solid; BOD, Biochemical oxygen demand; TN, Total nitrogen; TP, Total phosphorus.

Plants roots releases organic and inorganic compounds (root exudates) in the rhizosphere that causes changes at the soil root interface. This is an effective alternative technology which can replace mechanical conventional clean-up technologies that often needs high capital inputs, labour and energy. Phytoremediation is an in-situ remediation technique that uses the inherent capacities of living plants. It is also an eco-friendly, solar energy driven clean-up technology based on the principle of using nature itself to clean nature. Some hyperaccumulator species and their accumulation level are summarized in Table 7.

Most of the phytoremediation processes are targeted on inorganic pollutants through different attempts which is termed as phytoextraction (the utilization of metal accumulating species to transport and accumulate metals from the soil to roots and above ground biomass), rhizofiltration (the utilization of plant roots to absorb, precipitate and concentrate toxic contaminants from polluted effluents, phytovolatilization (some metal pollutants such as As, Hg and Se occur in gaseous forms in the environment; scientists have recently discovered genetically-modified plants that are capable of absorbing metals in their elemental forms from the soil, thus converting them biologically to gaseous species within the plants and release them into the atmosphere) and phytostabilization (the utilization of plants in lowering down the mobility of metals) (Mandal, 2014).

APPLICATIONS OF PHYTOREMEDIATION IN INDIA

One of the most promising applications of

Figure 1. Location of studies lakes.

phytoremediation techniques is the possibility of deriving additional benefit from the plant system during or after the prevention or clean-up technology. Ali et al. (1999) studied the physico- chemical parameters of Nainital lake and the functions of macrophytes in phytoremediation and biomonitoring of metallic ions that are toxic in nature. Reports showed that the concentrations of metals such as Cr, Cu, Fe, Mn, Ni, and Pb are much higher than their recommended maximum permissible limits. Metal extracting capacity of existing plants is highly significant for biomonitoring studies. Sharma et al. (2014) prove to be an effective phytoremediation technique to restore the quality of water by harvesting both the submerged and the floating species from the littoral zone of the lake. Recently large numbers of submerged, free floating and emergent plant species have been recognized that can

Table 3. Different sources of heavy metals (Lone et al., 2008).

S/N	Heavy metals	Sources
1	As	Semiconductors, wood preservatives, mining and smelting coal power plants, herbicides, volcanoes, petroleum refining, animal feed additives
2	Cu	Electroplating industry, mining, biosolids , smelting and refining
3	Cd	Geogenic sources anthropogenic activities metal smelting and refining,fossil fuel burning, application of phosphate fertilizers, sewage sludge.
4	Cr	Electroplating industry, sludge, solid waste, tanneries
5	Pb	Mining and smelting of metalliferous ores, burning of leaded gasoline,municipal sewage, industrial wastes enriched in Pb, paints
6	Hg	Volcano eruptions, forest fire, emissions from industries producing causticsoda, coal, peat and wood burning
7	Se	Coal mining, oil refining, combustion of fossil fuels, glass manufacturing industry, chemical synthesis (e.g., varnish, pigment formulation)
8	Ni	Volcanic eruptions, land fill, forest fire, bubble bursting and gas exchange in ocean, weathering of soils and geological materials
9	Zn	Electroplating industry, smelting and refining, mining, biosolids

As, Arsenic; Cu, Copper; Cd, Cadmium; Cr, Chromium; Pb, Lead; Hg, Mercury; Se, Selenium; Ni, Nickel; Zn, Zinc.

Table 4. Major heavy metals contaminated states in India (Mandal, 2014).

Chromium	Lead	Mercury	Arsenic	Copper
Ranipet, Tamil Nadu	Ratlam, Madhya Pradesh	Kodaikanal, Tamil Nadu	Tuticorin, Tamil Nadu	Tuticorin, Tamil Nadu
Kanpur, Uttar radesh	Bandalamottu Mines, Andhra Pradesh	Ganjam, Orissa	West Bengal	Singbhum Mines, Jharkhand
Vadodara, Gujarat	Vadodara, Gujarat	Singrauli, Madhya Pradesh	B a l l i a Uttar Pradesh	Malanjkhand, Madhya Pradesh
Talcher, Orissa	Korba, Chattisgarh			

be effectively used in phytoremediation of metal pollutants from water bodies (Rai et al., 1995; Nirmal Kumar et al., 2006; Prasad, 2007; Shah and Nongkynrih, 2007; Shrivastava, 2008; Dixit and Dhote, 2009; Mishra and Tripathi, 2009; Narendra et al., 2012; Swain et al., 2014; Phukan, 2015; Shafi et al., 2015; Kumar and Chopra, 2016; Shekhar and Prashik, 2016). Researches on phytoremediation in different states of India are shown in Table 8.

CONCLUSION

Phytoremediation for pollution control has many drawbacks as well and it requires further intensive research on plants and site-specific conditions. It is comparatively a slow process than other treatment processes viz chemical, physical process. Plants with low production yields and reduced root systems do not carry out effective phytoremediation and do not prevent the leaching of contaminants into the aquatic system. Environmental conditions play an important role in phytoremediation as the growth and the survival of plants are negatively affected to extreme environmental conditions. In this technology multiple metal polluted bodies are desire more metal accumulator species and thus it requires a broad range research studies before its applications. There are also several limitations like metals must be in their bio-available form to plants. The metals will pass down the root systems without accumulation if it is water soluble. Despite these drawbacks, phytoremediation technology has many applications

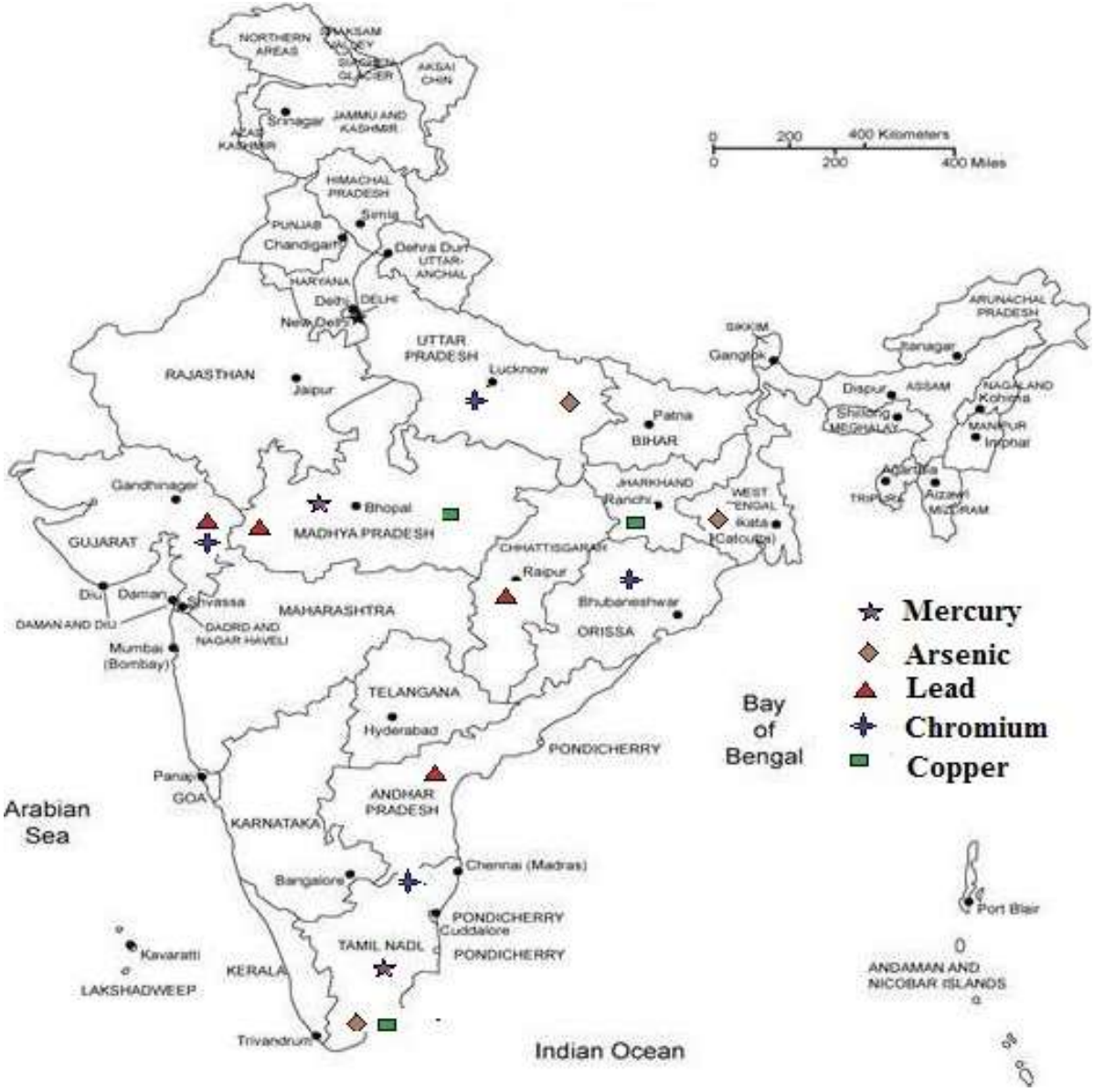

Figure 2. Heavy metals contaminated states of India.

Table 5. Permissible level of heavy metal.

S/N	Heavy metals	BIS, IS:10500- Desirable (mg/l)	Indian Council of Medical Research (mg/l)
1	Iron	0.3	1.0
2	Copper	0.05	3.0
3	Fluoride	0.06-1.2	1.5
4	Arsenic	0.05	0.05
5	Cadmium	0.01	0.01
6	Chromium	0.05	0.05
7	Lead	0.1	0.1
8	Mercury	0.001	0.001
9	Nickel	0.02	0.02

Table 6. Status of heavy metals pollution in different lakes of India.

States	Lakes	Heavy metals						References
		Pb	Fe	Mn	Cu	Cr	Zn	
Andhra P.	HussainSagar	0.84	-	-	-	-	-	Suneela et al. (2008)
Chattisgarh	Bhilai	0.26	0.822	-	0.002	0.326	0.0533	Tiwari et al. (2015)
Gujarat	Sarkhej lake	0.06	-	0.63	-	-	-	Patel and Vediya (2012)
Himachal P.	Renuka lake	0.35	1.49	0.87	0.00	-	0.15	Singh and Sharma (2012)
Jammu and Kashmir	Wular lake	0.9	-	0.9	0.6	-	2	Sheikh et al. (2014)
Karnataka	Bhattrahalli lake	0.002	0.283	0.059	0.003	0.003	0.009	Veena et al. (2016)
Kerala	Ashtamudi lake	0.001	8.41	-	0.02	0.01	0.03	Karim and Williams (2015)
Maharashtra	Futala lake	0.026	0.035	-	-	0.042	0.048	Puri et al. (2011)
Manipur	Loktak lake	0.7	-	-	-	1.3	3.6	Singh et al. (2015)
Odisha	Chilika lake	0.385	1.1	-	0.29	0.07	0.247	Nayak et al. (2010)
Punjab	Harike lake	0.53	1.30	0.02	0.26	0.12	0.69	Brraich and Jangu (2015)
Tamilnadu	Kolavai lake	0.138	7.820	-	0.126	0.008	0.232	Babu et al (2013)
Uttar P.	Laxmi Tal	1.52	1.49	1.64	0.07	0.33	0.02	Sharma et al (2014)
Uttarakhand	Nainital Lake	…	0.011	0.007	0.024	-	0.216	Gupta et al. (2010)
Madhya P	Shahpura lake	0.06	-	-	0.39	-	-	Anu et al. 2011
Andhra P	Cherlapally Lake	-	0.50	8.2	0.50	-	0.22	Amruthakalyani and and Gangadhar (2014)
Rajsthan	Lake Anasagar	0.122	0.660	-	0.072	-	0.963	Dutta et al. (2009)
Tamil Nadu	Chemberambakkam Lake	0.29	0.284	0.052	0.019	0.035	0.026	Batvari and Surendran (2015)

Pb, Lead; Fe, Iron; Mn, Manganese; Cu, Copper; Cr, Chromium; Zn, Zinc.

Table 7. Some hyperaccumulator species and their accumulation level.

Plant species	Metal	Results	Reference
C. papyrus	Pb	3.14 mghg^{-1}	Mugisa et al. (2015)
Phragmites australis	Pb	1.68 mghg^{-1}	
Hydrilla verticillata	Cd	3.58 mg g-^{1} DW at 1 mg l^{-1}con.	He et al.(2016)
Hydrocotyle ranoncloides	Cd	6.28 mg kg^{-1}	VahdatiRaad and Khara (2012)
	Pb	77.80 mg kg^{-1}	
Ceratophyllum demersum	Cd	4.46 mg kg^{-1}	VahdatiRaad and Khara (2012)
	Pb	53.11 mg kg^{-1}	
Alyssum heldreichii	Ni	11800 (mg kg^{-1})	Bani et al. (2010)

Table 7. Cont.

Alyssum markgrafii	Ni	19100 (mg kg^{-1})	
Alyssum bertolonii	Ni	10900 (mg kg^{-1})	Li et al. (2003)
Alyssum caricum	Ni	12500 (mg kg^{-1})	
Alyssum corsicum	Ni	18100 (mg kg^{-1})	Bani et al. (2010)
Alyssum murale	Ni	4730–2010 (mg kg^{-1})	
Myriophyllum spicatum	Cu	74.97 (mg kg^{-1})	Kamel (2013)
Ceratophyllum demersum,	Cu	96.3 mgl^{-1}	
Eicchornia crassipes,	Cu	53.8 mgl^{-1}	
Lemna gibba,	Cu	36.4 mgl^{-1}	
Phragmites australis	Cu	129.21mgl^{-1}	
Typha domingensis.	Cu	153.2 mgl^{-1}	
Salvinia *sp*	Cr	0.94 (mg kg^{-1} dry matter)	Espinoza-Quinones et al. (2005
	Cu	1.42 (mg kg^{-1} dry matter)	
	Zn	6.80 (mg kg^{-1} dry matter)	
Thlaspi caerulescens	Cd	263 (mg kg^{-1})	Lombi et al. (2001)

Pb, Lead; Cd, Cadmium; Ni, Nickel; Cu, Copper; Zn, Zinc; Cr, Chromium.

Table 8. Worked carried out heavy metal remediation through plants in different states of India.

State	Plant species	Metal	Results	Reference
Delhi	*Lemna minor*	Ni	3500 mg/kg	Kaur et al. (2008)
		Cd	4 mg/l	
		Cu	9 mg/l	
Uttarakhand	*T. natans*	Fe	12 mg/l	Kumar and Chopra (2016)
		Ni	3 mg/l	
		Pb	4 mg/l	
		Zn	11 mg/l	
Madhay Paradesh	*E. crassipes; J. Americana; A. philoxeroides; T. latifolia*	Cu	1; 3;1; 7 (kg/ha/year)	Archana Dixit et al. (2011)
		Zn	4; 30; 6; 6 (kg/ha/year)	
		Mn	300; 13; 27; 79 (kg/ha/year)	
		Fe	19; 120; 45; 23 (kg/ha/year)	
Maharashtra	*E. crassipes; Azolla*	Cu	0.013; 0.006 gm/l	Shekhar and Prashik (2016)
		Cr	0.071; 0.0625 gm/l	

Table 8. Cont.

Odisha	*E. crassipes*	Cd Cu	0.56 mg/l 0.48 mg/l	Swain et al. (2014)
Mizoram	*Spirodelapolyrhiza*	Cd Ni Pb	4.5 mg/g 3.4 mg/g 3 mg/g	Prabhat kumarai and Tripathi (2011)
Assam	*Hydrilla verticillata*	Cr Cd	2 mg/g 3 mg/g	Phukan et al. (2015)
Jammu&Kashmir	*Azolla pinnata*	Cu;Pb;Cr;Cd; Zn	88;335;0.2;0.03;2.04 ppm	Shafi et al. (2015)
Gujarat	*E. colonum* *E. crassipes* *H. verticillata* *I. aquatic* *N. nucifera* *T. angustata* *V. spiralis*	Cd;Co;Cu;Ni; Pb;Zn	0.56;8.16;113;11;4;2; 53 0.79;25;44;28;9;709 0.15;5;16;5;7;457 24;24;54;14;5;328 .35;7;1600;5;8;423 2;14;104;20;6;276 0.83;3.09;27;4;82;377	Kumar et al. (2008)
Uttar Pradesh	*B.monnieri* *E.crassipes* *H.verticillata* *I.aquatica* *M.minuta*	Cr;Ni;Cu;Pb	94;4;7;5, ug/g 46;26;8;4.5 µg/g 25;3;4;7; µg/g 25;3.5;3;6.5 µg/g 10;6;3;3.5 µg/g	Narendra et al. (2012)
Kerala	*Eichhornia* sp. *Pistia* sp. *Salvinia* sp.	Cu;Fe;Pb	172; 137.5; 107.5 mg/l 27.5; 17.5; 12.5 mg/l 12.5; 17.5; 6.5 mg/l	Preetha and Kaladevi (2014)
West Bangal	*Typha sp* *Pistia* sp. *Salvinia* sp. *Eichhornia* sp.	Pb; As; Cu;Cd	1.1;0.045;0.187;0.074 mg/g 0.47;0.032;0.135;0.052 mg/g 0.47;.018;0.155;0.038 mg/g 4;0.029;0.115;0.039 mg/g	Sukumaran (2013)
Karnataka	*E. crassipes*	Pb Cu	1.069 mg/l 1.488 mg/l	Seema et al. (2013)
Meghalaya	*S. mucronatus* *R. rotundifolia*	Cd	7000 µg/g 5000 µg/g	Marbaniang and Chaturvedi (2014)

Cd; Cadmium; Ni, Nickel; Cd, Cadmium; Zn, Zinc; Cr, Chromium; Cu, Copper; As, Arsenic; Fe, Iron; Mn, Manganese; Co, Cobalt.

worldwide and research laboratories are at presently engaged to deal with these limitations.

CONFLICT OF INTERESTS

The authors have not declared any conflict of interests.

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Review

Deforestation in power line construction in the Central African Region

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Power transmission lines in forest regions like the southern parts of Cameroon are subjected to numerous failures arising from natural hazards, including earth faults and line ruptures provoked by swaying or falling neighbouring trees and their branches. To pre-empt this problem, those trees which represent a potential threat to the operation of the power line must be eliminated. Deforestation during line construction phase therefore becomes inevitable, and this leads in turn to the destruction of flora and fauna. The need hence arises to implement a deforestation strategy during power line construction which limits the negative impact of loss of forestry and wildlife resources on the environment to an acceptable level. In this paper a method is proposed which limits the level of destruction of vegetation and respects modern environmental standards during the construction of power lines through dense forest regions. It is shown that the required right-of-way depends on the quantity of power to be transmitted, on the voltage level chosen for the transmission, on the type of accessories used for the line construction and on the relief of the line track. Consequently, the relevant parameters for deforestation have been identified, listed and analysed. This leads to a good overview of the required deforestation level in the design and realisation of a power transmission line. The environmental impact assessment of transmission line projects can hence be better quantified and compared in aspects that relate to the protection of trees in the fight against global warming and desertification.

Key words: Power line construction, right-of-way, deforestation, environmental impact, global warming, desertification

INTRODUCTION

Demographic and economic growth have both led to a perpetual increase in electrical power demand in recent years. Since the consumption centers are generally not close to the generation sites, new power transmission lines must be constructed regularly. Due to technical and economic considerations, such power lines sometimes

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run through dense protected forest areas. Apart from the fact that dense forest vegetation renders access to the installations and maintenance work extremely difficult, recent statistics show that over 80% of supply disturbances on power transmission lines are single-phase line-to-earth faults provoked by felling of trees or sawing of tree branches (Désiré, 2004). These provoke short circuits and cause line ruptures. It therefore becomes indispensable for supply reliability to get rid of trees along the power line which represent a potential danger to it. The trade-off is of course the destruction of flora, and possibly fauna too.

Fragmentation is a severe threat to tropical rainforests. However the habitat loss and less extensive fragmentation caused by roads can also be a threat, not only through allowing access to remote areas, but also through a series of insidious associated impacts. These include abiotic and biotic edge effects adjacent to road clearings, the disturbance impacts caused by vehicle operation, invasion by weeds, feral and alien fauna and disease, and faunal mortality from vehicle collisions.

In combination, these can create a significant barrier to movements of rainforest biota. Impacts can be ameliorated through clever road design and sustainable vehicle operation (Goosem, 2007).

A compromise needs therefore to be sought that takes care of the aspects of improved reliability in power supply, while also limiting the destruction of the environment to acceptable modern standards. Using objective and quantitative methods to circumscribe the exact minimum of unavoidable deforestation is the subject of analysis of this paper.

In the first part, a survey of the complete deforestation process is done. The term *deforestation* is defined, and deforestation techniques are presented. The factors influencing these techniques are treated. The actual reasons for tree-felling are dealt with and a general formulation derived from there to serve as a rule for the establishment of a security corridor during deforestation for line construction.

An analysis of the relief of the zone allows the proper specification of the needed security corridor, even for areas with difficult landscape.

METHODS AND ORGANISATION OF THE ACTIVITY OF DEFORESTATION IN POWER LINE CONSTRUCTION

In Laurance et al. (2009), linear infrastructure such as roads, highways, power lines and gas lines are omnipresent features of human activity and are rapidly expanding in the tropics. Tropical species are especially vulnerable to such infrastructure because they include many ecological specialists that avoid even narrow (<30-m wide) clearings and forest edges, as well as other species that are susceptible to road kill, predation or hunting by humans near roads. In addition, roads have a major role in opening up forested tropical regions to destructive colonization and exploitation.

Deforestation is the act of felling of trees over a large area as could, for example, be needed for the right-of-way of a long power transmission line through the equatorial forest. This can be done in an intuitive manner on site or could be conceived to follow a pre-defined pattern. Whatever the case, both technical and environmental constraints must be given focused attention. The technical constraints include transmissible power, voltage level and the accessories used. Environmental constraints in their turn should include existing settlements, their cultures, river paths, track landscape and community construction standards. In order to undertake a systematic description of the line construction track and scientifically formulate and specify the “level of deforestation”, the following part-corridors are defined as shown in Figure 1:

A_d: Deforestation Reference Axis or Power Line Axis *(This coincides with the lengthwise centre of the transmission line).*
L_c: Central Lane *(The central lane is the corridor in which the power line will actually be positioned. The vegetation here must be reduced to less than 20 cm of height to permit the easy unraveling of conductors during their laying).*
C: Main Track Corridor *(This is the central lane plus the flanking security zones. This area should have vegetation of not more than 30 cm of height to allow for the free movement of construction equipment).*
L_d: Deforestation Limit *(The vegetation can be allowed to be more than 30 cm of height beyond this area).*
n: Out-of-corridor tree-distance to deforestation limit *(The out-of-corridor area is the one beyond the deforestation limit). Any tree within this zone that is considered dangerous for the line operation must be cut down).*
Z_s: Flanking security zones *(A security margin to be determined based on voltage level).*
L_s: Main Corridor Security Margin.
d_a: Average Distance between an out-of-corridor tree and the power line axis.
h_c: Maximum out-of-corridor tree-height.

The determination of the sizes of the various corridors defined above depends firstly on the voltage level, which prescribes the accessories to be used and is in turn chosen based on the transmissible power, and secondly on whether the line is single-phase or three-phase. Generally in Cameroon, the single-phase systems are medium-voltage lines used for primary distribution and running through rural areas, while the three-phase systems are for transmission voltages above 90 kV that are used to transport bulk power to urban areas.

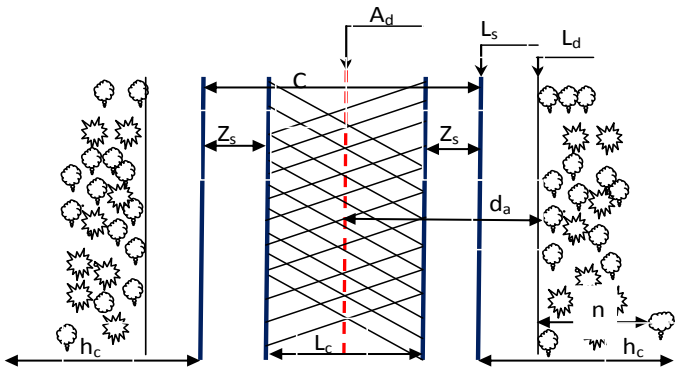


Figure 1. Areal view of the line construction path.

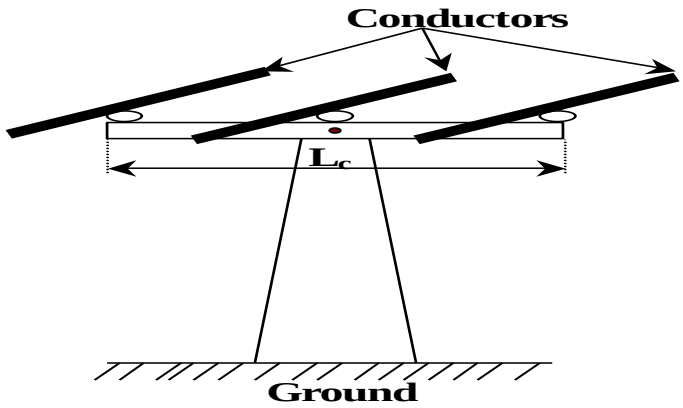


Figure 2. Horizontal conductor layout resulting in widest right-of-way.

Practical measures to reduce the negative impacts of roads and other linear infrastructure on tropical species are also highlighted (Laurance et al., 2009).

USING THE LINE REACTANCE TO DETERMINE THE CENTRAL LANE

The design of every transmission line requires that the quantity of power to be should first be specified. For short and medium-length transmission paths, the line resistance is small and often neglected compared with the line reactance. The power that can be transported over a lossless transmission line is given by the following formula (Désiré, 2004) and Lavanchy (1952):

$$P = \frac{U_1 U_2}{x} \sin\theta \quad , \qquad (1)$$

Where:

x is the reactance of the transmission line
θ is the transmission angle
U_1 is the Sending-end voltage
U_2 is the Receiving-end voltage

The maximum transmissible power is therefore inversely proportional to the reactance of the line only, when the sending-end and receiving-end voltages are fixed. Hence, the reactance of a given line can be used as a guiding parameter for the determination of the right-of-way.

Considering, for example, that the specific linear inductive resistance of an overhead electric line can be calculated using the following formula (Levêque, 1996a):

$$x_0 = 0{,}145 \log \frac{2D_m}{d} + 0{,}016\mu_r \quad , \qquad (2)$$

Where:
D_m is the average distance between conductors in mm
d is the diameter of conductor in mm
μ_r is the magnetic permeability of the conductor material

And that:
For a given conductor length and known specific linear reactance of the conductor material, the line reactance can be calculated as (Levêque, 1996b):

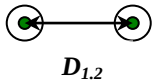
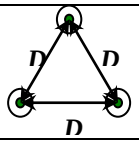
$$x = x_0 l \quad , \qquad (3)$$

Where:
x_0 is the specific linear reactance in Ω/km;
l is the length of conductor of an electric line.

Then the reactance of each power transmission line can be seen to depend on the ratio of conductor-spacing to conductor-diameter, as well as the length of the transmission line. The former determines the width of the right-of-way, while the latter is responsible for the length of the right-of-way. For a given maximum transmissible power and voltage level, the line reactance can be quickly determined. From the line length and conductor diameter d, the conductor spacing D is also determined. Hence the central lane L_c defined earlier is known. Tree-felling must be within this L_c area.

The central lane will also vary depending on the layout of the conductors. In practice, the average distance between the conductors is fixed by the type of accessories chosen and how the conductors are suspended on these accessories. The area covered by the suspension accessories is largest and leads to maximum destruction of vegetation, when the conductors are displayed horizontally as shown in Figure 2.

Table 1. Average distance between conductors (Lavanchy, 1952; AES-SONEL, 2004a).

Conductor layout	Average distance
	$D_m = D_{1.2}$
	$D_m = \sqrt[3]{D_{1.2}\, D_{1.3}\, D_{3.2}}$
	$D_m = \sqrt[3]{2}D = 1.26\, D$

Depending on how the conductors are placed in space, their average distance can be determined using the formulas in Table 1.

AIM OF DEFORESTATION AND ITS PROCEDURES

Generally in power line construction, the aim of deforestation is to secure the future energy transportation from the generation site to the consumption centers at the lowest possible cost. This entails:

i) Keeping the central lane free for movement of equipment;
ii) Protecting the central lane against any possible tree growth;
iii) Minimizing the number of times tree-cutting under the power line is carried out during operation.

Such interventions would require qualified human resources and appropriate equipment, thereby leading to increased running costs.

Deforestation parameters

The following formulas can be used to predict the total right-of-way to be created by the deforestation exercise:

$$E_d = 2x_d + 2Z_s + L_c, \quad (4)$$

$$x_d = L_d - L_s, \quad (5)$$

Hence, the entire deforestation surface can be simply calculated by multiplying the determined right-of-way by the length of the transmission line, as follows:

$$S_d = E_d L, \quad (6)$$

L is the length of the entire transmission line.

This gives a handy figure for the environmental destruction causable by the line construction and is a basis for the comparison of similar projects in their environmental impact assessment studies.

Below are parameters used in calculations to determine the choice of the way the deforestation can be carried out:

A_d is the Deforestation Reference Axis or Power Line Axis
L_c is the Central Lane
C is the Main Track Corridor
L_d is the Deforestation Limit
n is the Out-of-corridor tree-distance to deforestation limit
Z_s is the Flanking security zones
L_s is the Main Corridor Security Margin
d_a is the Average Distance between an out-of-corridor tree and the power line axis
h_c is the Maximum out-of-corridor tree-height.

Deforestation equation

After deforestation works, it can be asserted that the energy transport line is viable and secured, as the cutting of trees represents no danger anymore. This security is assured by the deforestation equation (D.E.), which is given by the following relation (Désiré, 2004; AES-SONEL, 2004a, b).

$$h_a \leq d_a - \frac{C}{2} \quad (7)$$

Where:
h_a is the dangerous out-corridor tree height
d_a is the average distance between an out-corridor tree and the power line axis
C is the main track corridor.
Considering that some out-corridor trees are grouped while others are dispersed, then the average distance between an out-corridor tree and the power line axis can be calculated as:

$$d_a = \frac{d_{ar} + d_{aep}}{2}, \quad (8)$$

d_{aep} is the average distance of dispersed trees;
d_{ar} is the average distance of grouped trees.

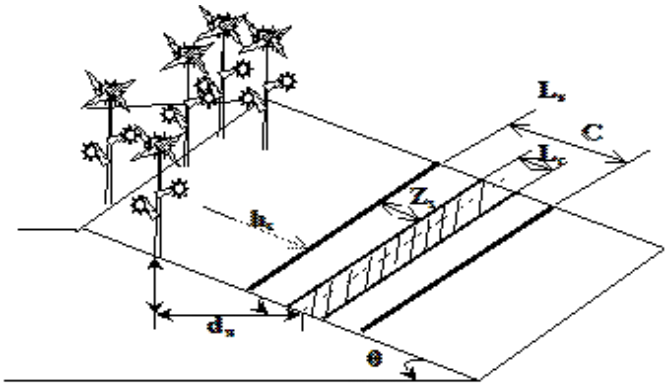


Figure 3. Areal view of the security corridor of the inclined terrain.

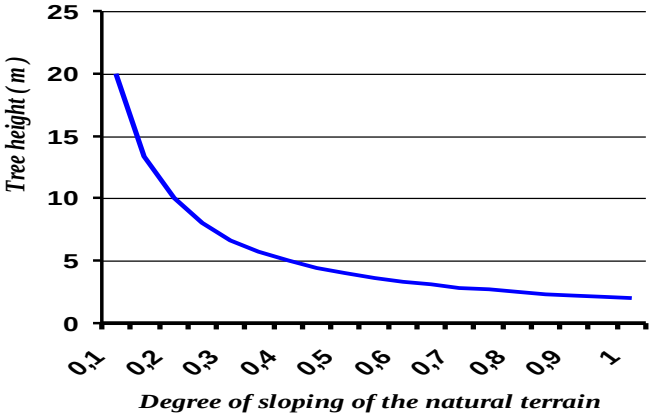


Figure 4. Dangerous out-corridor tree height of the inclined terrain.

From Equation 7, it is seen that all the maintained out-corridor trees considered not to be dangerous have a height less than the average distance of an out-corridor tree, increased by half of the main track corridor width.

Eccentricity

When the electric power transmission lines pass through an inclined terrain, it is necessary to take the eccentricity into account. Eccentricity here refers to the movement, due to gravity, of the tree stem from its stump when falling down a slope. In this case, a safety margin (***sm***) is introduced, such that ***sm*** ϵ] 0, 3], in meters.

For such an inclined terrain as shown in Figure 3, D.E. becomes:

$$h_a < \frac{1}{\cos\theta}\left(d_a - \frac{C}{2}\right), \quad (9)$$

Where **cosθ** is the angle of the inclined plane.

This formula exposes the fact that the maximum out-corridor tree height h_c is bigger for the inclined plane than for level land because of the introduction of the safety margin (sm). Also, trees at the top side of the inclined plane must be subjected to very regular controls to ensure that they do not grow to a height bigger than h_a as given in Equation 9. Such trees must be felled, if they attain the height $\mathbf{h_a}$ value given in Equation 9. Deforestation on an inclined plane is therefore more than would be necessary on normal level land.

Figure 4 shows the dependence of the maximum out-corridor tree height (h_a) on the slope of the inclined plane (cosθ). As an example, the average distance d_a between an out-corridor tree and the power line axis is taken here to be 12 m, while the corridor width C is in turn taken to be 20 m. This curve exposes qualitatively that, as the inclination of the slope of the terrain increases, that is, cosθ decreases, the dangerous out-corridor tree height also increases rapidly.

These interrelationships have been furthermore applied to the case of a 30-kV line with a main track corridor C of 40 m and a central lane of 8 m. The out-corridor in this case is unlimited. Also there are some grouped trees at a distance of 34 m while some dispersed trees are at a distance of 40 m. Applying Equation 8, the average distance between an out-corridor tree and the central axis of the power line will be 37 m. In this case, the height of an out-corridor tree has to be less than 17 m (that is, applying Equation 7). All out-corridor trees of height 17 m or greater are dangerous to the power line, and must be cut down. The value of the security zone of such a power line is therefore 16 m.

Rainforest species sometimes use regrowth connections along gullies to cross the powerline corridor. Mitigation of the fragmentation effects caused by powerline grassy swathes can best be achieved by strengthening extant canopy connections in regrowth gullies, and by establishing new connections across the clearings (Goosem and Marsh, 1997).

CONCLUSION

A direct interdependence between the right-of-way of a projected power transmission line and the maximum transmissible power has been established. This permits the design Engineer to quantify the area of forest and vegetation to be destroyed by a power transmission undertaking. Hence a tool is provided for comparative environmental impact assessment studies, which today constitute a compulsory part of multilaterally funded power line construction projects.

Furthermore, a direct relationship between the transmission voltage level and the height of out-corridor trees to be felled has been given as a mathematical

formula. This permits a quick identification and assessment of the trees in a project area that need to be felled. This study reveals in parallel that trees standing on an inclined plane need a wider security margin away from the power line so that falling trees do not temper with power system operation.

It is finally evident from the results obtained that this tool helps to reduce the destroyable forest area, since forest destruction no longer needs to be arbitrary and disorderly.

In agreement with the work of Laurance et al. (2009), tropical forests mainly exist in developing nations like those in the Central African Region, which are being transformed by ongoing industrialization, population growth and natural-resource exploitation. Electricity transmission lines, roads and other linear infrastructure are rapidly expanding in many of these countries and have a key role in opening up forests to hunting, illegal mining, land speculation and destructive exploitation. The problem is usually aggravated by the kind of attention given by the governments and other stake holders, to the environmental impact assessments conducted before many huge electrification projects.

CONFLICT OF INTERESTS

The authors have not declared any conflict of interests.

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Review

Bioremediation of soil and water polluted by cyanide: A review

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Cyanide is a chemical that is widely distributed in the environment, mainly as a result of anthropogenic activities. Only small quantities are naturally produced. Most industrial activities use this chemical compound for manufacturing a product as electroplating or for extracting gold. Exposure to cyanide results in negative health impacts to the wildlife and humans. In nature, cyanide occurs in several species and fates, of which the free cyanide forms are the most toxic ones. Cyanide can be removed by chemical or biological processes. Biological treatment called bioremediation, which is cost-effective and eco-friendly, is the most applied process to remove cyanide from contaminated environments. This technology focused on the use of microorganisms to remove pollutants. Many microorganisms have been reported to transform the cyanide in another less toxic compound, or to consume cyanide for their growth. The reactions are influenced by environmental parameters such as pH and temperature and by the nutriment availability. Bioremediation technologies were few applied in most of African Countries. Future works should focus on how to adapt the bioremediation technologies that already applied in other parts of the World in African context.

Key words: Biotreatment, chemical compound, environment, microorganism.

INTRODUCTION

Historically, cyanide has been used as chemical weapon in First World (Gupta et al., 2010). Actually, many industrial company and gold mining extraction sector use this compound. The leachate that released into the environment contains often residual cyanide. Cyanide is a chemical product that is universally recognized as

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poison (Morocco, 2005). It is highly toxic to living organisms (World Health Organization, 2004). Many researches on the toxicity and the removal of cyanide have been published in the literature. Different technologies are available for cyanide removal, such as, alkaline chlorination or biological oxidation processes (Patil and Paknikar, 2000; Young and Jordan, 1995) and acidification and/or destruction by chemical oxidation (Akcil, 2003; Young and Jordan, 1995). Among these technologies, the most used one is chemical oxidation (Botz, 2001; Dubey and Holmes, 1995; Parga et al., 2003; Roshan et al., 2009; Young and Jordan, 1995). But, the chemical oxidation processes are more expensive and could to produce other hazard compounds (Sharma, 2012; Vidali, 2001). Therefore, most of researches focused on the biological processes for destroying the cyanide compounds (Akinpelu et al., 2015; Fernandez et al., 2004; Huertas et al., 2010; Mekuto et al., 2013; Yamasaki et al., 2002; Wu et al., 2014). Microorganisms and/or plants could be transform cyanide on compounds less toxic. Cyanide is converted to carbon and nitrogen source by various enzymes present in microorganism (Ibrahim et al., 2015). The several researchers have been reviewed a bioremediation tests in different scale and conditions (Baxter and Cummings, 2006; Dash et al., 2009; Luque-Almagro et al., 2016). But, few of them show the applicability of the bioremediation technologies in the African countries. The objective of this paper is to thinking about the technology most adapted in Africa for remediating the natural environment polluted by cyanide, in which the sources, uses, behaviour and toxicity of cyanide are describe, and then different biodegradation test of cyanide by microorganism and plant and their efficiency are shown with the recent bioremediation technologies application in the world

CYANIDE

The term cyanide refers to all of the cyanide compounds that can be determined as the cyanide ion (CN^-) (Donato et al. 2007; Franson 1992). Cyanide is a group of compounds which contains a C≡N group: one atom of carbon linked with one atom of nitrogen by three molecular bounds. Cyanide compounds are usually categorized into 3 groups: the first group called free cyanide is related to the cyanide ion CN- (produced by the dissolution of sodium or potassium cyanide in water) and the hydrogen cyanide gas (HCN); the second group is related to weak and moderately strong complexes formed between cyanide ion and some metals such as Zn, Ni, Ag, Cd, Hg; the third group is related to strong complexes formed between cyanide ion and Fe ion (Botz et al. 2005; Nsimba 2009). Other forms of cyanide include cyanates and nitriles. Cyanide is produced by both natural and anthropogenic processes.

NATURAL PROCESSES

Cyanide is produced naturally in the environment by various bacteria, algae, fungi and numerous species of plants including beans, fruits, vegetables and roots. Today, cyanogenic compounds can be found in more than 3000 species of plants, animals, microbes and fungi (Stevens and Strobel, 1968; Ward and Lebeau, 1962). Many common plants contain the natural form of cyanide, cyanic glucoside (Aazam, 2014). Several plants produce cyanides, however in most cases; cyanide is present in extremely small quantities. Plants produce cyanide as a defence mechanism against herbivores (Jones, 1998; Nsimba, 2009; Randviir and Banks, 2015)

Incomplete combustion during forest fires is believed to be a major environmental source of cyanide, and incomplete combustion of substances containing nylon produce cyanide through depolymerization (Li et al., 2000).

Anthropogenic processes

Significant quantities of cyanide is a by-product of various industrial processes, including coal coking, coal gasification and steel manufacturing as well as petroleum refining (Nsimba, 2009). Cyanide also originates from metal finishing, ore extraction, and hydrometallurgical industries (Aazam, 2014). The principal anthropogenic forms of cyanide are hydrogen cyanide (HCN), cyanogen sodium (NaCN) and cyanogen potassium (KCN). Anthropogenic inputs of cyanide into the environment are greater in quantity than natural inputs (Nsimba, 2009). The process of degassing coal produces a raw gas containing hydrogen sulphide (H2S) and HCN. At US gas (work) sites it is typical to use 8 to 21 kg of gas purification material per 1000 m3 of gas produced (Theis et al., 1994; Kjeldsen, 1999). The spent iron ore contains high quantities of sulphide (typically 40 to 50%) and substantial quantities of cyanide (typically 1 to 2% by weight) (Kjeldsen, 1999; Theis et al., 1994; Young and Theis, 1991). During the electroplating process, the degreasing bath contains potassium or sodium cyanide and sodium hydroxide (Kjeldsen, 1999; Mohler, 1969). In gold mine extraction, tailing ponds containing gold mine wastes are sources of cyanide contamination (Alesii and Fuller, 1976; Boucabeille et al., 1994; Kjeldsen, 1999; Thompson and Gerteis, 1990).

INDUSTRIAL USE

Cyanide is used by humans in many cases. Every year,

in industry, massive quantities of cyanides are used in metal extraction, electroplating, pesticides, metal hardening, photography, printing, dyeing, and many other manufacturing processes. It is also used in the production of organic chemicals such as nitrile, nylon, and acrylic plastics (Aazam, 2014).

The use of cyanide also facilitates the storage of salt. Potassium ferrocyanide (K4Fe(II)(CN)6) and sodium ferrocyanide (Na4Fe(II)(CN)6) in maximum concentrations of 200 mg kg-1 have been used as anti-clumping additives in road salt in order to facilitate handling and distribution (Ohno, 1990; Kjeldsen, 1999).

Gold mining use

Cyanide is also used in the chemical extraction of gold from low-grade ores by the heap leach process (Kjeldsen, 1999; White and Markwiese, 1994). This is the predominant process in the gold extraction industry that has been applied commercially since 1887 (Adams and Lloyd, 2008). Gold occurs in multi-phase mineralization including base-metal-rich mesothermal and base-metal-poor epithermal assemblages; it is recovered by a combination of amalgamation with mercury and leaching with cyanide (Carling et al., 2013).

The first recorded use of cyanide to extract gold from ores was in 1889 at the Crown Mine in New Zealand (Dorr, 1936; Johnson, 2014). A typical cyanide heap leach process has a series of steps (White and Markwiese, 1994). The gold-bearing ore is extracted, crushed to a nominal size, and piled on a constructed liner. An alkaline cyanide solution is sprayed on the pile. The cyanide solution is buffered to a pH of about 11 to keep cyanide from forming hydrogen cyanide, which can be lost through volatilization. The cyanide in solution complexes with gold (and other metals) extracting the metals from the ore. The cyanide solution is captured by the liner and is recirculated through the pile to further extract cyanide complexes metals (Kjeldsen, 1999). The stoichiometry by which this reaction occurs is known as the Elsner (1846) equation (Jeffrey, 2000):

$$4Au + 8CN^- + O_2 + 2H_2O \rightarrow 4Au(CN)_2^- + 4OH^-$$

The fundamental process of gold extraction involves the formation of $Au(CN)_2^-$, a stable ion (Sadler, 1990) which brings gold into solution form and then be recovered by adsorption onto carbon. This aqueous ion is exploited by the gold industry most commonly through carbon in leach or carbon in pulp processes (Earls et al., 1995; Environment Australia, 1998; Minerals Council of Australia, 1996; Souren, 2000; Staunton and Jones, 1989). The current accepted mechanism describing gold dissolution in cyanide is shown in its simplest form by this equation (Donato et al., 2007):

$$4Au + 4NaCN + O_2 + 2H_2O \rightarrow 2NaAu(CN)_2 + 2NaOH + H_2O_2$$

After leaching, the crushed ore is either left on the pad or removed to another area for disposal (Kjeldsen, 1999). Therefore, artisanal gold mining is a substantial source of mercury and cyanide contamination in developing countries worldwide (Appleton et al. 1999; Carling et al. 2013; Cordy et al. 2011; Taylor et al., 2005).

War use

Another use of cyanide is for war. Cyanide is a likely weapon for terrorists due to its notoriety, lethality, and availability. Battlefield use of cyanides was proposed by Napoleon III during the Franco-Prussian war, to improve the lethality of bayonets. The French introduced gaseous HCN to World War I in 1915, and used 4000 tons in battle (Morocco, 2005). HCN gas was used in the gas chambers in the World War II holocaust, in prison for the execution of criminals with death sentences, and also as a chemical warfare agent (Nsimba, 2009).

CYANIDE BEHAVIOUR

In general, cyanide has three forms: CN_{free}, CN_{WAD} and CN_{SAD} (Botz et al., 2005). Free cyanide is the CN^- and HCN. Silver (Ag), Cadmium (Cd), Copper (Cu), Mercury (Hg), Nickel (Ni) and Zinc (Zn) are some examples of metals that form CN_{WAD} and while iron (Fe) forms C NSAD (Botz et al., 2005; Nsimba 2009). In nature, these forms of cyanide are present depending on the region.

In air

In air, cyanide is present mainly as gaseous hydrogen cyanide. A small amount of cyanide in air is present as fine dust particles. This dust eventually settles over land and water. Rain and snow help remove cyanide particles from air. The gaseous hydrogen cyanide is not easily removed from the air by settling, rain, or snow. The half-life (the time needed for half of the material to be removed) of hydrogen cyanide in the atmosphere is about 1 to 3 years. Most cyanide in surface water will form hydrogen cyanide and evaporate. However, the amount of hydrogen cyanide formed is generally not enough to be harmful to humans (U.S. Agency for Toxic Substances and Disease Registry (ATSDR) 2006).

In water

Cyanide in water

In water, there are various forms of cyanide such as:

CNfree (examples: CN- and HCN), CNWAD (examples: KCN, NaCN, cyanogen calcium (CaCN), cyanogen chloride (CNCl), cyanogen bromide (CNBr)) (ATSDR 1997; Xie and Hwang, 2000), CNSAD, cyanate (CNO-) and thiocyanate (SCN-).

In most natural waters, HCN is predominate (Doudoroff, 1976; Flynn and Haslem, 1995; Moran, 1998), but it is transported by rainwater in negligible quantities because of its solubility in water (ATSDR, 1997).

CNCl can be formed in drinking water during the interaction between humic substances and chloramines, occuring during chlorination of water (ATSDR, 1997; Boening and Chew, 1999). The half-life of cyanide in water is not known. Cyanide in water does not build up in the bodies of fish (ATSDR, 2006).

Physical properties

Cyanide in water takes gaseous and aqueous forms (Nsimba, 2009). Three gaseous forms of cyanide are present in water: HCN, CNCl, and CNBr (Nsimba, 2009; Xie and Hwang, 2000).

Thiocyanates are often present in the form of ferricomplexes, which can give a red coloration to water at concentrations of approximately 5 mg L-1 (Environmental Resources, 1987; Kjeldsen, 1999).

Chemical properties and reactivity

In water, free cyanide takes the form CN^- and aqueous HCN (HCN(aq)). Carbon (C) is associated with nitrogen (N) by triple bond.

The CN^- is a versatile ligand that reacts with many metal cations to form metal-cyanide complexes. These species, which are typically anionic, have a general formula $\mathrm{M(CN)}_x$, where M is a metal and x is the number of cyano-groups, dependant on the valence number of M (Donato et al., 2007).

Dissociation

HCN (aq) is a weak acid and can dissociate into cyanide ion according to the following dissociation reaction (Sharpe, 1976; Smith and Mudder, 1991):

$$\mathrm{HCN(aq)} \leftrightarrow \mathrm{H}^+ + CN^- \quad \mathrm{pKa} = 9.24 \text{ at } 25°\mathrm{C}$$

The distribution of HCN and CN- species is a function of pH for a simple aqueous solution at 25°C. At pH 9.2; CN- and HCN are in equilibrium. At pH 11; over 99% of the cyanide remains in solution as CN-. At pH 7; over 99% of the cyanide exists as HCN (Nsimba, 2009). The stability of metal-cyanide complexes is variable and requires moderate to highly acidic pH conditions in order to dissociate. Weak metal-cyanide complexes dissociate under mildly acidic conditions (pH 4 to 6). Strong metal-cyanide complexes require strong acidic conditions pH < 2 to dissociate (Nsimba, 2009).Depending on the type of metal, some simple cyanides can dissolve in water, forming metal ions and cyanide ions (Donato et al., 2007; Klenk et al., 1996):

$$\mathrm{M}_y(CN)_x \leftrightarrow M_y^+ + x(CN)^-$$

Dissolution

The gaseous forms of cyanide are soluble in water, but degrade by hydrolysis very rapidly at high pH. The hydrolysis degradation product is the cyanate ion (CNO^-), which can subsequently hydrolyze to carbon dioxide (CO2) and ammonia (NH3) at alkaline pH conditions (CDC, 2005; IPCS/INCHEM 2005; Nsimba 2009). The solubility of metal-cyanide complexes is influenced by pH and temperature (Botz et al., 1995) and the presence of ammonia (Donato et al., 2007; Franson, 1992).

Thiocyanates are frequently quite soluble in water and can produce relatively high SCN^- concentrations in groundwater. The potassium- and sodium ferrocyanides (which are added to road salt to prevent clumping) are readily soluble, but can re-precipitate in the presence of iron (III) (Kjeldsen, 1999).

Photo decomposition

If groundwater contaminated with ferrocyanide is exposed to daylight, iron-complexed cyanides may be transformed to free cyanide. Therefore, samples for cyanide analysis (total or weak acid dissociable cyanide) must be stored in dark containers (Kjeldsen, 1999).

Hydrolytic reactions

Cyanate hydrolyzes rapidly under acidic conditions to carbon dioxide and ammonia (FMC, 2005 in Nsimba, 2009)

$$CNO^- + 2H_2\mathrm{O} \rightarrow CO_2 + NH_3 + OH^-$$

HCN is formed in solutions of cyanide by hydrolytic reaction of CN^- in water (Doudoroff, 1976; Flynn and Haslem, 1995; Moran, 1998; Nsimba, 2009):

$$CN^- + H_2\mathrm{O} \rightarrow \mathrm{HCN} + OH^-$$

HCN is miscible with water, giving a weak acid. The CN

triple bond is readily hydrolyzed by strong alkali or acid giving formic acid and ammonia, with higher temperature favoring these reactions (Donato et al., 2007).

Hydrolytic reactions are catalyzed by some enzymes, such as cyanide hydratase, nitrile hydratase, cyanidase and nitrilase (Dash et al., 2008)

Oxidation

Free cyanide can be oxidized to form cyanate (CNO^-) or, depending on the pH, its protonated form fulminic acid (HOCN) (pKa=3.45 at 25°C) (Bard et al., 1985; Nsimba, 2009).

$$CN^- + H_2O \rightarrow CNO^- + 2H^+ + 2e^-$$
$$HCN + H_2O \rightarrow HCNO + 2H^+ + 2e^-$$

Volatilization

HCN is quite volatile with a vapor pressure of 84 000 Pa at 20°C. Henry's constant, (KH) is given as 1.7×10^{-3} (mg HCN (L air)$^{-1}$)$^{-1}$ (Gmelius Handbuch, 1977). Therefore if the concentration is 100 mg HCN L^{-1} in the pore water, and all the cyanide is assumed to be dissociated, then the equilibrium concentration in the gas phase in the pores will be 0.17 mg HCN (L air)$^{-1}$(Kjeldsen, 1999).

$$HCN\,(aq) \leftrightarrow HCN(g) \qquad LogK° = -2.8$$

Sorption

The only significant cyanide sorption observed was for iron cyanides at low pH-values, where the total cyanide concentration in the water phase is low and thus insignificant (Kjeldsen, 1999).

Substitution/transfer

Free cyanide can react with various forms sulfur ($S_xS^{2-}, S_2O_3^{2-}$) to form thiocyanate, which is relatively less toxic than free cyanide. The reactions of polysulfide and thiosulfate with the CN^- are as follows (Dash et al., 2009; Luthy and Bruce, 1979; Smith and Mudder, 1991; Zagury et al., 2004):

$$S_xS^{2-} + CN^- \rightarrow S_{x-1}S^{2-} + SCN^-$$
$$S_2O_3^{2-} + CN^- \rightarrow SO_2^{3-} + SCN^-$$

In soil and groundwater

Cyanide in soil and groundwater

The prevalent form of cyanide occurring in soils and groundwater are the following (Alesii, 1976; ATSDR, 1997; Shefchek et al., 1995):

1. HCN, also called cyanogen, hydrocyanic acid and prussic acid.
2. Simple cyanides (inorganic salts, for example NaCN, KCN).
3. Iron-complexed cyanides: Ferrocyanide ($Fe(II)(CN)_6^{3-}$), also called hexacyano- ferrate(II) and Ferricyanide ($Fe(III)(CN)_6^{4-}$), also called hexacyanoferrate(III).

An additional group is the so called nitriles, organic material with a R-CN composition where R refers to the organic radical. Various pesticides containing cyanides, as well as naturally occurring nitriles, originating from plants and microorganisms fall into this group (Fuller, 1984). A related group of compounds to the cyanides is the thiocyanates containing the (SCN) group (Kjeldsen, 1999).

Physical properties

The iron-complexed cyanides are often found as a Prussian blue colouration in the soil (ferri ferrocyanide $Fe(III)_4(Fe(II)(CN)_6)^{3-}$) and other similar iron-complexed cyanides. Cyanide concentrations of 100 to 500 mgCN kg^{-1} can produce bluish/greenish coloration in soils (Environmental Resources Ltd, 1987). Bog iron ore can also contain precipitated ferricyanides (Meeussen et al., 1990). Spent bog iron ore seems to contain very little free cyanide (Kjeldsen, 1999) SCN- are often present in the form of ferricomplexes, which can give a red coloration to soil. The red coloration is visible at concentrations of 50 mg kg^{-1} in soil (Environmental Resources Ltd. 1987; Kjeldsen, 1999)

In soils, cyanides are moderately mobile. The mobility depends on the pH condition and the percentage of iron oxides and the clay. They have a low mobility when pH is low and the percentage of iron oxides and the clay is high (ATSDR, 1997).

Chemical properties and reactivity

The behavior of cyanide compounds in soil and groundwater is governed by many interacting chemical and microbial processes (precipitation, dissolution, sorption and degradation). Therefore, cyanides in soil and groundwater have an extremely complex chemical behavior (Kjeldsen, 1999).

Dissolution and precipitation

If pH in soils is high, dissolution of iron cyanide can

result. And then, iron Fe (III) and $Fe(II)(CN)_6^{3-}$) are reform and they could be to raise to soil surface by capillary. On the surface, water is eliminated by evaporation and the ion could to precipitate, resulting in novel iron cyanide complexes (Bureau et al.,2011). Therefore, the rate of dissolution is affected by (1) buffer capacity of the soil, (2) transport of water through soils, and (3) alkalinity of water being transported through the soil (Kjeldsen, 1999; Meeussen et al., 1995).

Adsorption

Cyanide adsorption onto soils is dependent on cyanide speciation and soil mineralogy. Natural soils contain a mixture of many minerals varying greatly in physical characteristics. Aluminum oxides are ubiquitous soil components and important adsorbents in soil systems. Gibbsite ($Al(OH)_3(s)$) is the most common form of aluminum oxide found in soils (Sposito, 1984).

Sorption

Sorption of cyanide is limited, because most soil particles are also negatively charged. This limited sorption has been confirmed by column experiments with various cyanide compounds (Alesii, 1976; Zhang and Hendrix, 1991).

Retardation of cyanide due to sorption processes is generally of minor importance in most soils (Kjeldsen, 1999).

Solubility

The solubility of iron cyanide in soil is dependent on pH. Thus, reprecipitation of cyanide can occur if conditions change. Most near surface soils contain various iron hydroxides, which dissolve according to the following reaction (assuming that iron is in the $Fe(OH)_3$ form) (Kjeldsen, 1999):

$$Fe(OH)_3 + 3H^+ \leftrightarrow Fe^{3+} + 3H_2O \quad \text{where} \quad \log K_s = 3.54$$

Transformation/mineralization

Biological oxidation and ultra-violet (UV) irradiation can be transformed CN_{SAD} or CN_{WAD} and SCN^- into NH_3, sulphate (SO_4), CO_2, metal (Fe2+, Fe3+, Na+,…) (Kjeldsen, 1999).

Complexation

This was done to obtain iron-cyanides like ferricyanide (Fe III) and ferrocyanide (Fe II) (Kjeldsen, 1999)

In artisanal small scale gold mining catchment area

In general, three cyanide forms are observed in gold mining process waste solutions: CN_{free}, CN_{SAD} and CN_{WAD}. These forms are shown in Table 1. However, other compounds derived from cyanide are present such as cyanate (CNO^-), CNCl and SCN^-(Souren, 2000; Donato et al., 2007). The main WAD cyanide complexes in mining tailings waste are $Cu(CN)_3^{1-}$, $Zn(CN)_4^{2-}$, $Ni(CN)_4^{2-}$ and $Fe(CN)_6^{3-}$ (Botz et al., 1995; Ou and Zaidi, 1995; Donato et al., 2008). Iron cyanides present in the metallurgical process are $Fe(II)(CN)_6^{3-}$ and $Fe(III)(CN)_6^{4-}$ (Donato et al., 2007; Franson, 1992; Klenk et al., 1996).

Transports and fate

In artisanal and small scale gold mining, cyanide can exist in tailings ponds, surface soil, surface water, atmosphere, soil and groundwater.

Water bearing cyanide in tailings pond is in contact with tailings solids and sediment, and the atmosphere. The fate and transport of aqueous cyanide is represented by a cyanide cycle encompassing a complex set of chemical reactions involving free cyanide radical and various metal-cyanide complexes. Biodegradation of free and WAD cyanide dominates in the tailings sediment, while metal-cyanide complexation, precipitation, biological oxidation (BO), photolysis, and volatilization are the primary fate mechanisms operating in the tailings pond water. Figure 1 shows that cyanide recycling occurs between the tailings pond water and the atmosphere (A → B →C → A). For example, a portion of free cyanide in the water column (A) can volatilize as gaseous HCN (B) into the atmosphere where it can be adsorbed into moisture droplets (C) and re-enter the ailings pound via precipitation. The other portion can be biologically degraded to ammonia, carbone dioxide, and nitrate (Donato et al., 2007; Smith and Mudder, 1991).

Cyanide can be transported into surface water. Both biological and physicochemical processes control the fate of cyanide in this zone. These include: Transport processes (advection and dispersion), photolysis (dissociation of iron/other metal-cyanid complexes to free cyanide via UV irradiation), volatilization of free cyanide into the atmosphere, microbiological degradation of free cyanide and weakly-complexed cyanide, plant uptake and assimilation of free and metal-complexed cyanide, rhizosphere mediated degradation of cyanide species, adsorption of free and metal-cyanide complexes onto sediments, precipitation/dissolution of iron and other metal-cyanide complexes (Bushey, 2003; Donalto et al., 2007).

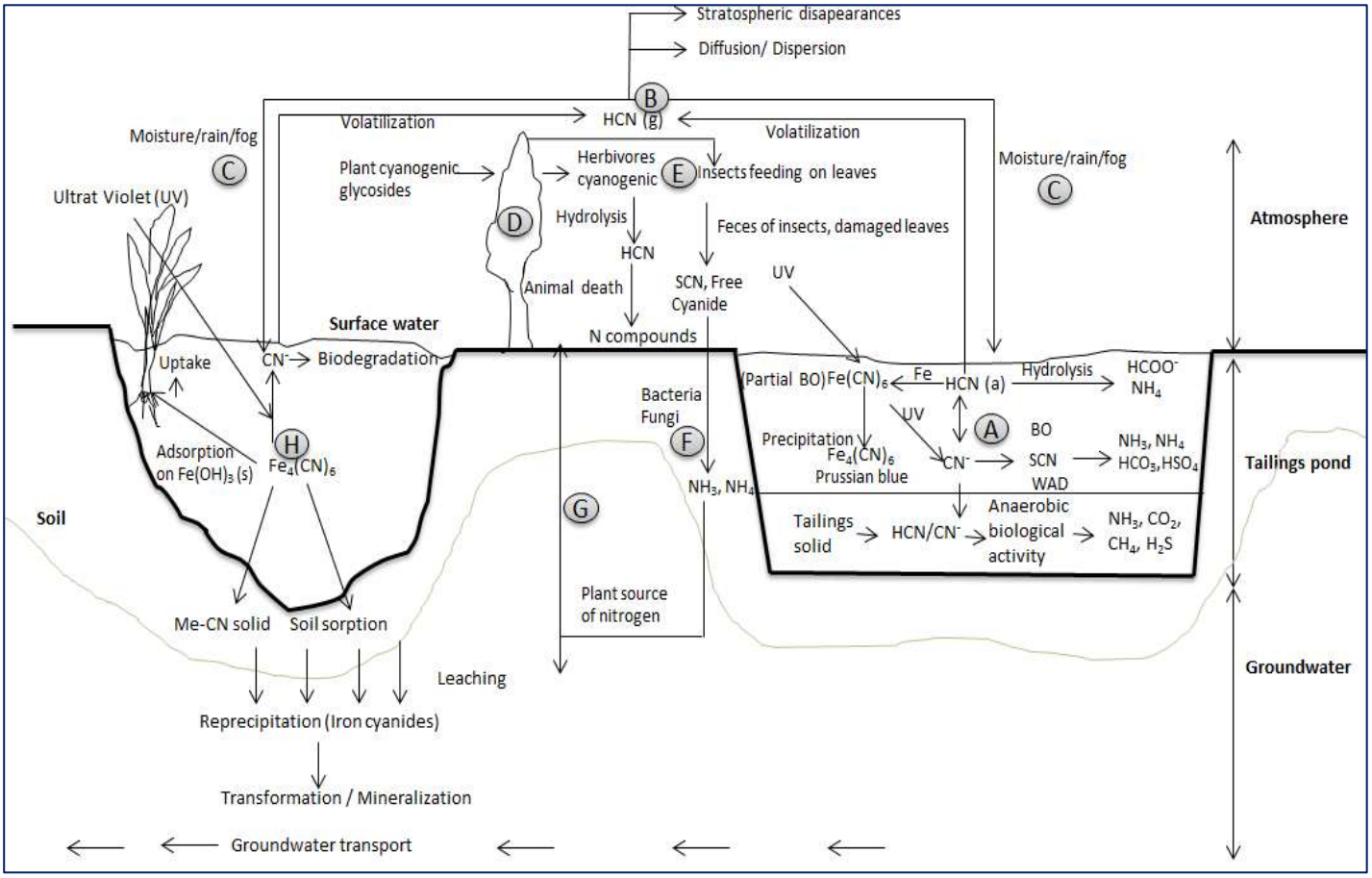

Figure 1. The different processes governing the behaviour of cyanide in nature and in a mine tailing pond (Bushey, 2003; Kjeldsen, 1999; Smith and Mudder, 1991).

Similar to the tailings pond, cyanide recycling occurring between the surface water and the atmosphere can be represented by:

H → B →C → H.

At the soil surface, cyanide can be present, if it was often land disposed on site during past mining exploitation. Leachates from this disposed material may contain iron-cyanide complexes and small amounts of other weak acid dissociable complexes depending on geochemical conditions. These soluble cyanide species can be taken up by plants or can be transported along with the groundwater as shown in Figure 1. Furthermore, cyanide in the plants can be transformed by animal metabolism (herbivores, insects). Residuals of these animals (feces, rest of animal death) and damaged leaves can be metabolized by soil microorganisms and taken up again by plants. Therefore, there is a cyanide recycling process between soil surface and groundwater following (Donato et al., 2007):

D →E → F → G →D

Physicochemical properties and reactivity

Cyanide, because of its high dielectric constant (Klenk et al., 1996), complexes readily in metallurgical circuits to dissolve other metals, preferentially gold from ores (Donato et al., 2007; Smith and Mudder, 1991; Staunton and Jones, 1989)..

Cyanide allows gold to be dissolved at a range of pH levels, although the stability of cyanide is greater at a higher pH. For gold mining metallurgical purposes, elevated pH levels of 9.5 to 10 maintain an appropriate cyanide-stability for gold dissolution (Donato et al., 2007; Souren, 2000).

Free cyanide is not persistent in the tailings environment (Botz et al., 1995; Resource Assessment Commission, 1991) and will degrade through physical, chemical and biological processes, into other less toxic chemicals (Environment Australia, 2003). Natural

Table 1. Cyanide compounds commonly present in gold mining tailings waste (Donato et al., 2007; Environment Australia, 1998; Klenk et al., 1996; Minerals Council of Australia, 1996; Sadler, 1990; Smith and Mudder, 1991; Souren, 2000).

Cyanide species	Examples
Free cyanide	CN^- , HCN
Weakly to moderately bound metal complexes (Weak-acid-dissociable) (WAD)	Zinc $Zn(CN)_4^{2-}$; Copper $Cu(CN)_2^{2-}$, $Cu(CN)_3^{2-}$; Including free cyanide
Strongly bound metal complexes (Strong-acid-dissociable) (SAD)	$Fe(CN)_6^{4-}$, $Fe(CN)_6^{3-}$

degradation, primarily by volatilization of cyanide in tailing storage facilities, is the most common method of removing cyanide in the gold mining waste (Botz et al., 1995; Donato et al., 2007; Environment Australia, 1998; John, 1988; Longe and Devries, 1988; Minerals Council of Australia, 1996; Ou and Zaidi, 1995; Sadler, 1990; Simovic et al., 1985; Smith and Mudder, 1995; Souren, 2000). A shallow pond with a large surface area increases the rate of conversion to gaseous HCN from the liquid phase (Environment Australia, 2003), although such spatial features attract wildlife (Donato, 1999). Aeration and mixing have a similar effect (Environment Australia, 2003). Once the pH of a tailings dam drops below 8 (Ou and Zaidi, 1995) or 9.2 (Staunton and Jones, 1989), the majority of cyanide is liberated via the gaseous state into the atmosphere (Donato et al., 2007). Unlike CN_{free}, WAD forms of cyanide, such as the copper complex, are resilient in the tailings dam environment (Brooks and McGlynn, 1987; Donato et al., 2007; Minerals Council of Australia, 1996; Sinclair et al., 1997; Staunton and Jones, 1989) and subsequently release cyanide ions under varying environmental conditions (Botz et al., 1995; Donato, 1999; Henny et al., 1994) and absorption by wildlife (Donato et al., 2007; Reece, 1997; Ryan and Shanks, 1996).

In addition the specific chemical profiles of metal cyanide complexes are important. Copper and zinc cyanide complexes are insoluble in water (Franson, 1992), but soluble in ammonia (Klenk et al., 1996).

Heavy-metal cyanide complexes are characterized by their stability, insolubility or being slightly soluble (Donato et al., 2007; Franson, 1992; Klenk et al., 1996).

The concentrations of bioavailable cyanide complexes to wildlife in the tailing environment depends strongly on the type and valence of the complex metal ion, as well as temperature, the pH of the solution, the pH of wildlife digestive tracts, dissociation rates and the concentration of the solution (Donato et al., 2007; Hallock ,1990; Klenk et al. 1996; Smith and Mudder, 1995).

TOXICITY

In nature, various forms of cyanide are present depending on the environment. The most toxic form is free cyanide.

Humans and the environment are highly affected by cyanide. Cyanide is the most significant contaminant that affects wildlife mortality (Donato et al., 2007; Henny et al., 1994). The most important exposure routes to humans are: ingestion and dermal contact, inhalation of volatilized cyanide, and groundwater exposure (Donato et al., 2007; Henny et al., 1994; Kjeldsen, 1999; Minerals Council of Australia, 1996; Ryan and Shanks, 1996; Wiemeyer et al., 1985).

Other potential effects can occur on terrestrial species (plants and animals) and on surface water species (by recharge of cyanide containing groundwater to surface waters) (Donato et al., 2007; Henny et al., 1994; Kjeldsen, 1999). Hydrogen cyanide and other cyano-compounds that liberate free cyanide ions are highly toxic to almost all forms of fauna (Souren, 2000; Kumar et al., 2017). The toxicity is related to the inverse of the bond strength of metal atoms and cyanide ligands (Klenk et al., 1996; Sadler, 1990; Staunton and Jones, 1989).

Many researchers have reported the lethal toxicity of several cyanide complexes to birds (Barcroft, 1931; Davis, 1981; Eisler, 1991a, b; Reece, 1997). Lethal limits varied between species (Table 2) (Donato et al., 2007)

Humans

Human can be exposed to cyanides by breathing air and drinking water, touching soil or water containing cyanide, or eating foods that contain cyanide (ATSDR, 2006), with potentially lethal results (Table 3).

PHYSICAL AND CHEMICAL TREATMENTS OF CYANIDE

Cyanide could be removed by physical, chemical or biological treatments. Natural cyanide attenuation is also possible.

The physical and chemical treatments of cyanide operate on the principle of converting cyanide into a

Table 2. Effects of free cyanide on some birds and other animals (Ballantyne, 1987; Bapat and Abhyankar, 1984; Christel and Eyer, 1977; Donato et al., 2007; Eisler, 1991a, b; Hagelstein 1997).

Species	Dose	Comment
Mallard Duck	0.53 mg CN kg^{-1} Bird Weight (BW);	No deaths
	1.43 mg CN kg^{-1} BW	Lethal dose (LD) 50 (C.I at 95% 2.2 to 3.2)
Turkey Vulture	36 mg NaCN kg^{-1} BW	Average time of death was 19 min
Rock Dove	1.6 mg CN kg^{-1} BW	Minimum LD
Black Vulture	2.54 mg CN kg^{-1} BW	Acute oral LD50
	3.7 mg CN kg^{-1} BW	All dead within 16 min
Japanese Quail	4.5 mg CN kg^{-1} BW	Acute oral LD50 for adult females
American Kestrel	2.12 mg CN kg^{-1} BW	Acute oral LD50
Domestic Chicken	11.1 mg CN kg^{-1} BW	Acute oral LD50
European starling	9.0 mg CN kg^{-1} BW	Acute oral LD50
Cattle	200 mg HCN kg^{-1} BW	Lethal
Dog	24 mg NaCN kg^{-1} BW	Lethal single dose
Mouse	8.5 mg CN kg^{-1} BW	LD 50 lethal single dose
Rat	5.1-5.7 mg NaCN kg^{-1} BW	LD 50 lethal single dose

Table 3. Lethal dose of cyanide for human depending on exposure way.

Exposure way	Lethal dose	Sources
Inhalation	200 - 314 mg HCN m^{-3}	Chaumont and Weil, 1960 ; Yacoub et al., 1974
Ingestion	0.56 - 1.52 mg CN kg^{-1}	Shete and Kapdnis, 2012; United State Environmental Protection Agency, 1987
Dermal contact	100 mg CN kg^{-1}	Rieders, 1971

less toxic compound through an oxidation reaction. Several destruction processes are well proven to produce treated solutions or slurries with low levels of cyanide as well as many metals: alkaline chlorination process (Botz 2001b; Dash et al., 2009; Dubey and Holmes, 1995; Parga et al., 2003; Young and Jordan, 1995), sulfur dioxide and air process, copper-catalyzed hydrogen peroxide process, Caro‘s acid process, the iron-cyanide precipitation, activated carbon polishing, ion exchange, reverse osmosis, ozonation, etc. (Ackil, 2003).

Most of these methods are expensive and have several disadvantages (Wild et al., 1994). For example, alkaline chlorination process is not effective in the case of cyanide species complexed with metals such as nickel, silver, etc. due to slow reaction rates (Patil and Paknikar, 2000). The process also produces sludge, which requires specific license for disposal. Another disadvantage is that it is relatively expensive due to the quantity of chlorine required. Further reason, the addition of excess chlorine increases the total solids content of water, making it undesirable for recycling and reuses purposes and leaves a residue with a high chlorine content which is toxic to aquatic life (Kao et al., 2003, 2006). In addition, various chlorinated organics may be produced if the wastewater contains organic substances (Dash et al., 2009).

Natural cyanide attenuation

It is well reported that cyanide solutions placed in ponds or tailings impoundments undergo natural attenuation reactions, which result in the decrease of the cyanide concentration. These attenuation reactions are dominated by natural volatilization of hydrogen cyanide, but other reactions such as biological degradation, oxidation, hydrolysis, photolysis and precipitation also occur (Botz et al., 2005). At several sites, ponds or tailings impoundments are intentionally designed to maximize the rate of cyanide attenuation. Advantages of natural attenuation include lower capital and operating costs when compared to chemical-oxidation processes (Ackil, 2003)

Bioremediation of entity polluted by cyanide

Cyanide is a chemical compound that microorganism or plants can transform to another compound less toxic.

Usually, microorganisms or plants are used for remediating environments polluted by cyanide. Bioremediation refers to the use of microorganism (Elkins, 2013) and phytoremediation refers to the use of plants. Biological methods are preferred for cyanide removal because of their low operation cost, their ability to remove a wide range of cyanide compounds, and their ability to produce high quality effluents (Botz et al., 2005).

Biodegradation mechanism

There are many groups of microorganism discovered which can transform simple or complex cyanide compounds, including bacteria such as *Klebsiella oxytoca* (Chen et al., 2008), *Pseudomonas fluorescens* P70 (Dursun et al., 1999), fungus such as *Fusarium solani* (Barclay et al., 1998), *Fusarium oxysporum* (Akinpelu et al., 2015) and algae such as *Scenedesmus obliquus* (Gurbuz et al., 2009).

Cyanide is used as a nutrient by the bacteria for their growth, acting as nitrogen source. Some bacteria are able to use cyanide compounds as both a carbon and nitrogen source. Therefore, supply of external carbon source is no longer needed for these bacteria. Other bacteria need glucose as carbon source for survival in presence of cyanide (Bouari, 2012; Dursun et al., 1999).

The biodegradation occurred into two steps:

The first step is the oxidative breakdown of cyanides, and subsequent sorption and precipitation of free metals into the biofilm. Cyanide and thiocyanate are then converted to ammonia, carbonate and sulfate (Ackil, 2003):

$$CN^- + \frac{1}{2}O_2 + 2H_2O \rightarrow HCO_3^- + NH_3$$

$$SCN^- + 2O_2 + 3H_2O \rightarrow SO_4^- + NH_4^+ + HCO_3^- + H^+$$

$$M(CN)_x^{y-x} + x/2O_2 + 3xH_2O \rightarrow M^{y+} + xNH_4^+ + xHCO_3^- + H^+$$

In the second step, ammonia is converted to nitrate through the conventional two step nitrification process shown as:

$$NH_4^+ + \frac{3}{2}O_2 \rightarrow NO_2^- + H^+ + H_2O$$

$$NO_2^{2-} + \frac{1}{2}O_2 \rightarrow NO_3^{3-}$$

The ease of degradation of metal cyanides depends on their chemical stability: Free cyanide is the most readily degradable, followed by metal cyanide complexes of Zn, Ni, and Cu; iron cyanide the least degradable (Mudder et al., 1998).

Bioremediation capacity

focused on: Correlation of the growth kinetics of the bacteria and the rate of cyanide removed, evaluation of the environmental parameters on the degradation of different cyanide compounds or determination of the minimum inhibitory concentration (MIC) cyanide compounds for the microorganism. Tables 4 and 5 show a comparison of studies evaluating cyanide biological transformation, respectively in water and in soil

In Tables 4 and 5, the same bacteria were used for biological treatment of cyanide but the potential effectiveness varies depending on the composition of the medium, type and initial concentration of cyanide and organic matter, pH, and temperature.

In the water, the optimal condition is formed by a pH ranging from 5.2 to 10.5 for bacteria, from 6 to 8.5 for fungus and pH 12 for plant. Temperature is usually held between 25 and 50°C, with the majority around 30°C for bacteria, 43°C for fungus and 40°C for plants. Cyanide hydratase is often used by microorganism as enzyme for degrading cyanide. *Pseudomonas Fluorescens* NCIMB 11764 was identified as the potential cyanide degrading bacteria, its performance is between 105 and 706 µmol min^{-1} for degrading free cyanide with or without enzyme as catalyzer (Kunz et al. 1992; Kunz et al., 1998). The mixed bacteria composed by *Klebsiella pneumoniae* and *Ralstonia* sp. have also a high potential with a velocity 1042 µg L^{-1} min^{-1} for degrading thiocyanate (Chaudhari and Kodam 2010). For Fungus, *Gloeocercospora sorghi* is the most effective. Its maximal velocity is 4.4 mmol min^{-1} mg^{-1} (Basile, 2008; Jandhyala, 2002).

In the soil, the optimal condition was formed by a pH around 7 for bacteria and pH 4 for fungus. The cyanide degrading bacteria have a temperature between 30 and 37°C. But, for fungus, it stays around 30°C. For degrading cyanide, bacteria and fungus use various enzymes as: Thiocyanate hydrolase, cyanidase and hydratase amidase. Microorganisms have a faculty to degrade strong acid dissociable cyanide in the soil than in the water. *Pseudomonas putida.* was identified as the potential cyanide degrading bacteria (Bipinraj et al., 2003). The fungus*, F. oxysporum* N-10 is the most effective with a velocity 0.02 and 1 mM day^{-1} respectively in the mixed (Barclay et al., 1998) and single culture (Yanase et al., 2000).

Glucose was often used as organic matter in water and soil and the final product of biodegradation is formed by ammonia or ammonium. While earlier studies only focused on the microorganism application, after the discovery of co-culture bacteria or fungus, many current studies are focused on the addition of agricultural wastes or wastes in the microorganism mixed culture.

Research about phytoremediation is not investigating deeply.

Bioremediation technologies

More choice of bioremediation technologies were existed

Table 4. Comparison of potential cyanide bioremediation in water.

Microorganism	Enzyme	Origin	Compound to be removed	Optimum condition	Degradation efficiency	Final product
Bacteria						
Thiobacillus intermedius (Singleton and Smith 1988)	Rhodanese	Salt swamp Salt water	CN^- / 50 mM	Salt swamp : pH 8.1 Salt water : T 25°C	0.021 µmol min^{-1} (without enzyme) 0.042 µmol min^{-1} (with enzyme) 0.015 µmol min^{-1}	SCN^- Sulfite (SO_3^{2-})
a. Klebsiella sp. *b. Klebsiella pneumoniae* *c. Pseudomonas putida* (Silva-avalos et al., 1990)	Nhase	Creek water	Tetracyanonickelate (II){K2[Ni(CN)4] (TCN) KCN	0.25 - 16 mM TCN 0.25 mM KCN / T 41 °C		$Ni(CN)_2$
Pseudomonas fluorescens NCIMB 11764 (Kunz et al., 1992)			KCN/50 mM	20 - 50 mM pH 7 / T 31°C	85%/6 h (aerobic condition) 89%/12 h (anaerobic condition)	Formamide($HCNOH_2$) or formate ($HCOO^-$)
P. fluorescens NCIMB 11764 (Kunz et al., 1998)	Cyanide oxygenase Keto-acid 23 mM		HCN	pH 7 T 30°C	760 µmol min^{-1} ml^{-1} (after 72 h) (without acid)	NH_3
P. fluorescens NCIMB 11764 (Fernandez et al*.,* 2004)	cyanide oxygenase		KCN	10 - 50 µmol T 30 °C	90 - 100%	$HCOO^-$
a. *Neurospora crassa,* *b. Gibberella zeae,* *c. Aspergillus nidulans* (Basile 2008)	Cyanide hydratase	Waste water	KCN 20 mM Metal-cyanide complexes	pH: 5.2 - 9 (a) / 6 - 8.5 (b) 7,5 (c) / 6 - 7 (a), (b), (c) T 27- 43 °C	< 80% (a), (b), (c) (after 48 hours)	
Thiobacillus thioparus THI115 (Yamasaki et al., 2002)	thiocyanate hydrolase	Lake water	SCN^-	T 30°C	93% (in 38 h)	Carbonyl sulfide (COS)
Bacillus sp. (*Bacillus safensis, Bacillus lichenformis, and Bacillus tequilensis* (Mekuto et al., 2013)			Cyanide compounds	T 37°C	65.5% (200 mg CN L^{-1}) 44.3% (400 mg CN L^{-1})	
Micromonospora braunna (Shete and Kapdnis, 2012)	Cyanide hydratase	Garden soil	KCN (N source) : 10-1000 ppm Dextrose (C source)	T 30°C (aerobic condition)	98.79% (pour 100 ppm in 18 h)	HCOOH NH_3
B. safensis + *B. lichenformis* + *Bacillus tequilensis* (Mekuto et al., 2013)		Wastewater	KCN: 200 and 400 mg CN L^{-1}	T 37°C	65.5% (over 8 days) for 200 mg CN L^{-1} 44.3% (over 8 days) for 400 mg CN L^{-1}	

Table 4. Cont.

B. safensis + B. lichenformis + B. tequilensis + Agrowaste (Ananas comosus extract: 1% v/v, Beta vulgaris extract:1% v/v, Ipomea batatas extract: 1% v/v, spent brewer's yeast: 1% v/v, and whey: 0.5% w/v) (Mekuto et al., 2013)		Wastewater	KCN: 200 and 400 mg CN L^{-1}	T 37°C pH = 9.5	89.5 % (over 8 days) for 200 mg CN L^{-1} 59.75 % (over 8 days) for 400 mg CN L^{-1}	
Burkholderia cepacia C-3(Adjei and Ohta, 2000)			Free cyanide 260 mg L^{-1} Fructose	T 30°C pH =10	80%	
Pseudomonas sp. (Kao et al., 2003)			Free cyanide 100 mg L^{-1} Lactate, sucrose	T 28-30°C pH 9-9.2	60%	
Pseudomonas sp. (Akcil et al., 2003)			Free cyanide 400 mg L^{-1} Whey	T 30°C pH 9-9.2	89%	
Klebsiella oxytoca (Kao et al., 2003)			Free cyanide 21 mg L^{-1} Glucose	T 30°C pH 7	99.9%	
Trametes versicolor			Free cyanide 400 mg L^{-1} Citrate	T 30°C pH 10.5	30% (Cabu et al., 2006) 100 % (after 42 h)(Akinpelu et al., 2015)	Ammonium Nitrogen (NH^{4+}-N)
Klebsiella oxytoca (Chen et al., 2008)	Nitrogenase		Free cyanide 157 mg L^{-1} Glucose	T 30°C pH 7	26%	
Pseudomonas pseudoalcaligenes CECT5344(Huertas et al., 2010)			Free cyanide 40 mg L^{-1} Acetate	T 30°C pH 9.5-10	99.9%	
Klebsiella pneumoniae + Ralstonia sp.(Chaudhari and Kodam, 2010)	Thiocyanate hydrolase	Wastewater (Effluent industrial sites)	KSCN (Thiocyanate)	T 37ºC pH 6.0	500 - 2500 mg L^{-1} day^{-1}	H_2S
Bacillus sp. CN-22 (Wu et al., 2014)	Cyanide dihydratase	Wastewater (Electroplating effluent)	HCN 700 mg L^{-1}	T 31°C pH, 10.3	200 - 6.62 mg L^{-1} 72 h^{-1}	HCOOH NH_3
Bacteria + cassava peels (Siller and Winter, 1998)		Wastewater	KCN	T 25-37°C pH 6-7.5	400 mg L^{-1} day^{-1}	HCOO- (formate) NH_3
Enterobacter sakazakii (a) *Azotabacter sp* (b) *Rhizobium sp* (c) (Ninan et al., 2013)		Wastewater (effluent de Sago)	KCN	MIC 5000 ppm (a) MIC 50 ppm (b), (c)	99% (after 96 h)	

Table 4. Cont.

Pseudomonas fluorescens + *Chlorella vulgaris.* (Kiruthika, 2008)			Cyanide 0.5 mg + glucose 1 g (a) Cyanide 0.5 mg + glucose 1 g + NaCl 1 g (b) Cyanide 1 mg + glucose 1 g + NaCl 1 g (c) Cyanide 1 mg + glucose 1 g + NaCl 1g (d)	T 30°C pH 7.2 (a), (b) T 30°C pH 8.5 (c), (d)	60% (a) 58% (b) 54% (c) 51% (d)	
Fungus						
Fusarium solani (Barclay et al., 1998)	Cyanide hydratase		KCN 80 mM		Km: 4.7 mM Vmax : 1.7 microM min^{-1} mg^{-1},	
Gloeocercospora sorghi (Jandhyala, 2002)	Cyanide hydratase		KCN 30 mM		Km: 90 mM Vmax: 4.4 mmol min^{-1} mg^{-1}	
Gloeocercospora sorghi (Basile, 2008)	Cyanide hydratase	Wastewater	KCN 20 mM Metal-cyanide complexes	pH: 6 – 8.5 / T: 27- 43°C	< 80% (after 48 h)	
Aspergillus awamori (Santos et al., 2013)	Nitrilase	Wastewater	KCN 0-475 ppm	Citrus peel , T : 45 -50˚C and pH: 4.0 to 5.5		
Fusarium oxysporum + *Beta vulgaris(Agrowaste)* (Akinpelu et al., 2015)		Gold mining wastewater	Metal cyanide + KCN 500 mg CN L^{-1}		83 - 263 mg CN L^{-1}	120- 210 mg NH4 +-N L^{-1}
Plants						
Citrus sinensis (Santos et al., 2013)		Citrus sinensis solide waste	Free cyanide (F-CN) 100 mg L^{-1} + 0.1% (w/v) of unhydrolysed *Citrus sinensis* (a) Free cyanide (F-CN) 100 mg F-CN $L-^{1}$ + 0.1% (w/v) of unhydrolysed *Citrus sinensis* (b) F-CN + heavy metals 10 mg L^{-1} (c)	T 50 °C pH 12 (a) et (b) T 40 °C pH12 (c)	17.82 % (a) 62.48 %, (b) 26.35 % (c)	

for removing cyanide. It could be conducted *in-situ* or ex-situ or by using bioreactors (Sharma, 2012). Each method has its specificity and most of them are cost-effective as shown as in Table 6, which summarize the advantages of the different technologies, and their conditions of application.

Bioremediation application in the world

Most of researches were focused on the

Table 5. Comparison of reports on cyanide biological transformation in soil.

Microorganism/plants	Enzyme	Source	Compounds to be removed	Optimum condition	Degradation efficiency	Final product
Bacteria						
a. *Pseudomonas putida* *b. Pseudomonas picketti* *c. Klebsiella pneumonia* (Silva-avalos et al., 1990)		Sewage sludge (a) Soil (b), (c)	TCN KCN	0.25 - 16 mM TCN 0.25 mM KCN T 41°C (with use of benzyl-amine for (a) ad		$Ni(CN)_2$
Thiobacillus thioparus THI115 (Yamasaki et al., 2002)	thiocyanate hydrolase	Soil	SCN^-	T 30°C	93% (in 38 h)	COS
P. putida (Bipinraj et al., 2003)		Wet soil	SCN^- 2 mM, KCN 0.2 mM, cyanocuprate (TCC) 0.5 mM, tetracyanonickelate (TCN) 0.5 mM Glucose 2 mM or Ferrous sulphide 1% or Thiosulphate 1 %	Alkali condition (4% Nacl) pH 7.5, T 30°C	With glucose : 99% TCC (in 109 cells/ml, 4 h) 92% TCN (in 109 cells/ml, 4 h) 95% KCN (in 109 cells/ml, 6 h) 96% SCN^- (in 109 cells/ml, 6 h) With ferrous sulphide : 81 % KCN (in 6 h) 91 % TC (in 9 h) With thiosulfate: 40 % TC (in 72 h)	
Alcaligenes xylosoxidans subsp (Ingvorsen and Godtfredsen, 1991)	cyanidase	soil	HCN	T 37°C	1% (in 55 h)	$HCOO^-$ NH_3
Fungus						
F. oxysporum N-10 (Yanase et al., 2000)	Hydratase Amidase	Soil	Tetracyanonickelate II (TCN) 0.5 mM and 20 mM	T 30°C	20-30 % (1 week) (for 0.5 mM TCN) 30 % (6 days) (for 20 mM TCN)	HCOOH, $HCOO^-$, NH_3
Fusarium solani + *Trichoderma polysporum* (Barclay et al., 1998)		Gasworks site soil	Tetracyanonickelate $K_2Ni(CN)_4$ 0.25 mM (a) Hexacyanoferrate $K_4Fe(CN)_6$ 0.5 mM (b) and $K_3Fe(CN)_6$ 0.5 mM (c)	pH 4	95% (b), (c): after 28 days 90% (a): after 28 days	
F. oxysporum + *Scytalidium thermophilum* + *Penicillium miczynski* (Barclay et al., 1998)		Gasworks site soil	Hexacyanoferrate $K_4Fe(CN)_6$ 0.5 mM	pH 4	32% after 28 days	

Table 6. Methods applied in bioremediation (Sharma, 2012; Shukla et al., 2010; Vidali, 2001).

Technique	Examples	Advantages	Conditions of application	References
In situ	Biosparging Bioventing Bioaugmentation	Most efficient Non Invasive Relative passive Naturally attenuated process, treat soil and water	Biodegradation abilities of indigenous microorganisms Presence of metals and inorganic compounds Environmental parameters Biodegradability of pollutants Chemical solubility Geological factors Distribution of pollutants	(Bouwer and Zehnder, 1993; Colberg and Young, 1995; Niu et al., 2009)
Ex-Situ	Land farming (Solid-phase treatment system) Composting (Anaerobic, converts solid organic wastes into humus-like material) Biopiles	Cost efficient ,Simple, Inexpensive ,self-heating Low cost Rapid reaction rate, Inexpensive, self-heating Can be done on site	Surface application, aerobic process, application of organic materials to natural soils followed by irrigation and tilling To make plants healthier good alternative to land filling or incinerating practical and convenient. Surface application, agricultural to municipal waste	(Antizar-Ladislao et al. 2007, Antizar-Ladislao et al. 2008)
Bioreactor	Slurry reactors Aqueous reactors	Rapid degradation kinetic optimized environmental parameters Enhances mass transfer Effective use of inoculants and surfactant	Bioaugmentat toxicity of amendments Toxic concentrations of contaminants	(Behkish et al., 2007)

bioremediation of water, soil and solid wastes in lab-scale. Application of bioremediation *in situ* or *ex-situ* is few limited. Figure 2 show the bioremediation applied on the entity polluted by cyanide in the World. Most of them were observed in America such as United States (Akcil and Mudder, 2003; Fernandez et al., 2004; Kunz et al., 1992; Kunz et al., 1998; Pinedo-Rivilla et al., 2009; Silva-avalos et al., 1990; Singleton et al., 1988; Wang et al., 1996), Mexico (Meehan et al., 1999), Ecuador (Diaz and Caizaguano, 1999); Europa such as France (Dumestre et al., 1997; Ferret, 2012), Germany (Ingvorsen en Godtfredsen, 1991), Switzerland (Brandl et al., 2003), Italy (Cipollone et al., 2007), Spain (Luque-Almagro et al., 2005; Luque-Almagro et al., 2008; Quesada et al., 2007), United Kingdom (Baxter and Cummings, 2006; Ezzi and Lynch, 2005), Ukraine (Podolska et al., 2003); Asia such as China (Zhou et al., 2007), India (Bipinraj et al., 2003; Shete and Kapdnis, 2012), Japan (Yamasaki et al., 2002), Iran (Mirizadeh et al., 2014; Naghavi et al., 2012), Malaysia (Maniyam et al., 2011), Taiwan (Kao et al., 2003), Thailand (Potivichayanon and Kitleartpornpairoat, 2010) and Oceania such as Australia (Markwiese and White 1991). Few Africans countries as Ghana (Adii, 2011), Nigeria (Oyedeji et al., 2013), South Africa (Akinpelu et al., 2015; Dent et al., 2009; Jandhyala et al., 2003; Mekuto et al., 2013; Ntwampe and Santos, 2013; Santos et al., 2014; Van Zyl et al., 2011) and Burkina Faso (Razanamahandry et al., 2016) have been applied bioremediation in laboratory scale. Only South Africa country had applied the cyanide bioremediation in field scale (Shumba, 2008). But, cyanide is widely used in the artisanal gold process extraction in African countries (Adii, 2011).

CONCLUSION

This review summarizes the bioremediation technologies applied for cyanide decontamination. Potentiality of bioremediation technologies depends on the existence of cyanide degrading bacteria population; the availability of cyanide as contaminant and the environment factors. Bioremediation is a natural process; it takes a little time, as an acceptable waste treatment process for contaminating material such as soil. Bioremediation also requires a very less effort and can often be carried out on

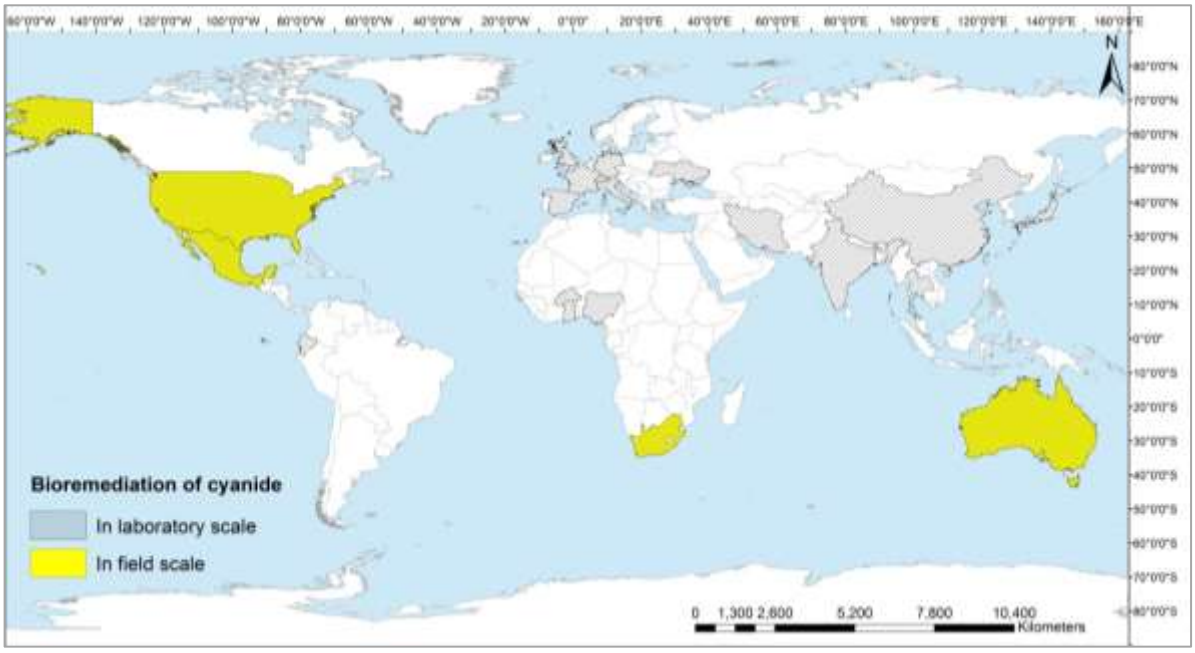

Figure 2. Bioremediation of cyanide in the world.

site. It is a cost effective process than the other conventional methods that are used for clean-up of hazardous waste and it does not use any dangerous chemicals. Bioremediation technologies could be applied in large scale and in different contaminated unit by cyanide as liquid, solid and gas industrial wastes. Nevertheless, the choice of single or mixed microorganism is very important for applying bioremediation. Field scale of cyanide bioremediation is few applied in the World, especially the African countries. Future works should be focused on how to adapt the bioremediation technologies that have already applied in other parts of the world in African context.

CONFLICT OF INTERESTS

The authors have not declared any conflict of interests.

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Seasonal macrophyte diversity and water quality in an urban wetland

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The ecosystem services of encroached urban wetlands that receive wastewater and surface run-offs have become more challenging due to climate variability. Our study assessed the seasonal macrophyte diversity and water quality of the streams flowing into and out from Pece wetland in Gulu Municipality. The macrophyte species in the wetland were sampled along five transects. Water samples from the streams were also collected and analyzed in the laboratory. Results indicated forty two macrophyte species that were in twenty four families. Macrophyte diversity and equitability were higher at the wetland edge, but were not significantly different in the entire wetland (p = 0.41, respectively). The recorded faecal coliforms (FC), total suspended solids, electrical conductivity and turbidity in the streams were higher in the wet season than the dry season. The wetland doubled the retention of FC during the dry season and relatively less retention of total suspended solids was recorded in the dry season. The urban expansion and farming might alter the macrophyte abundance and richness in Pece wetland, thus affecting the ecosystem services.

Key words: Diversity indices, faecal coliforms, physico-chemical, surface-run-offs, wastewater.

INTRODUCTION

Wetlands are not only rich in diversity but are also providers of ecosystem services such as ground water recharge, flood control and sediment filtration (Schuyt, 2005). They also provide water purification service when surface run-offs (Schuyt, 2005), raw sewage, and partly treated wastewater containing nutrients (Mugisha et al., 2007) is discharged through them. However, worldwide, the area of wetlands is decreasing, and there is increased pollution and a decline in the ecological functions of such areas (Kyambadde et al., 2004; Bassi et al., 2014). The Millennium Ecosystem Assessment (2005) has estimated that up to half of the world's wetlands are lost due to human activities. This has affected water access to millions around the world and it is becoming more challenging due to climate variability, which also threatens wetlands ecosystem services in the

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drought or flood prone areas. Globally, sanitary conditions and poor water quality have continued to be a great health risks (Horwitz and Finlayson, 2011) and are of widespread concerns (Goshu et al., 2010).

Wastewaters that flow through wetlands have improved quality (Odong et al., 2013; Kyambadde et al., 2004). Their flow in the wetlands also influences the macrophyte growth and development (Kanyiginya et al., 2010; Mugisha et al., 2007; Kansiime and Nalubega, 1999). These result into enriched diversity, and better wetlands' functioning and stability (Lan et al., 2010). Nonetheless, creating new channels and widening of existing ones result into more wastewater entering the wetlands which also affect the distribution of wastewater within the wetlands thus affecting their efficiency in wastewater treatment (Kansiime and Nalubega, 1999). Notwithstanding, Pece wetland that drains run-offs from Gulu Municipality is under pressure of indiscriminate solid wastes and wastewater disposal (Opio et al., 2011). The original macrophytes in some sections of the wetland have also been cleared to give way for urban development and farming (NURP, 1997). The changes might alter the abundance and richness of aquatic biota and there is likelihood of invasive species developing in the wetland (Trombulak and Frissell, 2000). These may also reduce the ability of the wetland to improve on the wastewater quality that flow through it.

A section of Pece wetland also contains stabilization ponds or lagoons that provide tertiary wastewater treatment and the effluent from the lagoons is discharged into the wetland (Sarah, 2014). The lagoons were intended to treat wastewater and sewage for a smaller population. Consequently, the growing human population that is connecting their sewer lines to the lagoons may compromise the efficiency. To improve on the efficiency, such lagoons in the tropical conditions are regularly de-silted to enhance wastewater retention.

In understanding the importance of urban wetlands, an investigation on the wastewater purification of Pece wetland in Gulu Municipality, and also the documentation of macrophyte richness and evenness were made. We hypothesized that there is no seasonal differences in the macrophyte diversity and also in the water quality of the streams flowing into and that flowing out from the wetland.

MATERIALS AND METHODS

Study area

Pece wetland located in Gulu District (Figure 1) has a catchment area consisting of intermediate savannah grasslands characterized by open canopy of trees that are 10 to 12 m high and underlying grasses of 80 cm tall. The area has a bimodal rainfall pattern with a short dry spell in July and one long dry season from late November to early March. The average monthly rainfall ranges between 14 mm in January to 230 mm in August.

Assessment of macrophyte diversity indices

The diversity indices offer important information about rarity and commonness of species in a community and are important means to understand plants community structure. In this study, 5 transects (T1, T2, T3, T4 and T5) (Figure 1) that were established in Pece wetland were used to assess the seasonal macrophyte abundance and equitability. The distance between transects were approximately 5 km to ensure that the composition of the plant species in both degraded and undegraded areas were captured. The wetland was also demarcated into side A (north and eastern side) and side B (south and western side) using the main water channel. Each of the 5 transects had 6 plots established at an interval of 15 m apart. Each transect had 2 m^2 plots which were used to assess the presence of grasses and herbs and 4 m^2 for shrubs and trees. Plots 1A and 1B represent plot 1 on side A and B, respectively. Therefore, plots 1A, 2A and 3A were alternate to 1B, 2B and 3B. Plant species in the plots were identified and counted after 30 days interval. These covered seasonal variations from the months of February to May, 2015.

Water quality assessment in the inflow and outflow streams of Pece wetland

Water quality assessment was done in the inflow and the outflow streams of the wetland. Water samples were taken biweekly from the inflow and outflow streams (Figure 1). Grab water samples were collected at 10 cm below the water surface of each stream between 8:00 am and 12:00 pm in clean sterilized 500 ml plastic bottles. The bottles were rinsed with the stream water before the samples were collected. The samples were immediately stored in a cool box (4°C) and then transported to Uganda National Water and Sewerage corporation laboratory, Gulu branch on the same day for analysis.

Faecal coliforms analysis

Membrane Filtration Method was used to determine numbers of FC as described by APHA (1992). 100 ml of the water samples were filtered through a membrane of pore space 0.45 nm diameter to retain all the bacteria which was then placed onto a Lauryl membrane sulphate broth pad. This was incubated at 44°C for 12 h and yellow colonies formed were counted as FC. This was expressed in colony forming units (CFU) per 100 ml of water sample.

Physico-chemical water analysis

The total suspended solids (TSS) and turbidity was measured using DR6000 spectrophotometer and 2100Q turbidimeter made by Hach Company, Loveland-Colorado USA, respectively. A blank consisting of 25 ml of deionized water was used to calibrate the spectrophotometer and turbidimeter reading to 0 mg/L. 25 ml of water samples were stirred and put in the spectrophotometer and turbidimeter and readings were taken in mg/L and NTU, respectively. Electrical conductivity (EC) was measured using HQ40D multimeter (Hach Company, Loveland-Colorado USA). The EC probe was rinsed with deionized water and readings taken in µS/cm after inserting the probe into the water samples in a beaker.

Discharge measurement

Floatation method was used to determine water flow at the

Figure 1. Location of Pece wetland in Gulu District, Uganda showing transects and water sampling points.

sampling points of each inflow and outflow streams. Pieces of paper of equal size were made to float on the water surface and their movement downstream was timed within a distance of 1 m. This was replicated three times for each sampling point and the speed of the floating paper was calculated from the distance travelled over the time taken. An average speed for each sampling

site was calculated during each visit. The depth and width of the streams channel were also measured using a tape and these were used to calculate the cross-sectional area of the individual streams. The average speed of the pieces of paper multiplied by the cross-sectional area was the water discharge for each sampling site. The load of pollutants into and out of the wetland was calculated by multiplying discharge (m^3/s) with the concentration of the pollutants.

Data analysis

Abundance for each species was calculated as:

$$\frac{\text{Sum of individual species}}{\text{Sum of all the species}} \quad (1)$$

Simpson diversity indices were used to characterize macrophyte species diversity (D) and the equitability (E) in the wetland (Equations 2 and 3, respectively).

$$D = \frac{1}{\sum_{I=1}^{S} P_I^2} \quad (2)$$

where S is the total number of species in the community (richness) and P_i is the proportion of species (S) made up of the *i*th species.

The equitability (E_D) was expressed as a proportion of the maximum value D could assume, if individual in the community were completely evenly distributed. Equitability took a value between 0 and 1, with 1 being complete evenness.

$$E_D = \frac{D}{D_{Max}} = \frac{1}{\sum_{I=1}^{S} P_I^2} X \frac{1}{S} \quad (3)$$

The FC concentration, the load and retention were presented in logarithmic scale. The retention of FC and TSS in the wetland were calculated as:

$$\text{Retention} = \left(\frac{\text{Total inflow load - Total outflow load}}{\text{Total inflow load}} \right) \times 100 \quad (4)$$

The calculation of the wetland retention was based on the assumption that there was diffused water flow within the wetland, and the processes involved in the retention of FC and TSS were the same during the different seasons. Furthermore, it was assumed that the wetland received inflows from only the nine streams and the run-offs from the catchment at the wetland edge had no influence on the hydrology within the wetland.

The MINITAB software version 17 was used for data analysis. Nonparametric Kruskal-Wallis tests were performed to verify the degree of significance of the variances in the diversity and equitability of macrophytes, and water quality variables so as to compare temporal and spatial variations in the wetland. Parametric analysis was also done for results that indicated significant difference between sites. Data were checked for normality and equal variance and those that did not conform to the conditions were transformed. Tukey multiple comparisons were performed on transformed data to determine similarities and differences between the individual sites. The symbols ≠ and = indicate significant difference and non-significant difference between the sites, respectively. For all p-values ≤ 0.05, H_0 was rejected.

RESULTS

Macrophyte diversity in Pece wetland

A total of 42 species belonging to 24 families were identified (Table 1). The most common family in the wetland was Poaceae. *Digitaria scalarum* Chiov. represented 26.4% of the species abundance while *Cyperus papyrus* L. and *Cyperus rotundus* L. exhibited 17.5 and 12.2%, respectively. The abundance of 39 plant species were below 7% and *Crassocephalum crepidioides* S. Moore., showed the lowest abundance (0.001%). *D. scalarum* was found in all transects (T1-T5), while the rest of the plants were found in particular transects. Four species (*Rottboellia cochinchinesis* (Lour.) Clayton., *Eurphobia heterophylla* L.*, Cleome gynandra* L., and *Bidens pilosa* L.) were found only during the wet season, while five others (*Physalis angulata* L.*, C. crepidiodes* L., *Amaranthus hybridus* L.*, Adenia cissampeloides* (Planch ex Benth.).Harms and *Hibiscus esculenta* L.) were found only during the dry season. *Blechnum cartilaginum* SW.*, Dichrostachys cinerea* R.Vig.*, Acacia polycantha* Wild.*, Hymenocardia acida* Tul. and *Stipa capillata* L., showed seasonal differences in their abundance.

Macrophyte equitability declined at the upstream (T1-T3) and lower diversity was recorded at the downstream section (T4-T5) of the wetland (Figure 2). The downstream section of the wetland exhibited lower and higher variability in the macrophyte equitability, respectively, an indication of uneven distribution of the macrophyte in the downstream wetland section. The upstream section of the wetland was characterized by construction and edge gardening. Overall, macrophyte diversity and equitability in the wetland were not significantly different ($p = 0.41$ respectively).

Plots 1A and B closest to the water channel had lower diversity while plots 3A and B farthest from the channel showed higher diversity indices (Figure 3). The south and the western side (Plots 2B and 3B) showed higher variability. The seasonal macrophyte diversity indices of all the plots were not significantly different. Analysis of macrophyte equitability in the plots showed higher values during the dry season except for plots 1A and B (Figure 4). These plots are closest to the main water channel in the wetland. There was no significant difference in the seasonal macrophyte equitability in the plots.

FC numbers in the streams and their loads into and out from Pece wetland

FC numbers in the inflow streams and the outflow stream ranged from $3.5\times10^3 \pm 2.8\times10^3$ to $4.4\times10^4 \pm 7.4\times10^3$ and $1.8\times10^1 \pm 0.32\times10^1$ to $5.5\times10^3 \pm 7.7\times10^2$ CFU/100 ml during the wet season and the dry season, respectively.

Table 1. Plant species abundance in the wetland.

Family	Species	T1	T2	T3	T4	T5	Sum	Overall Abundance	Seasonal abundance Dry season	Seasonal abundance Wet season
Agavaceae	*Agave sisalana* Perrine.	-	-	-	-	36	36	0.00191	0.0026	0.0026
Amaranthac eae	*Alternanthera philoxeroides* (Mart.) Griseb.	-	88	121	83	226	518	0.02742	0.0356	0.0356
	Amaranthus hybridus.	2	-	-	-	-	2	0.00011	0.0003	-
Arecaceae	*Phoenix reclinata* Jacq.	-	-	-	170	-	170	0.009	0.0114	0.0114
	Vernonia amygdalina Delile.	-	171	-	-	-	171	0.00905	0.0029	0.0029
	Bidens pilosa	-	-	2	-	-	2	0.00011	0.0002	-
	Crassocephalum crepidioides	-	1	-	-	-	1	5.29E-05	0.0002	-
Bignoniaceae	*Kigelia africana* (Lam.) Benth.	4	-	-	-	-	4	0.00021	0.0003	0.0003
Blechnaceae	*Blechnum cartilagineum*	-	61	78	229	-	368	0.01948	0.0208	0.0131
Cannaceae	*Canna paniculata* Ruiz & Pav.	-	-	-	20	-	20	0.00106	0.0011	0.0011
Cleomaceae	*Cleome gynandra*	-	4	-	-	-	4	0.00021	-	0.0003
Colchicaceae	*Gloriosa superba* L.	-	-	-	-	229	229	0.01212	0.0037	0.0037
Combretaceae	*Combretum molle* Eng. & Deils.	-	-	20	-	-	20	0.00106	0.0016	0.0016
	Terminalia glaucescens Planch. Ex Benth.	-	-	3	-	-	3	0.00016	0.0002	0.0002
Commelinacea	*Commelina benghalensis* L.	51	309	40	-	-	400	0.02117	0.037	0.037
Cyperaceae	*Cyperus papyrus*	-	1068	572	1657	-	3297	0.17451	0.226	0.226
	Cyperus rotundus.	1142	-	102	548	518	2310	0.12227	0.0976	0.0976
Euphorbiaceae	*Euphorbia heterophylla*	-	-	12	-	-	12	0.00064	-	0.001
Fabaceae	*Vigna unguiculata* (L.) Walp.	-	105	-	-	-	105	0.00556	0.0008	0.0008
	Dichrostachys cinerea	-	33	-	41	-	74	0.00392	0.0039	0.0041
	Albizia glaberrima Benth.	-	-	42	-	-	42	0.00222	0.0029	0.0029
	Acacia polyacantha.	-	-	-	-	8	8	0.00042	0.0006	0.0003
Lamiaceae	*Leucas martinicensis* R.Br.	133	-	-	-	-	133	0.00704	0.0122	0.0122
	Vitex doniana Sweet.	-	-	12	-	-	12	0.00064	0.001	0.001
Malvaceae	*Hibiscus esculentus.*	-	6	-	-	-	6	0.00032	0.001	-
Onagraceae	*Ludwigia octovalvis* (Jacq.) P.H.Raven	62	50	76	215	-	403	0.02133	0.0302	0.0302
Passifloraceae	*Adenia cissampeloides*	6	-	-	-	-	6	0.00032	0.0011	-
Phyllanthaceae	*Hymenocardia acida.*	-	-	16	-	-	16	0.00085	0.0013	0.0007
Poaceae	*Digitaria scalarum*	1667	161	673	318	2160	4979	0.26354	0.2186	0.2186
	Stipa capillata.	-	-	1102	77	-	1179	0.0624	0.0202	0.0422
	Hyparrhenia rufa Stapf.	-	-	-	-	1078	1078	0.05706	0.0765	0.0765
	Paspalum dilatatum Trin.	825	-	-	-	-	825	0.04367	0.1019	0.1019
	Cynodon dactylon (L.) Pers.	593	-	-	-	-	593	0.03139	0.0356	0.0356
	Brachiaria brizantha Stapf.	-	-	368	-	-	368	0.01948	0.0138	0.0138

Table 1. Contd.

	***Setaria sphacelata* (Schum.). Stapf & C.E.hubb.**	-	-	91	276	-	367	0.01943	0.0058	0.0058
	***Pennisetum purpureum* Schumach.**	-	-	8	175	-	183	0.00969	0.0029	0.0029
	***Sporobolus pyramidalis* P. Beauv.**	-	-	69	-	-	69	0.00365	0.005	0.005
	Rottboellia cochichinesis	-	-	-	-	54	54	0.00286	-	0.0044
Rhamnaceae	***Zizyphus abyssinica* Hochst. Ex A. Rich.**	-	-	-	-	12	12	0.00064	0.001	0.001
Sapindaceae	***Cardiospermum halicacabum* L.**	-	81	2	23	-	106	0.00561	0.0034	0.0034
Solanaceae	***Physalis angulata***	1	19	-	-	-	20	0.00106	0.0003	0.0003
Vitaceae	***Cyphostemma adenocaule* (Steud.) Desc.**	-	298	97	66	56	517	0.02737	0.0263	0.0263
	Total						18893			

The highest FC number was at S4 (Figure 5) that received run-offs and leakages from the municipal sewer lines. A recharged stream from underground (S9) showed lowest FC number. The FC numbers were not significantly different between the sampling points during the wet season ($p = 0.304$), but was significantly different during the dry season ($p = 0.000$). Multiple comparison between sites during the dry season indicated S9 = S5 ≠ S10 = S8 = S7 = S1 = S3 = S6 = S2 = S4.

The FC loads into and from the wetland ranged from $9.9\times10^4 \pm 1.7\times10^4$ to $2.5\times10^6 \pm 3.2\times10^5$ CFU S^{-1} and $3.2\times10^2 \pm 5.6\times10^2$ to $4.2\times10^5 \pm 8.1\times10^3$ CFU S^{-1} during the wet and the dry season, respectively (Figure 6). During the wet season, S10 (Outflow stream) and S5 which was characterized by less settlement, had the highest and lowest FC loads, respectively. FC load at S1, which received wastewater from commercial buildings in the municipality, was the highest during the dry season. Higher FC values were also recorded for S4 and S6 that are characterized with motor vehicle washing. The lowest FC load was at S9 a recharge stream from underground. Except for S7 that exhibited higher variability in the seasonal discharge, seasonal loads at the rest of the sites were not significantly different. The FC loads between the sites were not also significantly different during the wet and the dry season ($p = 0.124$ and 0.4, respectively).

Physico-chemical parameters in the inflows into and outflow from Pece wetland

The effluent from the stabilization ponds (S2) exhibited the highest TSS concentration in both seasons and lowest TSS was at S9 and S5 during the wet and the dry season, respectively (Table 2). Sites S2, S3, S4 and S10 showed seasonal differences in TSS concentration. TSS concentrations between the sites was not significantly different during the wet season ($p = 0.145$) but it showed significant difference during the dry season ($p = 0.007$). Multiple comparison between sites during the dry season indicated S2 ≠ S5 ≠ S7 = S8 = S4 = S1 = S6 = S9 = S3 = S10.

The TSS load ranged between 283.6 ± 44 mg S^{-1} to 26,331 ± 25,046 mg S^{-1} during the wet season and 344 ± 202 to 15,470 ± 805 mg S^{-1} during the dry season (Table 3). Seasonal TSS loads for the sites were not significantly different. There was a significant difference in TSS load between the sites during both the wet and the dry seasons ($p = 0.002$ and $p = 0.001$, respectively). Multiple comparison of TSS load showed, S10 ≠ S3 ≠ S6 ≠ S1 ≠ S9 ≠ S2 = S4 = S7 = S8 = S5 during the wet season, while S10 ≠ S9 ≠ S5 ≠ S6 = S7 = S1 = S4 = S8 = S3 = S2 during the dry season.

The sampling site (S3) that is characterized by fertilized cropping had the highest electrical conductivity during the wet season, while effluent from the stabilization ponds (S2) exhibited the highest EC during the dry season (Table 2). EC was lowest at S9, which is recharge water from the underground. Significant seasonal differences in EC showed up for S1, S3, S5, S6 and S10. Overall, there was significant difference in the EC between sites during the dry and the wet seasons ($p = 0.00$, respectively). Multiple comparison indicated S1 ≠ S9 ≠ S2 = S4 = S6 = S5 = S3 = S10 = S7 = S8 in the wet season, while S4 ≠ S1 ≠ S2 ≠ S9 ≠ S6 = S10 = S8 = S7 = S3 = S5 during the dry season.

Sites S1 that receive run-offs from the municipality and S2 that receive effluent from the lagoons, recorded the highest turbidity during the wet and the dry seasons, respectively (Table 2). Turbidity showed a significant difference between the sites during the wet and the dry seasons ($p = 0.002$ and 0.000 respectively). Multiple comparison indicated, S1 ≠ S9 ≠ S2 = S7 = S4 = S5 = S6 = S3= S8 = S10 during the wet season

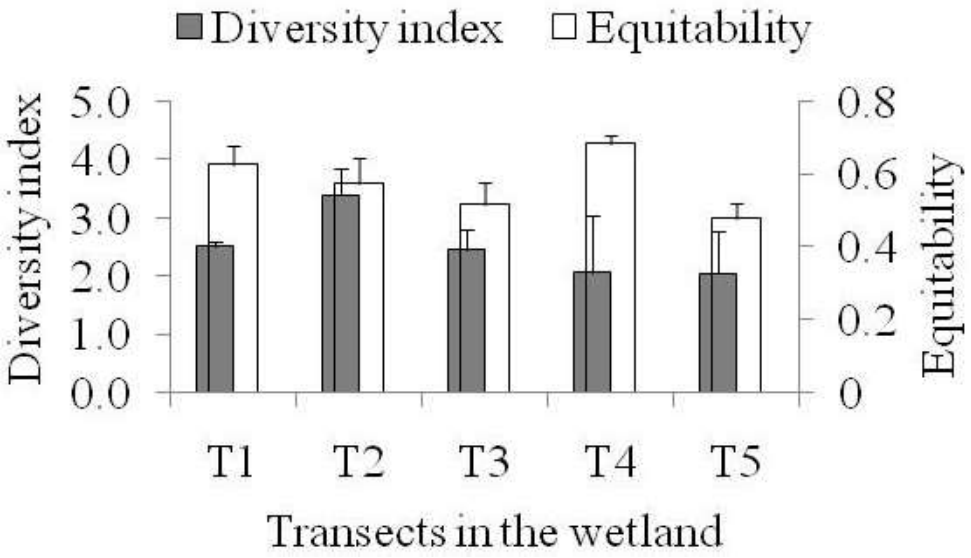

Figure 2. Mean Simpson macrophyte diversity and equitability indices in the wetland. Error bars indicate standard deviation of each transect (n = 4, for both diversity index and equitability).

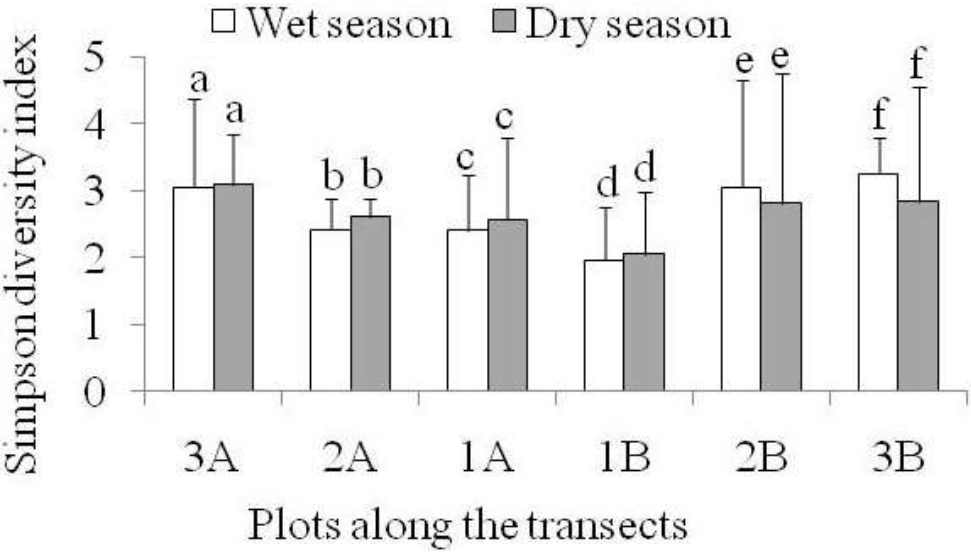

Figure 3. Mean Simpson macrophytes diversity index along plots in transects. 3A-1A and 1B-3B are plots on side A (north and eastern sides) and side B (south and western sides) respectively. Error bars indicate standard deviation (n = 10 for the wet and the dry seasons). Similar letters on the graph for each plot indicate no significant difference.

and S2 ≠ S9 ≠ S5 ≠ S8 = S6 = S1 = S3 = S4 = S7 = S10 during the dry season.

FC and TSS retention in Pece wetland

Overall, FC and TSS inflow into the wetland exhibited higher load than the outflow (Table 4). There was no significant difference in total FC load into and the outflow load from the wetland during the wet season ($p = 0.26$) but a significant difference occurred in the dry season ($p = 0.009$). TSS indicated significant difference between the total inflow load into and the outflow load from the wetland for the wet season but there was no significant difference during the dry season ($p = 0.500$ and 0.100

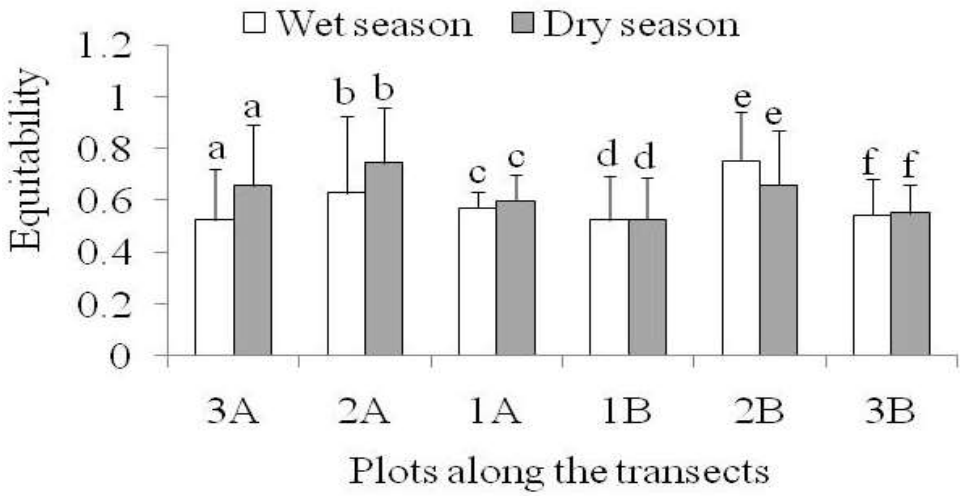

Figure 4. Mean macrophyte equitability in the plots. 3A-1A and 1B-3B are plots on the side A (north and eastern side) and side B (south and western side), respectively. Error bars indicate standard deviation (n = 10, for the wet and the dry seasons). Similar letters on the graph for each plot indicate no significant difference.

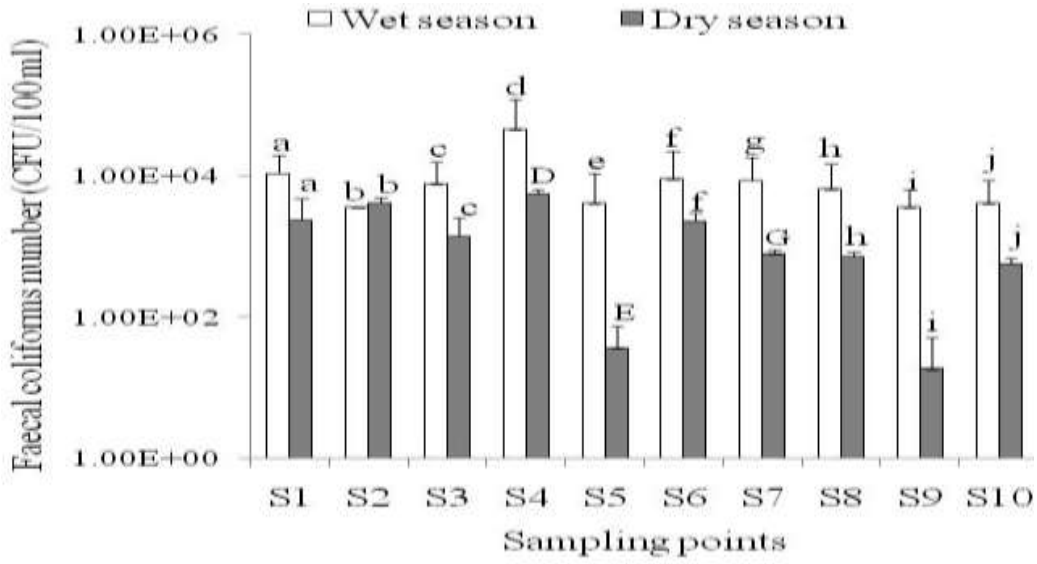

Figure 5. Faecal coliforms numbers in the inflow streams (S1-S9) and outflow stream (S10). Error bars indicate standard deviation (n = 6 for the wet season, n = 5 for the dry season) and logarithmic scale has been used. Similar letters in lower case for each site indicate no significant difference. Similar letters but in different cases for each site indicate significant difference.

respectively). There were higher retention of FC and TSS in the wetland during the dry season (Table 4).

DISCUSSION

The majority of the macrophytes that were identified in Pece wetland were grasses and herbs. Jogo and Hassan (2010) reported that in most wetlands, shrubs and trees are always harvested as fuel biomass when alternative sources are scarce. The wetland showed lower macrophyte diversity and equitability indices downstream. Lower macrophyte diversity and equitability indices were associated with lesser ionic content (Chappuis et al.,

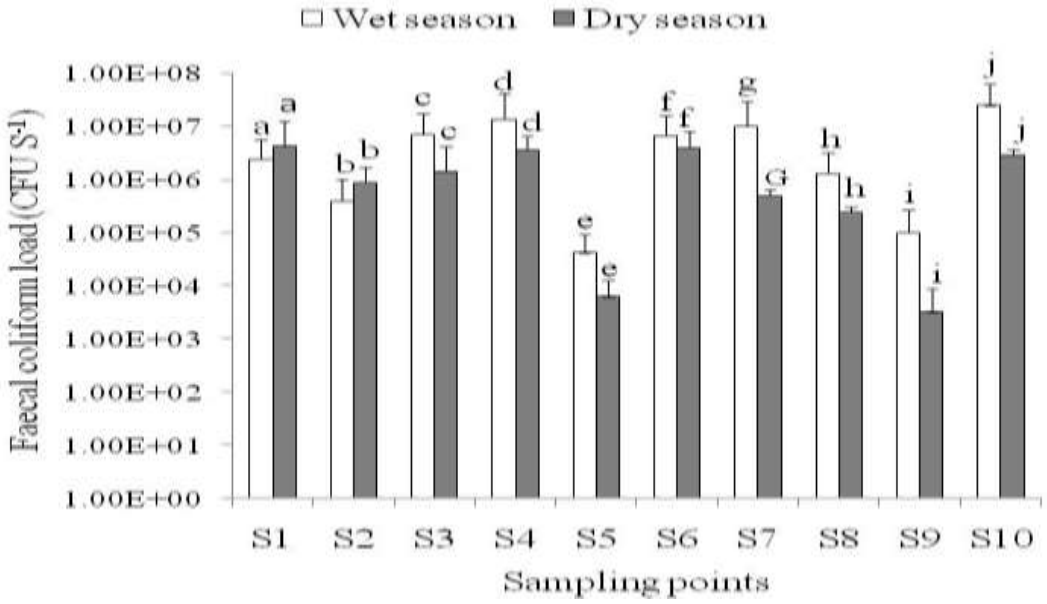

Figure 6. Faecal coliforms load in the inflow streams (S1-S9) and outflow stream (S10). Error bars indicate standard deviation (n = 6 for the wet season, n = 5 for the dry season) and logarithmic scale has been used. Similar letters in lower case for each site indicate no significant difference. Similar letters but in different cases for each site indicate significant difference.

Table 2. Seasonal physico-chemical parameters in the inflow streams (S1-S9) and outflow stream (S10) of Pece wetland (n = 6 for the wet season, n = 5 for the dry season).

Sampling point	TSS (mg/L)		EC (µS/cm)		Turbidity (NTU)	
	Wet	Dry	Wet	Dry	Wet	Dry
S1	152 ± 21.7	36 ± 15.7	334.5 ± 90.5*	224.8 ± 57*	355.7 ± 272	49.4 ± 10.4
S2	179.5 ± 0.7*	171.2 ± 66.8*	386 ± 19.8	231 ± 41	155 ± 106	117.2 ± 32.4
S3	63.6 ±15*	29.2 ± 6.5*	946.8 ± 192.8*	34.8 ± 4*	76.5 ± 21.8*	48.6 ± 9.4*
S4	76.4 ± 35.2*	41.6 ± 17*	325.9 ± 155	230.6 ± 53.8	94.7 ± 34*	44.8 ± 11.3*
S5	85.3 ± 28	20.8 ± 2.6	227.9 ± 178.6*	35.8 ± 11*	90.6 ± 30.3*	23.4 ± 4.9*
S6	90 ± 9.2	34.6 ± 10.2	181.5 ± 42.7*	72.4 ± 13*	105.8 ± 97.8	55.6 ± 13
S7	66.7 ± 63.9	44.4 ± 15	81 ± 19*	39.7 ± 12*	168.2 ± 167*	42.7 ± 9.5*
S8	57.5 ± 38.2	41.6 ± 7	68.7 ± 30	41.5 ± 13.6	94.2 ± 16	60.8 ± 25
S9	28.7 ± 3.4	33.8 ± 16.4	57.4 ± 30	30.4 ± 11	31.1 ± 29	24.7 ± 8
S10	49.7 ± 20.8*	29.2 ± 7.8*	117.2 ± 32*	51.1 ± 18.9*	63.1 ± 21*	35.1 ± 10.4*

*Means are significantly different between the seasons for each variable.

2014). However, low diversity is also associated with low disturbance (Evangelista et al., 2012; Tao et al., 2008) that allow dominant species to thrive. For example, the papyrus plant in the wetland blocks light from reaching the undergrowth species, therefore, lowering equitability. However, the effect of tremendous burning in the downstream section of the wetland during the dry season cannot be ruled out. The survival of species like *Hyparrhenia rufa* in the downstream section could be attributed to their resistance to fire or regeneration after fire.

Succession processes have also been reported as presenting advanced plant stage with different species composition (Santos and Thomaz, 2008). The reported diversity indices in the areas further away from the main water channel (towards wetland edge) is attributed to the effect of succession phenomena. On the other hand, the highest macrophyte diversity in the upstream (T2) could

Table 3. Seasonal TSS load in the inflow streams (S1-S9) and outflow stream (S10) of Pece wetland (n = 6 for the wet season, n = 5 for the dry season).

Sampling point	TSS load (mg S^{-1})	
	Wet season	Dry season
S1	9,875.3 ± 7,185	2,711 ± 2,581
S2	5,840.3 ± 265	3,371 ± 2,581
S3	4,310 ± 2,887	2,239 ± 2,115
S4	2,210 ± 880	2,626 ± 2,578
S5	432.1 ± 426	344 ± 202
S6	14,492.9 ± 8,398	6,569 ± 737
S7	5,136.1 ± 957	2,990 ± 1,772
S8	731.8 ± 500	1,437.3 ± 468
S9	283.6 ± 42	516.9 ± 269
S10	26,331.1 ± 25,046	15,470 ± 805

Table 4. Retention of FC and TSS in Pece wetland during the dry and the wet seasons (n = 6 for the wet season and n = 5 for the dry season for both variables).

Variable	Season	Inflow load	Outflow load	Retention (%)
FC (CFU S^{-1})	Wet	$4.2 \times 107 \pm 3.4 \times 10^{7}$	$2.5 \times 10^{7} \pm 3.7 \times 10^{6}$	41
	Dry	$1.5 \times 10^{7} \pm 1.6\ 8 \times 10^{6}$	$2.5 \times 10^{6} \pm 8.3 \times 10^{5}$	83
TSS (mg S^{-1})	Wet	45,696.8 ± 34,073	26,331.2 ± 18,021	42.4
	Dry	28,784.5 ± 9,707	15,469.9 ± 8,053	46.3

be ascribed to sediment accretion from the Gulu municipal run-offs and the massive wetland edge gardening. The run-offs concentrate alluvial soils at the wetland edge that provide adequate condition for subsistence farming (Tao et al., 2008). *H. esculenta* was a common agricultural plant at Pece wetland edge. Its occurrence as agricultural crop at the wetland edge has also been reported by Jogo and Hassan (2010).

The low equitability downstream was in the areas of small channel flow of about 4 to 5 m. Chappuis et al. (2014) reported association between species equitability and water body area. Therefore, hydrological gradient, such as water table depth does not only influence species distribution but also spatial distribution of vegetation across topographical gradients (Silvertown et al., 1999). The higher diversity in the dry season compared to the wet season could be due to wetland flooding in the wet season which may not favor macrophytes that are less water tolerant. The water regime during the different seasons is important in determining aquatic macrophyte species (Chappuis et al., 2014). Therefore, reduced flooding of wetlands margin results into higher abundance, richness and diversity of plants that do not tolerate high water level (Andrew et al., 2015). The higher diversity in the dry season could also be due to anthropogenic disturbances (edge gardening). Tao et al. (2008), reported higher diversity in disturbed areas, with only fugitive species surviving, and lower in undisturbed areas with only the competitive dominant species surviving.

There was highest turbidity at S1 during the wet season. Such high turbidity have been related to the increased amount of run-offs which is common in municipal areas (Löptien and Meier, 2011). Agricultural activities in and around wetlands contributing to increased turbidity has also been reported (Carrasco et al., 2013) and this explains the high values at S7 and S8. During the wet season, the high turbidity at S2 may perhaps be due to inadequate treatment of TSS in the lagoon prior to discharge. In addition, greater inflow volume into the lagoons also reduces TSS treatment performance (Borne et al., 2013). The low turbidity at S9 may be due to the fact that S9 has underground water source that is filtered by the soil particles.

Furthermore, flow velocity is an important parameter which influences TSS removal within wetlands (Yahyapour and Golshan, 2014). The wetland streams recorded a lower flow velocity during the dry season, a reason for lower load of TSS during the season. The highest TSS load at S2, a point source may be as a result

of low TSS sedimentation or high development of algal plants in the lagoons. On the other hand, the high TSS load in the outflow (S10) is the accumulation from the inflow streams and also the contribution from the detrital decomposition in the wetland.

The high EC values at S1 and S3-S6 during the wet season could be due to run-offs from municipality. But also during the dry season, the fertilized cropping of tomatoes and cabbages increase ionic content of water flowing into the wetland (Brodie and Mitchell, 2005). The cultivation in the wetland during the dry seasons enhances nutrients released at the outflow stream (S10) during the wet season. Implying mineralization (organic decomposition) that frees particulate nutrients is high during ploughing and nutrients are flushed off from the wetland during high water levels. On the other hand, high EC at S2 may be explained by the mineralization effect of organic wastes in the lagoons.

Although, FC numbers at S4 and S1 were above the standards of 10,000 FC/100 ml (ULRC, 1999) during both seasons, the retention range was 41 to 83%, which is within that reported for a tropical wetland in Tanzania (43 to 72%) (Kaseva, 2004). High retention of FC is a result of increased water retention time in the wetland which allows for adequate attachment of FC to the rhizomes of wetland plants and also detrital sedimentation of the microbes (Sundaravadivel and Vigneswaran, 2001; Kansiime and Nalubega, 1999). The rooted papyrus increased the removal of FC in the mat by providing oxygen in the rhizosphere (Kyambadde et al., 2004) which encourages the biological removal mechanism that include antibiosis, predation by protozoa, nematodes and protists (Jasper et al., 2013).

Physical factors (mechanical filtration by the rooted zone and attached biofilm, and sedimentation) and chemical factors (oxidation, exposure to biocides which may be excreted and adsorption to organic matter) could have contributed to the FC retention (Stefanakis et al., 2014). The wetland recorded high coliforms retention during the dry season. The high retention during the dry seasons is also due to the lethal effect of solar radiation (Ansa et al., 2012).

TSS retention in wetlands were reported to range from 22 to 57% for United State Kingdom (Ockenden et al., 2012), a range which falls within the TSS reduction efficiency of Pece wetland. However, Opio et al. (2002) reported 96% treatment efficiency for Kinawataka wetland in Kampala, Uganda. The differences could be due to climatic variability, and the types of macrophyte species involved in the treatment, streams discharge and the flow dynamics within the wetlands.

This study reveals that random discharge of pollutants and changes in water levels in wetlands due to human activities are able to induce great changes in the rate of aquatic vegetation community succession (Chow-Fraser et al., 1998). Sustainable use of wetland is particularly challenging in tropical environments in areas of dense human settlement (Balmford and Bond, 2005). Wetlands in Africa are being modified or reclaimed, often driven by economic and financial motives (Schuyt, 2005). Protection and restoration of wetlands are essential for future sustainability of the planet and the overall well-being of society (Clarkson et al., 2013).

However, Pece wetland still has diverse species of grasses and herbs with very few species of shrubs and trees. The equitability index in some transects and plots were above 0.5, an indication that the wetland still has species fairly distributed although some sites had FC counts and physico-chemical values that were above National Environmental Management Authority (NEMA) standard for discharge of effluent into water or on land. Therefore, the agricultural activities in the wetland should be done in line with Uganda wetland policy so that the ecological functions of the wetland are not interfered with.

CONFLICT OF INTERESTS

The authors have not declared any conflict of interests.

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Using Citizen Science Approach to monitor water, sanitation and hygiene Related Risks in Karonga Town, Malawi

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Relatively few studies have explored how resilience of water, sanitation and hygiene (WASH) systems to hazards can be enhanced under the current and future development and climatic challenges pressures in urban areas. This study employed the citizen science approach to build the capacity of citizens and integrate communities into scientific research on water quality and WASH related risk monitoring. Data was collected with assistance of 8 self-motivated and trained citizen science research counterparts. Standard sampling procedures were used to collect water samples from a total of 27 unsafe water sources in Karonga Town. The water samples were analysed for biological, physical and chemical parameters using standard methods. Personal observations were done to determine major sanitary risks impacting on a water sources in the town. It was observed that water from the majority of water samples collected from shallow wells, rivers/streams, lake and boreholes were highly contaminated with *Escherichia coli*, which were considerably higher than Malawi Bureau of Standards water quality specifications for drinking water. In general, the water is of low mineralization with rock-water interactions and surface pollution from anthropogenic activities such as agricultural activities and municipal wastes being responsible for input of biological, chemical and physical pollutants especially into the unlined and uncovered water sources. The results of the water quality index (WQI) and water quality (WQ) ratings indicated that water is not suitable for direct human consumption prior to treatment. It is recommended that onsite treatment and point of use water treatment interventions should be instituted and advocated to improve human health, livelihoods and to build resilience to WASH related risks and hazards in Karonga Town.

Key words: Citizen Science, resilience, urban risks, water quality index, water, sanitation, hygiene.

INTRODUCTION

There have been increasing worldwide scientific research interests on urban areas as critical points for climate change adaptation over recent decades. This trend has been ignited by two vital attributes of urban areas: (1) Urban areas are places that concentrate risk related to changes in climatic pressure, owing to the high

population densities, infrastructural development and investment (2) Urban areas possess significant potential in response to risks as a result of their high concentration of resources. One of the greatest risks to urban dwellers in developing countries is the unavailability of improved water, sanitation and hygiene (WASH) provision (United Nations Office for Disaster Risk Reduction (UNISDR), 2012).

Access to water and sanitation is a vital element in determination of natural hazards' social vulnerability, not only for attaining instantaneous needs, but also for the broader use of relevant disaster prevention (UNISDR, 2012). Particularly, the state of access to improved WASH provision is a global crisis, and addressing the post 2015 United Nations (UN) Sustainable Development Goals (SDGs) is critical since the 2015 Millennium Development Goal (MDG) for sanitation lagged significantly behind the other goals (Global Analysis and Assessment of Sanitation and Drinking-Water (GLAAS), 2012). According to GLAAS (2012), 83% of countries significantly lagged behind the national targets they set for sanitation.

Literature on the performance of Malawi in the achievement of the MGDs is mixed. Mamba and Gondwe (2010) and National Statistical Office (NSO) (2010) reported that out of 14 million Malawians, only 62% (95% urban and 58% rural) have access to safe drinking water and 64% (90% urban and 60% rural) have adequate improved sanitation. The Government of Malawi (2013) reported that the majority of households relied on unsafe water sources such as shallow wells and rivers domestic purposes for domestic purposes. This is the case because the majority of people in peri-urban, informal settlements and rural areas are not supplied with to piped water by utility providers (Water Boards). Nevertheless, the World Health Organization/United Nations Children's Fund Joint Monitoring Programme (WHO/UNICEF JMP) Report (2015) reported that Malawi was one of the countries in Sub-Saharan Africa that registered better progress in provision of safe and potable water by about 67%.

The majority of cities in low-income countries that experience rapid urbanization, significantly struggle to meet basic WASH needs and keep up with service provision owing to shortfalls in financing, capacity and governance. This in turn results into severe health, social and economic implications (Koppenjan and Enserink, 2008). In other words, even without considering external hazards such as droughts or floods, urban areas may be unsustainable due to internal health hazards resulting from poorly designed, implemented and maintained WASH systems (Koppenjan and Enserink, 2008). In the area of climate change, resilience emphasises the capability of a system to be dynamic and to cope with climate change, recuperate and change itself in the long term (Pasteur and McQuistan, 2016; Szoenyi, 2016).

Empirical evidence on risk and climate change adaptation is limited for African cities, Intergovernmental Panel on Climate Change (IPCC), (2014). In the African context, cities are growing rapidly, both formally and informally. At the same time, levels of inequality are high, yet strong governance and service delivery is lacking in many areas. As a consequence, the risk space is characteristically higher, and the risk burden is usually borne largely by the urban poor in African cities.

Until recently, Karonga Town, Northern Malawi has been directly affected by serious disasters like earthquake, drought and floods. These disasters affect both quantity and quality of water in addition to sanitary facilities thereby affecting WASH governance and service delivery in the Town.

The most effective approach in management of WASH systems in urban areas such as Karonga is to develop programs that incorporate a holistic approach with respect to: Prevention, protection, preparedness (through technology transfer initiatives among others), emergency response and recovery and lessons learned (returning to normal conditions as soon as possible and mitigating both the social and economic impacts on the affected population) (Oates et al., 2014).

In addition to disasters, some factors such as: Climate, topography, chemical composition of recharge water, type of minerals in aquifer matrix (water-rock interactions), evapotranspiration and impact of anthropogenic activities within catchments (Stallard and Edmond, 1983; Deutsch, 1997; Parkhurst and Appelo, 1999; Rajmohan and Elango, 2004; Appelo and Postma, 2005; Subba Rao et al., 2006; Gupta et al., 2007; Jayaprakash et al., 2008; Devadas et al., 2007) are also vital factors impacting on water sources. In the context of climate change, recent work on the resilience of water supply and sanitation was conducted by Howard and Bartram (2010), Howard et al. (2010), Calow et al. (2011), and Batchelor et al. (2010). An understanding of these underlying factors as well as the monitoring and assessment of the water, sanitation and hygiene (WASH) risks is essential in sustaining usable water supplies and building resilience under climatic pressures in Karonga Town.

Despite the apparent urgency of current and future challenges from climate change and development, few studies have explored how the resilience of WASH systems to hazards can be enhanced in Karonga Town.

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Integrating the public into scientific research is a rapidly growing phenomenon known as citizen science, whereby scientific research projects are developed with some level of public engagement (Bonney et al., 2009; Hand, 2010; Shirk et al., 2012).

Citizen Science is a form of crowdsourcing using trained volunteers to help collect data that traditionally, financially, spatially, or temporally not be feasible in large scientific research projects (Gura, 2013). Citizen scientists can provide an inexpensive, substantial, and long-term labor force (Conrad and Hilchey, 2011; Gouveia et al., 2004; Stokes et al., 1990) capable of collecting reliable and large datasets in a relatively short time over large geographical areas (Fore et al., 2001; Foster-Smith and Evans, 2003; Newman et al., 2003; Bonney et al., 2009).

There are also many perceived challenges such as the integration of data collected by citizens into the scientific process, ensuring data quality, difficulties of working with volunteers (including maintaining their engagement) and quantifying success (Bonardi et al., 2011; Kremen et al., 2011).

This study employed the citizen science approach to monitor water quality and enhance understanding of how the resilience of WASH systems to water-related hazards (e.g. floods and water scarcity) can be improved. The main aim of the assessment of risks associated with water pollution in Karonga Town project was to build community capacity in the assessment of risks associated with water pollution based on principles of the citizen science water quality monitoring approach. Through the citizen science water quality monitoring approach, data was gathered and tested by self-motivated and trained non-professionals. This was envisaged as a contribution towards the Urban Africa Risk Knowledge (Urban ARK) Research Project in Karonga Town which aims to build a public participation in monitoring and recording water quality in their town. Objectives of the study were four fold:

1. To determine nature of the sources of domestic water in Karonga Town.
2. To identify WASH related risks impacting on domestic water sources in Karonga Town.
3. To determine levels of microbial (that is, faecal coliform) and physicochemical (that is, pH, electrical conductivity (EC), total dissolved solids (TDS), chlorides, sulphates, phosphates, nitrates, total hardness (TH), total alkalinity (TA), carbonates, bicarbonates, sodium ion, potassium ion, calcium ion and magnesium ion) in water sources in Karonga Town.
4. To build resilience of WASH systems to water-related hazards (e.g. floods and water scarcity) through training counterparts on basics of the citizen science approach to monitor the water quality and assess WASH related risks in their neighborhood.

MATERIALS AND METHODS

Description of the study area

Karonga Town covers a gazetted land size of 4,386 ha and is located about 225 km north of Mzuzu City in the Northern Malawi. Karonga Town is located on a low-lying North Rukuru River flood plain (Figure 1), with altitude range of 447 to 550 m above sea level. Located by the shores of Lake Malawi, Karonga Town experiences a sub-tropical climate with two distinct seasons (that is, dry season and wet season, from June to October and November to May, respectively). Though a small city it is Malawi's fifth largest and, owing to its location on a major regional trade route to the Port of Dar es Salaam (Tanzania), it is one of the mostly rapidly urbanising towns in a country with still a very low level of urbanisation of only 20%. With, a total population of about 41,000 in 2008 and growing at growth rate of 4.3% per year with a total fertility rate of over 6.0, the population of Karonga Town is projected to reach nearly 63,000 in 2018 (NSO, 2010). In 2013 the Karonga local council extended the boundary of the city. Consequently, the population became larger than currently known.

The degradation caused by pollution arising from WASH related risks such as excess water brought about by floods (that is, the urban storm water pollution), is serious, and affects a significant proportion of the population in Karonga Town. Changes in land use that increase impervious cover lead to further flooding, erosion, habitat degradation, WASH infrastructure damage and water quality impairment. Everyday activities such as driving, maintaining vehicles and lawns, disposing of waste, and even walking pets and animals often cover impervious surfaces with a coating of various harmful materials. Construction sites, failed septic systems, illegal discharges, and improper siting and construction of sanitary facilities such as pit latrines and solid waste dump sites also contribute substantial amounts of contaminants to runoff. When these contaminants enter Lake Malawi, streams, rivers as well as groundwater sources, they result in water pollution.

Data collection

The study employed both qualitative and quantitative methods of data collection. Random sampling was used to select sampling sites.

Recruitment and training of citizen science research counterparts and preliminary survey of the study area

To ensure smooth technology transfer and ensure quality of citizen science data collection and processing, this study used 8 self-motivated research counterparts, with minimum qualification of the Malawi School Certificate of Education (MSCE) (equivalent to O-level). The research counterparts were identified with assistance from the four local disaster risk management (DRM) committees in Karonga Town. Each DRM committee was requested to provide a list of potential men and women with good MSCE. It was easier for the DRM committees to identify the potential candidates because Karonga is one of the towns in Malawi that has high literacy rates with a number of unemployed MSCE holders. The identified candidates underwent some interviews and two successful self-motivated research counterparts were selected to represent each of the four DRM committees. Equal opportunity was provided to both men and women during the entire recruitment process. The successful research counterparts were trained to equip them with both theoretical and hands-on experience on principles of of citizen science for water quality monitoring and risk communication among

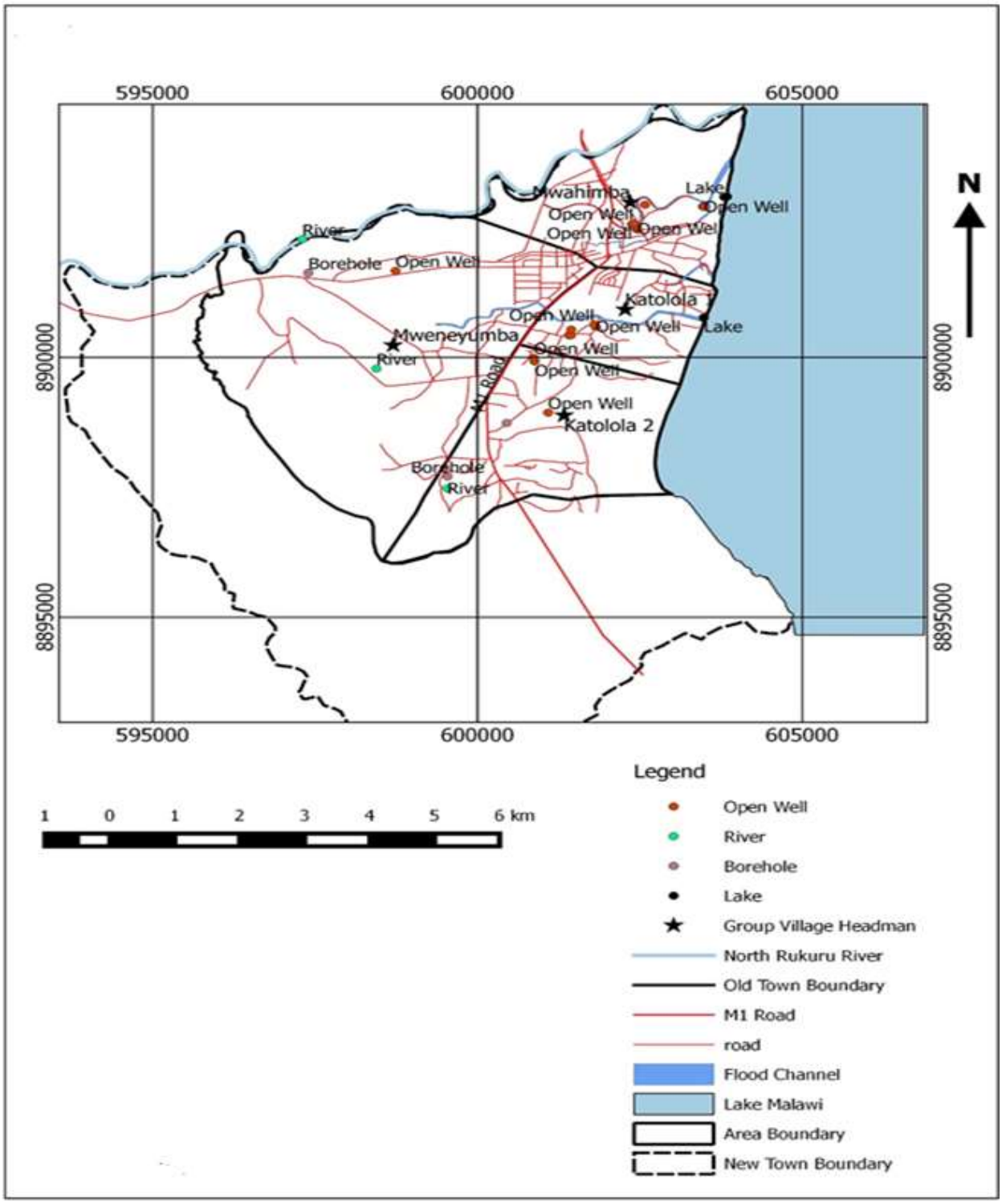

Figure 1. Map of Karonga Town showing sampled water points.

others.

The training sessions were conducted from 18 to 22nd January, 2016 and 21 to 23rd June, 2016 at the Karonga District Education Office (Figure 2). The training sessions covered the following topics: Introduction to the concept of citizen science, research ethics, basic bio-physicochemical properties of water, key water quality

Figure 2. Citizen science research counterparts' theoretical and hands on training sessions with portable multi-meters for onsite water quality analyses in Karonga Town.

monitoring parameters, the analytical protocol for water quality monitoring, water quality standards used in water quality monitoring, application of the Citizen Science Model in Urban ARK water quality monitoring, water sample collection (Figure 2), preservation and treatment, water sample analysis, water quality assessment using the water quality index (WQI), basics of quality control and assurance in Urban ARK water quality monitoring, basic analytical instrumentation techniques for water quality analyses, role of citizen science in building resilience and hands on experience on the application of citizen science for water quality monitoring and statistical treatment of analytical data. A debriefing session was conducted after hands-on experience sessions (Figure 2) to highlight grey areas and clarify what was expected of research counterparts when in the field. The data entry clerks were closely supervised and at the end, data was cleaned to check inconsistencies and insert missing values as well as wrongly entered values before starting data analysis.

Prior to the hands-on experiences in the field, a survey of the study area was done to: (1) Locates and map water sources and select the study sites. A global positioning system (GPS, GARMIN-GPSMAP 60Cx, USA) was used to collect the geo-referencing data (northings, eastings and elevation) of all sampled sites; (2) Brief members of the DRM committees and local leaders on principles of citizen science water quality monitoring and building WASH related risks resilience approach as part of technology transfer; (3) To conduct a household as part of assessing the full spectrum of risks. The household survey covering the whole town whose results partly inform this work used the sample frame provided by the National

Statistical Office, that is, 8007 households in the entire Karonga Town enumeration area (NSO, 2010), from which 380 households were selected for interviews (Appendix 1). A systematic random sampling was used, and every 26th household was interviewed. This household survey involved administration on an open-ended questionnaire to the selected households. The household survey questionnaire was developed in English and then translated and conducted in the common languages of the area, Chi-Nkhonde and Chi-Tumbuka. Translation relied on a team of the citizen science research counterparts from the community, who were given a two-day training session on administering the questionnaire. The questionnaire was pre-tested and revisions were made to improve both face and content validity prior to administration.

Personal observations were made to observe sanitary risks impacting on a particular water source in terms of: protection of the water sources, proximity of sanitary facilities such as toilets, rubbish pits, garbage dumpsites, stagnant water pool, animal kraal, garden, graveyards etc, nature of activities in the water resource's recharge zone and depth and mouth diameter of the water source.

Water sample collection and analysis

Data on water quality is based on water samples which were collected from 27 randomly selected unprotected water sources in Karonga Town. The water samples were collected in triplicate using standard sampling procedures (American Public Health Association (APHA), 2011; Malawi Bureau of Standards (MBS), 2005). The water samples were collected using pre-cleaned polyethylene bottles which were rinsed thrice on site with water from the sources prior to collection of water samples. The water samples were analyzed for the following physico-chemical parameters: pH, total dissolved solids (TDS), electrical conductivity (EC), faecal coliform and total coliform on-site. The levels of pH, TDS and EC were determined using a pH-TDS-EC multimeter (Hanna instruments, Model HI 9812). The multimeter was calibrated using standard buffer solutions of pH 4.00 and 7.00 before measuring pH.

For microbial analyses, the petrifilm and multiple tube method, also known as the dilution method or the most probable number method, and membrane filtration were used to examine faecal and total coliform organisms. Sample bottles were flame sterilized using tissue paper soaked in 70% methanol for 30 to 60 s and rinsed three times with source water to minimize the risk of external contamination (Paqualab Manual, 2005). In membrane filtration method, known volumes of water were filtered through each of two membrane filters consisting of a cellulose compound with a uniform pore diameter of 0.45 µm *in situ* within 60 s after sampling in accordance with internationally recognized standards techniques (American Public Health Association (APHA), 2011). The bacteria were retained on the surface of the membrane filter. Both membranes were incubated *in situ* in a potable incubator for preliminary period at relatively low temperature of 30 ◦C, and then changed to a higher temperature, one at 35 or 37°C and one at 44°C. Acid producing colonies were counted after a total incubation time of 18 h. The results were respectively a presumptive membrane faecal coliform count, and a presumptive membrane *E. coli* count.

For offsite physicochemical analyses, the levels of chlorides, fluorides carbonates, bicarbonates, sulphates, phosphates and nitrates were analysed using ion chromatography (Dionex DX 500). Titrimetric methods were used to determine the concentrations of Total hardness (EDTA) and total alkalinity (acid). The total concentrations of Na^+, K^+, Ca^{2+} and Mg^{2+} were determined using atomic absorption spectroscopy (Buck Scientific Model 200A) at specific wavelengths as follows: Potassium (K) (769.9 nm), calcium (Ca) (422.7 nm), magnesium (Mg) (285.2 nm) and sodium (Na) (589.0 nm). The levels of suspended solids were determined gravimetrically.

Data analysis

The quality of the chemical data was assessed by checking ion balances. The biological, physical and chemical water quality results obtained during this study was compared to maximum values recommended by the Malawi Bureau of Standards Board (MBS) (2005), the MS 733:2005 for water quality. For statistical analyses, R statistical software was used to describe temporal and spatial distribution of the analyses. Descriptive statistics, computed at 95% confidence level, provided the concentration mean, median, standard deviation, outliers, as well as normality distribution in different water sources. All chemical variables determined in samples were analyzed using analysis of variance (ANOVA) at 95% confidence level, Hierarchical Cluster Analysis (HCA) and Principal Components Analysis (PCA). The ANOVA was used to determine any statistically significant spatial variations among levels of the analyses (Appendix 1). The HCA and PCA were used as a quantitative and independent approaches for water classification allowing grouping of the water samples and making of correlations between chemical parameters and water samples, respectively.

The HCA was implemented in R using stats package, and was performed using a combination of Ward's linkage method and adopted the Euclidean distances as a measure of dissimilarity. The WQI was computed to turn multifaceted water quality data obtained from the survey, citizen science water quality monitoring program as well as laboratory analyses into simple information that is comprehensible and useable by the public to assess overall quality of water at a specific water points (Prasad et al., 2013). The computed WQI and water quality ratings were given single ratings which were used to categorise the water into different categories as shown in Table 1, the form as well as the extent of water pollution in the water sources used by households and businesses in the town and the risks this brings.

Ethical consideration

Ethical consideration was observed through following all the regulations set by the National Commission for Science and Technology (NCST) of Malawi as part of the Urban Ark research project in Karonga.

RESULTS AND DISCUSSION

Sources of domestic water in Karonga Town

The preliminary household survey, which utilized 380 households, showed that up to 90.2% of the respondents claimed to have access to potable water as follows: Piped water inside the house (17.4%), piped water on the plot (41.8%), on neighbors' plots (11.3%) and communal water pipe (19.7%) (Figure 3). However, 7.4 and 2.4% of the respondents access drinking water from boreholes and river/lake/well, respectively (Figure 3). It is against this background that the citizen science research team sought to establish type and nature of unsafe water sources. To this effect, it was established that the majority of households who reported accessing water samples from unsafe water sources (41%) were using

Table 1. The WQI rating scale.

WQI (%)	Water quality rating category and interpretation
95-100	Excellent water quality (does not require treatment before human consumption)
91-94	Very good water quality (does not require treatment before human consumption)
71-90	Good water quality (require minor treatment works before human consumption)
51-70	Medium or average water quality (reasonable potable water which require advance and conventional treatment before human consumption)
26-50	Fair water quality (polluted water that has doubtful potable use)
0-25	Poor water quality (highly polluted water that is unacceptable for human consumption)

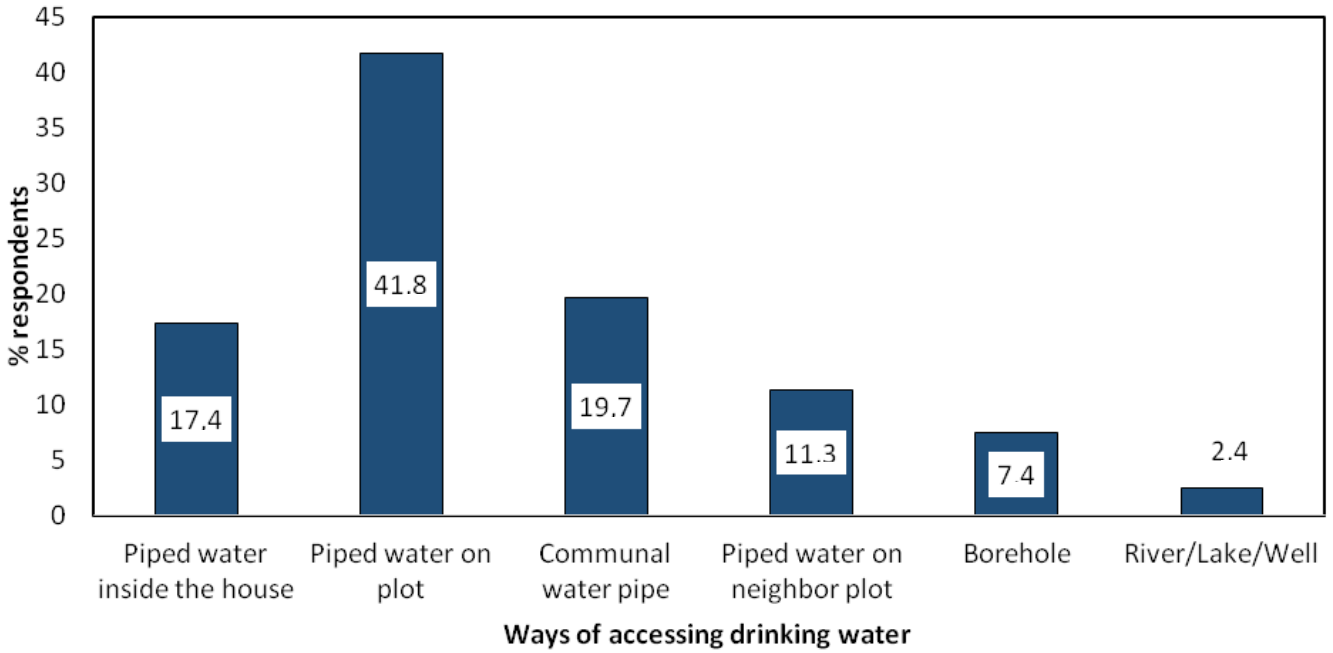

Figure 3. Ways of accessing water for respondents in Karonga Town.

uncovered shallow wells (with average depth= 3.4 ±0.9 m, and average diameter range of 0.48-1.1 m) as their most frequent and reliable source of water (Figure 4). The remaining 22, 15, 11 and 11% were using covered shallow wells, streams/rivers, Lake Malawi and boreholes as their most and reliable source of water, respectively (Figure 4). In case of both covered and uncovered shallow wells, the majority (78%) were reported to have been aged less than 5 years, suggesting that issues of access to water in Karonga Town were serious and demand for water was increasing over the years. On average, each unsafe water source was reported to be serving up to 50 or more households, suggesting that, on average more than 300 individuals were relying on each particular water source. The majority of the respondents pointed out that they opt for unsafe water sources because of the large monthly water bills provided by the Northern Region Water Board (NRWB), intermittent piped water supply and unavailability of steady and continuous flow of piped water provided by the NRWB during daytime. Similar observations were made in Mzuzu City Northern Malawi by Wanda et al. (2012a) who noted that people were resorting into unsafe water sources due to low water pressure and intermittent water supply problems by the utility.

Up to 51.2% of the respondents perceived the water from the boreholes, shallow wells, streams and lake as of good taste and safe for consumption with 70.5 and 64.1% indicating that the shallow well water was odourless and good turbidity, respectively (Table 2). Nevertheless, some waterborne diseases such as cholera (18.68%), diarrhea (6.05%) and dysentery (6.58%) were reported to be

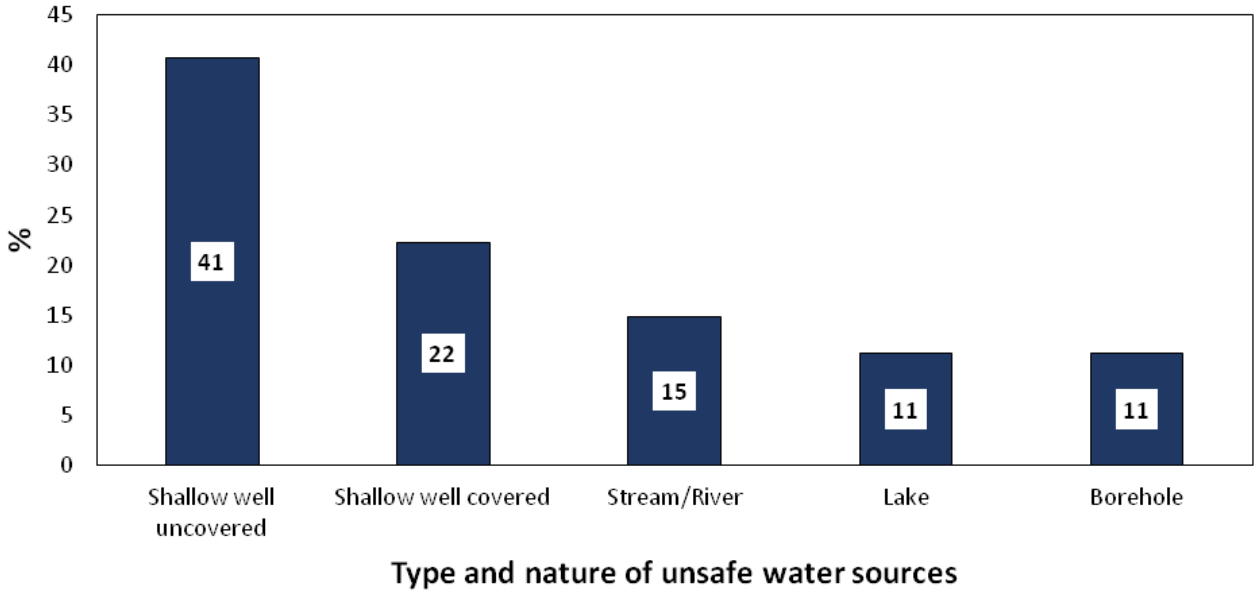

Figure 4. Type and nature of unsafe water sources in Karonga Town.

Table 2. Perception of some of the attributes of unsafe water sources (that is, lake, rivers, shallow wells and boreholes) by households.

Condition	Taste of water (%)	Odour of water (%)	Turbidity of water (%)
Good	51.2	70.5	64.1
Bad	46.1	28.5	35.9
Not sure	2.7	1.0	0.0
Total	**100.0**	**100.0**	**100.0**

commonly experienced by family and community members of households in Karonga Town (Figure 5). Occurrence of such diseases was largely attributed to use and consumption of contaminated water by the affected members of the community in Karonga Town (Manda and Wanda, 2017).

WASH related risks impacting on domestic water sources

An earlier study by Manda and Wanda (2017) reported the occurrence of multiple every day, small and large disaster risks which have the potential to worsen WASH related risks and health indicators thereby leading to premature deaths in Karonga Town. These disasters also affect both quantity and quality of water in addition to damaging sanitary facilities (Karonga District Council, 2010). This study found that up to 51.1% of the households used traditional pit latrines, 27.9% used ventilated improved pit (VIP) latrines, 13.2% used flush toilets connected to septic tanks, 4.2% used neighbours' pit latrines and 3.7% do not have toilets (Figure 6). On-site water source inspections revealed that all the shallow wells were not lined and covered. In addition, various sanitary factors that impact on groundwater, surface water sources as well as the health of households were noted. These included pit latrines constructed <100 m away from shallow wells/boreholes/rivers (27.4%), indiscriminate disposal of wastes (6.6%), graveyards located <100 m away from shallow wells/boreholes/rivers (1.1%), lack of hand washing facilities (8.2%), open defecation due to lack of toilets (2.4%), availability of stagnant waters close to boreholes and shallow wells (2.4%), lack of proper drainage system (5.0%) (Table 3 and Figure 7).

It was also observed that the sanitation around the majority of boreholes was very poor with dirty stagnant waters coming from water collection containers and washing sinks. The water sources were thus at risk of contamination through direct and indirect leakages from anthropogenic activities such as washing of clothes close

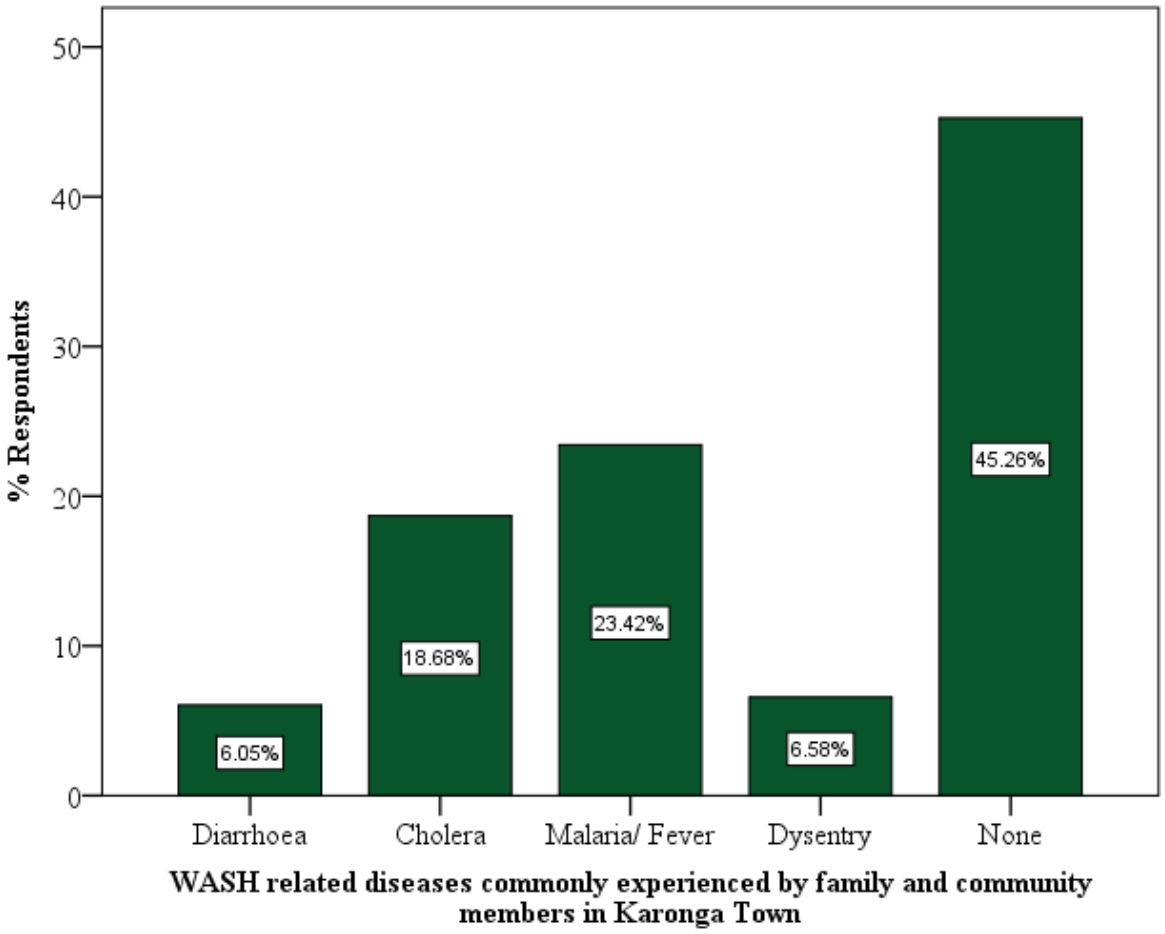

Figure 5. WASH related diseases commonly experienced by family and community members in Karonga Town.

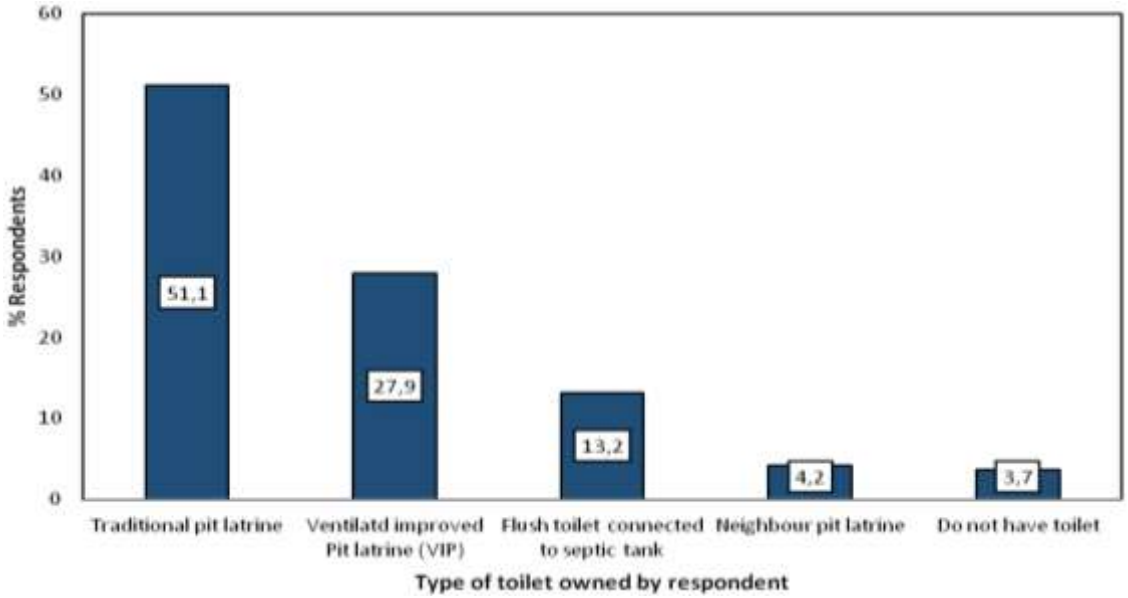

Figure 6. Type of toilets owned by respondents in Karonga Town.

Table 3. Summary of the sanitary risks impacting on a particular water source in Karonga Town.

Sanitary risks impacting a particular water source	%
Pit latrine <10 meters away from wells/boreholes/river	27.3
Indiscriminate disposal of wastes	6.5
Graveyard < 10 meters from wells/boreholes/river	1
Lack of hand washing facilities	8.1
Lack of toilet/pit latrine (open defecation)	2.3
Stagnant waters close to wells/boreholes/river	2.6
House connected to road/drainage	4.9
No risk	47.1
Total	**100**

Figure 7. Major sanitary risks impacting on water sources in Karonga Town such as proximity of water sources to sanitary facilities such as pit latrines, small gardens, stagnant water sources and uncovered nature of the water sources among others.

Table 4. Summary of descriptive statistics showing minimum, maximum, mean, standard deviation (p-value at 95% confidence level) and Malawi Bureau of Standards Board (MBS) water quality specifications (* = specifications not found).

Parameter	MBS limit	Min.	Max.	Mean	Std. Dev.	p-value
pH	6.0-9.5	5.20	8.30	6.92	0.65	0.171
TDS (mg/L)	2000	50.00	580.00	283.70	141.97	0.015
EC (µS/cm) at 20 ◦C	3500	252.00	1380.00	719.56	300.33	0.007
E-Coli (cfu/100mL)	0	0.00	7200.00	751.85	1535.82	0.911
Total Coliform (colonies/100 mL)	*	100.00	>20000	10596.29	8781.12	0.514
CO_3^{2-} (mg/L)	*	0.00	78.00	4.27	15.75	0.004
F^- (mg/L)	6	0.00	1.05	0.28	0.37	0.406
Mg^{2+} (mg/L)	200	0.60	42.45	6.66	9.29	0.943
Ca^{2+} (mg/L)	250	3.20	233.60	59.30	60.411	0.199
Na^+ (mg/L)	500	3.30	138.70	21.89	30.76	0.993
K^+ (mg/L)	1	0.10	3.60	0.87	0.89	0.975
Cl^- (mg/L)	750	1.00	178.00	21.69	37.18	0.129
SO_4^{2-} (mg/L)	800	1.10	126.20	9.66	23.39	1.000
HCO_3^- (mg/L)	400	16.00	746.44	220.60	210.42	0.652
NO_3^- (mg/L)	45	0.02	8.25	2.02	1.91	0.974
Turbidity (NTU)	25	0.00	230.00	20.06	43.16	0.000
Suspended solids (mg/L)	*	2.00	217.00	19.30	40.60	0.000
Total Hardness (mg/L)	800	15.00	616.00	175.41	174.33	0.307
Total Alkalinity (mg/L)	*	13.00	612.00	187.81	182.39	0.566
Electrical balance (%)		1.92	9.04	6.71	2.44	
WQI (%)		40.00	65.56	52.39	7.90	

to water sources, inadequate vegetation filtering buffers, rubbish pits, stagnant water pool, toilets, animal kraal, graveyards and dumpsites located close to the water sources.

As stagnant water pools become breeding sites for mosquitoes malaria was prevalent (23.42 %) in the study area (Figure 5). It is worth noting that the WASH related risks contribute to everyday risks that lead to premature deaths among the majority of residents in the town.

Similar observations were also made by Manda and Wanda (2017), who reported that inadequate provision for WASH services is one of the everyday risks impacting on the population, especially those from low income areas of the town. Manda and Wanda (2017) also observed that the majority of the population in Karonga Town lived in flood-prone areas, specifically along rivers, where flooding was annual. This suggested that the economic advantages of living in such flood prone places significantly outweigh the perceived risks of flooding (United Nations Office for Disaster Risk Reduction, 2011).

Microbial and physicochemical quality of the water

The mean values of the test water quality parameters for water samples from unprotected sources in Karonga Town have been presented in Table 4. The calculated electrical balance errors were ≤±10% (Table 4) suggesting good quality of the chemical data. Among all the physicochemical parameters analysed, pH is one of the most important parameters which determine the suitability of water for various purposes. The levels of pH ranged from 5.20 to 8.30 with an average of 6.9 indicating occurrence of slightly acidic to slightly alkaline waters (Table 4). It was observed that 7.4% of water samples registered pH values not in the range of the MBS specification of 6.0 to 9.5. These water samples had their pH values below 6.0. Such waters are acidic, soft and tend to be corrosive in nature and not suitable for direct human consumption (Wilkes University, 2007).

The levels of EC and TDS ranged from 255 to 1380 µS/cm and 50 to 580 ppm at 20°C, respectively indicating water sources of lower levels of mineralization (Table 4). Both EC and TDS were within MBS specifications of 3500 µS/cm and 2000 ppm, respectively. Similarly, in all water samples, levels of chlorides (range = 1.0 to 178 mg/L), sulphates (range = 1.10 to 126.20 mg/L), nitrates (range = 0.02 to 825 mg/L), total hardness (range = 15.0 to 616.0 mg/L), carbonates (range = 0 to 78.0 mg/L), sodium ion (range = 3.30 to 138.70 mg/L), calcium ion (range = 3.20 to 233.60 mg/L) and magnesium ion (0.60

to 42.45 mg/L) were within the permissible levels of both WHO and MBS (2005) water quality guidelines (Table 4). On the other hand, some of the water samples registered levels of potassium ion (22.2%), bicarbonate ion (22.2 %) and turbidity (11.1%) above the MBS water quality specifications of 1 mg/L, 400 mg/L and 25 NTU, respectively. The relatively higher turbidity values in water indicate the intrusion of run-off which could be attributed by the uncovered and unlined nature of shallow wells or soil disturbance and re-suspension within the well during water withdrawal. Similar observations were made by Pedersen and Price (2005), who also reported that increased levels of turbidity were as a result of inflow of sediments into water sources. Higher levels of HCO_3^- and K^+ ion could be attributed to mineralization processes within the water catchment (Wanda et al., 2011).

The levels of feacal coliforms (*E. coli*) ranged from 0 to 7200 colonies/100 mL (Table 4) which implied that the majority of water samples (56%) were above the WHO drinking water quality specification of 0 colonies/100 mL as well as the MBS standards and at high risk of faecal contamination arising from a number of sanitary risks such as open defecation, uncovered water sources and water sources located close to pit latrines. Such waters are not fit for domestic purposes prior to treatment. In terms total coliform, the levels ranged from 100 to 8700 colonies/100 mL with an average of 3073 colonies/100 mL. Generally, the average faecal coliform density was relatively high in water from the lake, streams, uncovered water sources compared to the covered ones and boreholes. This indicated possible contamination by human and animal faeces and possibly from naturally-occurring bacteria mainly because the water sources were open, not lined and often located very close to sanitary facilities which made the water highly prone to microbial contamination. The results agree with those of Tandlich et al. (2008), Pritchard et al. (2007), Pritchard et al. (2008) and Mkandawire and Banda (2009) who also reported possible microbial contamination of water sources due to their open, not lined nature coupled with their proximity to sanitary facilities. Computation of the analysis of variance showed that only turbidity had statistically significant spatial variation ($p < 0.05$) at 5% significance level (Appendix 2).

Overall water quality indices for each of the sampled sites

Principal component analysis

The principal component analysis isolated three major principal components (PCs) which controlled 70.01% of the observed variations in water quality (Table 5). These included PC 1, which controlled 39.19% of the variance, PC 2, which controlled 17.36% of the variance and PC 3, which controlled 14.46% of the variance. The PC 1 had high loadings in Ca^{2+}, total hardness, F^-, Mg^{2+}, EC, total alkalinity, TDS, HCO_3^- and SO_4^{2-}. Most of the parameters highly loaded in PC1 are products of mineralization processes, suggesting the influence of such process on the observed variations in water quality in the town. PC 2 had high loadings in turbidity, suspended solids, *E. coli*, chlorides, total coliform, Na^+, K^+ and NO_3^-. These factors could be attributed to the influence of anthropogenic activities such as municipal wastes, run-off and agricultural activities impacting on water resources in the area. PC 3 had high loadings on pH only. This suggested that pH is a major factor and that acidification of water bodies due to natural rainwater and other hydrochemical processes within catchments had a greater impact on the observed variations in water quality (Wanda et al., 2011).

Hierarchical cluster analysis

The hierarchical cluster analysis (HCA) revealed two main clusters (1 and 2) (Figure 8). Cluster 1 was composed of comprised of 12 water sources. Cluster 2 was composed of total of 15 water sources. The two clusters mainly differ in their level of mineralisation and microbial quality. Of the two clusters, Cluster 1 was more mineralized and had higher levels of E-coli and total coliform colonies/100 mL. Based on topographic patterns, cluster 1 members are located at lower topographic levels designated as discharge zones in terms of water flow. Furthermore, water sources in cluster 1 lay in areas where there was a higher rate of open defecation (3.7%) in the bushes, the lake as well as sand and some households relying on neighbors' toilets (4.2%) (Figure 6). It was generally observed that the water was of low mineralisation, indicating that the water in the study area was derived from recent recharge (Chimphamba et al., 2009; Wanda et al., 2011).

Overall water quality index

None of the sampled sites during entire study period registered a water quality index (WQI) of 100% (Table 6). The results of the water quality index (WQI) ranged of 40.00 to 65.56%, representing bad-medium water quality (WQ) ratings (Table 6). Specifically, up to 55.6% of the water sources registered a medium WQ rating whereas the remainder, 44.4% registered bad WQ rating. Nevertheless, the results suggested that none of the water sources was suitable for direct human consumption without treatment as all the waters were slightly polluted. The results suggested some grey areas and that there is need to intensify campaigns for substantial onsite water treatment or point of use water treatment works by households using the water for direct consumption

Table 5. Factor loadings for each parameter, a percentage variance for each principal components and the cumulative variance for the principal components principal components behind water quality in Karonga Town.

Parameter	Principal components (PCs)		
	1	2	3
Ca^{2+}	0.977		
Total Hardness	0.961		
F^-	0.931		
Mg^{2+}	0.931		
EC	0.927		
Total Alkalinity	0.899		
HCO_3^-	0.899		
TDS	0.822		
SO_4^{2-}	0.732		
Turbidity		0.946	
Suspended solids		0.941	
E-Coli		0.817	
Cl^-		0.776	
Total Coliform		0.739	
Na^+		0.729	
K^+		0.727	
NO_3^-		0.576	
pH			0.594
% of variance for each PC	39.190	17.363	14.455
Cumulative % for the PCs	39.190	56.553	70.008

(Wanda et al., 2012b).

Role of the citizen science approach in building resilience of WASH systems to water-related hazards in Karonga Town

Resilience has been defined as the capability of a society, community or system to undertake its economic, ecological and social growth and development strategies, while managing its disaster risk over time in a way that contributes to sustainable growth and helps to mitigate disaster risk (Szoenyi, 2016). Historically, DRM was the jurisdiction of stakeholders in the humanitarian sector who mostly focused on response and recovery as opposed to dealing with fundamental factors that lead to vulnerability. Even though the cost-benefits of risk reduction and preparedness measures are known to be higher than that for response and recovery, there still remains a higher tendency for increased financial support towards disaster recovery than to prevention (Pasteur and McQuistan, 2016). The citizen science approach used in this study was aimed at correcting this bias and shift the focus towards WASH risk and water related hazard reduction in Karonga Town. This approach is the first of its kind in the disaster disk reduction initiatives in Malawi which centered on building the capacity of communities in exploring enhance resilience of WASH systems the town. Integrating the public into scientific research through the citizen science approach has the virtue of improving management of water resources holistically with respect to: Prevention, protection, preparedness (through technology transfer initiatives among others), emergency response and recovery and lessons learned (returning to normal conditions as soon as possible and mitigating both the social and economic impacts on the affected population) (Oates et al., 2014). The approach used in this study was based on lessons learnt from some of the principles of the vulnerability to resilience (V2R) framework proposed by Pasteur and McQuistan (2016). Just like the V2R framework, our citizen science approach training sessions emphasized on the interlinkages between the wellbeing of the communities and local drivers of the WASH related risks and hazards at local, level (Pasteur and McQuistan, 2016). Figure 9 illustrates V2R framework, unravelling the complicatedness of a systems approach to resilience in a simple diagram.

Through the citizen science approach, both research counterparts and communities in Karonga were equipped with knowledge and skills of investing in WASH related risk reduction measures prior to occurrence of disaster

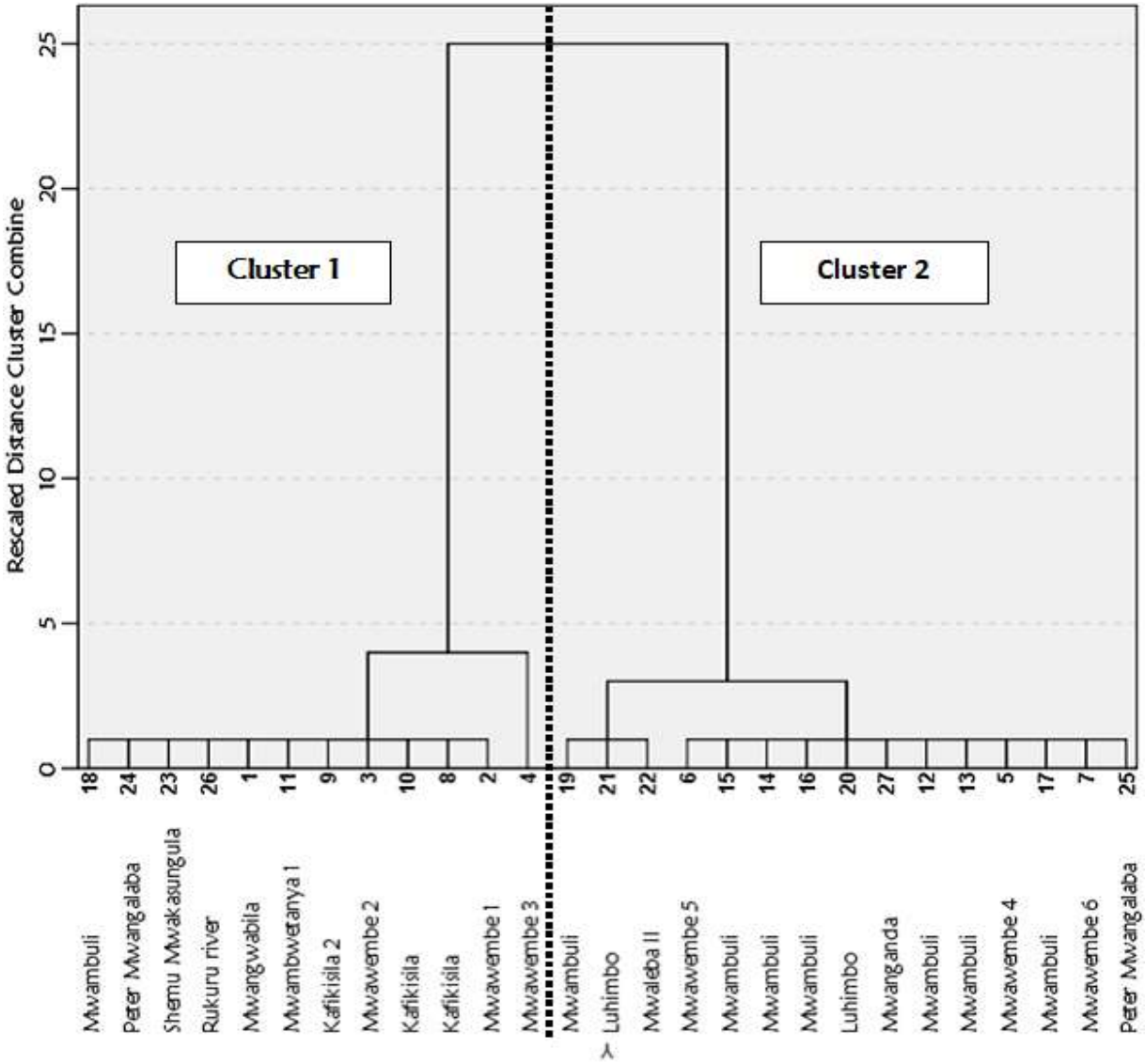

Figure 8. Dendrogram showing resulting of the hierarchical cluster analysis in Karonga Town.

events thereby assisting them to make informed and appropriate no-regret choices, principally in the context of an unpredictable future. Over and above, through the citizen science approach, the citizen science research counterparts and communities in Karonga Town have been empowered on WASH related risk knowledge, monitoring, communication as well as dissemination and the ability to respond. The research counterparts were involved throughout the research project including dissemination of research findings to communities and stakeholders as part of risk communication and technology transfer. Unlike the many perceived challenges outlined by Bonardi et al. (2011) and Kremen et al. (2011), the results of the citizen science water quality monitoring showed high level of accuracy in the obtained data. Electrical balance errors were ≤±10% showing good quality of the chemical data were obtained by the research counterparts throughcitizen science approach. This also suggested that with proper planning and implementation, the citizen science approach could help build resilience with high degree of accuracy and reliability.

CONCLUSIONS AND RECOMMENDATIONS

This study utilized the citizen science approach in assessing water quality, identifying WASH related risks and enhancing the understanding of how the resilience of WASH systems to water-related hazards (e.g. floods and

Table 6. Summary of results of water quality index and water quality rating.

Sample site	Description	Cluster	pH	*E. coli* (colonies/100 ml)	WQI (%)	WQ rating
Mwangwabila	Shallow well uncovered	1	6.6	200	45.78	Bad
Mwawembe 1	Shallow well uncovered	1	6.7	3400	43.68	Bad
Mwawembe 2	Shallow well uncovered	1	6.6	2000	43.16	Bad
Mwawembe 3	Shallow well uncovered	1	6.8	7200	46.47	Bad
Mwawembe 4	Shallow well uncovered	2	6.1	0	57.20	Medium
Mwawembe 5	Lake water	2	7.5	100	48.88	Bad
Mwawembe 6	Lake water	2	7.6	0	56.68	Medium
Kafikisila	Shallow well uncovered	1	6.8	1700	46.04	Bad
Kafikisila 2	Shallow well uncovered	1	6.9	0	57.45	Medium
Kafikisila	Shallow well uncovered	1	7.1	2000	44.18	Bad
Mwambwetanya 1	Lake water	1	7.8	400	50.10	Medium
Mwambuli	Shallow well covered (Toilet < 2m away)	2	6.7	0	58.93	Medium
Mwambuli	Shallow well uncovered but lined	2	7.0	0	60.65	Medium
Mwambuli	Shallow well covered (Toilet < 5m away)	2	6.8	0	64.38	Medium
Mwambuli	Shallow well covered (Toilet <10m away)	2	6.6	500	49.69	Bad
Mwambuli	Shallow well covered (Toilet < 8m away)	2	7.0	0	57.11	Medium
Mwambuli	Shallow well covered	2	6.6	300	45.69	Bad
Mwambuli	Shallow well uncovered	1	6.9	900	52.66	Medium
Mwambuli	Shallow well covered	2	6.4	100	54.08	Medium
Luhimbo	Borehole	2	7.0	0	65.56	Medium
Luhimbo	Shallow well uncovered but lined	2	7.0	0	62.76	Medium
Mwaleba II	Phapa river	2	8.1	0	63.79	Medium
Shemu Mwakasungula	Bwiba, Mwaskakata water stream	1	5.9	500	40.00	Bad
Peter Mwangalaba	Bwiba, water stream	1	5.2	800	40.16	Bad
Peter Mwangalaba	Borehole	2	7.3	0	52.55	Medium
Rukuru river	River	1	8.3	200	44.95	Bad
Mwanganda	Borehole-Rukuru Primary school	2	7.5	0	61.92	Medium

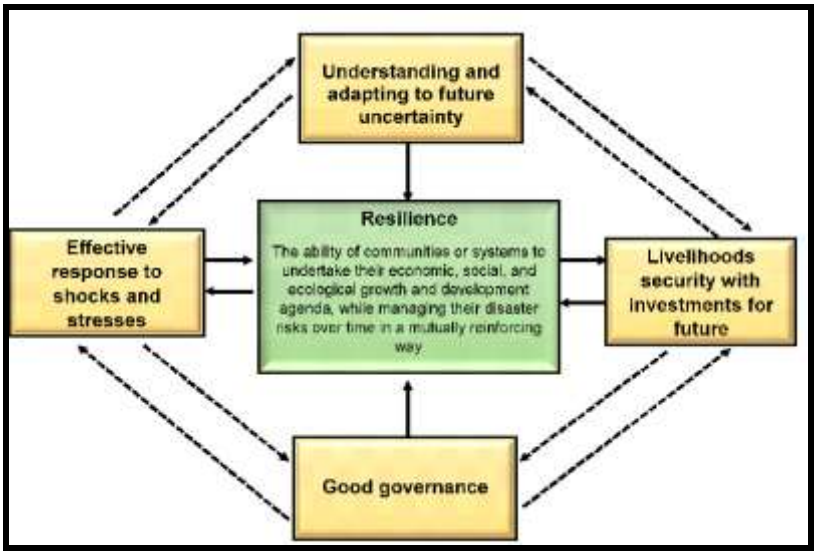

Figure 9. Framework for vulnerability to resilience (adapted from Pasteur and McQuistan, 2016).

water scarcity) can be improved. The community representatives took the lead in the assessment of risks associated with water pollution. Through the citizen science water quality monitoring approach, data was gathered and tested by trained non-professionals, the citizen science research counterparts. This capacity building or technology transfer was one of the major contributions of Urban Africa Risk Knowledge project in Karonga Town as it empowers local communities to understand the monitoring and recording of water quality in their area.

It was observed that water from the majority of shallow wells, rivers/streams, lake and boreholes were highly contaminated with E-coli, which were considerably higher than MBS water quality specifications for drinking water. In general, the water is of low mineralization with rock-water interactions and surface pollution from anthropogenic activities such as municipal wastes being responsible for input biological, chemical and physical pollutants especially into the unlined and uncovered water sources. The results of the water quality indices and ratings indicated that the water quality obtained from water sources in the studied area is not suitable for direct human consumption before onsite or household point of use treatment. It is recommended that onsite treatment and point of use water treatment interventions should be instituted to protect the households from further possible consequences of using the water. The application of citizen science approach can be tested in other urban centers in Malawi both to build capacity as we as to speed up scientific data collection for monitoring WASH related risks.

CONFLICT OF INTERESTS

The authors have not declared any conflict of interests.

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APPENDICES

Appendix 1. Households Interviewed per enumeration area.

Enumeration area	Total number of households in enumeration area	Number of households interviewed
EA701	299	14
EA702	176	8
EA703	125	6
EA704	92	4
EA705	256	13
EA706	136	6
EA707	251	12
EA708	672	31
EA709	169	8
EA710	214	10
EA711	492	23
EA712	557	26
EA713	465	21
EA714	165	8
EA715	129	6
EA716	106	5
EA717	292	14
EA718	146	13
EA719	295	17
EA720	149	7
EA721	367	19
EA722	61	3
EA723	152	7
EA724	104	5
EA725	304	14
EA726	289	13
EA727	285	13
EA728	90	4
EA729	203	9
EA730	71	3
EA731	93	4
EA732	85	4
EA733	66	3
EA734	134	6
EA735	31	1
EA736	361	17
EA737	125	6
TOTAL	**8007**	**380**

Appendix 2. The analysis of variance (ANOVA) table for all analyses together with the p-values at 95% confidence level.

Parameter	Group			Sum of squares	df	Mean Square	F	Sig.
pH	Between groups	(Combined)		7.784	16	.486	1.561	.240
		Linear term	Weighted	.155	1	.155	.497	.497
			Deviation	7.629	15	.509	1.632	.219
	Within groups			3.117	10	.312		
	Total			10.901	26			
TDS	Between Groups	(Combined)		392166.296	16	24510.394	1.859	.161
		Linear term	Weighted	83875.799	1	83875.799	6.361	.030
			Deviation	308290.498	15	20552.700	1.559	.242
	Within groups			131863.333	10	13186.333		
	Total			524029.630	26			
EC	Between groups	(Combined)		1483636.800	16	92727.300	1.076	.467
		Linear term	Weighted	240335.873	1	240335.873	2.790	.126
			Deviation	1243300.927	15	82886.728	.962	.542
	Within groups			861565.867	10	86156.587		
	Total			2345202.667	26			
E-Coli	Between groups	(Combined)		14241074.074	16	890067.130	.189	.998
		Linear term	Weighted	1183714.148	1	1183714.148	.251	.627
			Deviation	13057359.926	15	870490.662	.185	.998
	Within groups			47086333.333	10	4708633.333		
	Total			61327407.407	26			
Total coliform	Between groups	(Combined)		1178825962.963	16	73676622.685	.892	.596
		Linear term	Weighted	79982067.846	1	79982067.846	.968	.348
			Deviation	1098843895.117	15	73256259.674	.887	.596
	Within groups			825983666.667	10	82598366.667		
	Total			2004809629.630	26			
CO_3^{2-}	Between groups	(Combined)		2344.320	16	146.520	.357	.968
		Linear term	Weighted	15.974	1	15.974	.039	.848
			Deviation	2328.346	15	155.223	.378	.956
	Within groups			4102.080	10	410.208		
	Total			6446.400	26			
F^-	Between groups	(Combined)		2.563	16	.160	1.742	.188
		Linear term	Weighted	.011	1	.011	.116	.741
			Deviation	2.553	15	.170	1.851	.164
	Within groups			.920	10	.092		
	Total			3.483	26			
Mg^{2+}	Between groups	(Combined)		1065.009	16	66.563	.564	.852
		Linear term	Weighted	39.307	1	39.307	.333	.577
			Deviation	1025.702	15	68.380	.579	.836
	Within groups			1180.055	10	118.006		
	Total			2245.065	26			
Ca^{2+}	Between groups	(Combined)		70606.252	16	4412.891	1.817	.170
		Linear term	Weighted	3149.257	1	3149.257	1.297	.281
			Deviation	67456.995	15	4497.133	1.852	.164
	Within groups			24281.227	10	2428.123		
	Total			94887.479	26			

Appendix 2. Cont.

Na^+	Between groups	(Combined)		16039.487	16	1002.468	1.171	.411
		Linear term	Weighted	299.465	1	299.465	.350	.567
			Deviation	15740.021	15	1049.335	1.226	.381
	Within groups			8561.580	10	856.158		
	Total			24601.067	26			
K^+	Between groups	(Combined)		10.362	16	.648	.631	.801
		Linear term	Weighted	.388	1	.388	.378	.552
			Deviation	9.974	15	.665	.648	.783
	Within groups			10.258	10	1.026		
	Total			20.620	26			
Cl^-	Between groups	(Combined)		29914.779	16	1869.674	3.100	.037
		Linear term	Weighted	271.192	1	271.192	.450	.518
			Deviation	29643.587	15	1976.239	3.276	.032
	Within groups			6031.715	10	603.171		
	Total			35946.494	26			
SO_4^{2-}	Between groups	(Combined)		6951.162	16	434.448	.597	.827
		Linear term	Weighted	120.082	1	120.082	.165	.693
			Deviation	6831.080	15	455.405	.626	.800
	Within groups			7271.906	10	727.191		
	Total			14223.068	26			
HCO_3^-	Between groups	(Combined)		854774.888	16	53423.430	1.802	.174
		Linear term	Weighted	40960.099	1	40960.099	1.382	.267
			Deviation	813814.788	15	54254.319	1.830	.169
	Within Groups			296464.219	10	29646.422		
	Total			1151239.107	26			
NO_3^-	Between groups	(Combined)		68.754	16	4.297	1.628	.219
		Linear term	Weighted	.761	1	.761	.288	.603
			Deviation	67.993	15	4.533	1.718	.195
	Within groups			26.391	10	2.639		
	Total			95.144	26			
Turbidity	Between groups	(Combined)		48283.799	16	3017.737	219.525	.000
		Linear term	Weighted	47950.829	1	47950.829	3488.179	.000
			Deviation	332.970	15	22.198	1.615	.224
	Within groups			137.467	10	13.747		
	Total			48421.265	26			
Total hardness	Between groups	(Combined)		611572.185	16	38223.262	2.140	.112
		Linear term	Weighted	27511.121	1	27511.121	1.541	.243
			Deviation	584061.064	15	38937.404	2.180	.108
	Within groups			178578.333	10	17857.833		
	Total			790150.519	26			
Total alkalinity	Between groups	(Combined)		620014.207	16	38750.888	1.582	.233
		Linear term	Weighted	30009.713	1	30009.713	1.225	.294
			Deviation	590004.495	15	39333.633	1.606	.227
	Within groups			244963.867	10	24496.387		
	Total			864978.074	26			

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Full Length Research Paper

Geoelectrical logging for well screening in prolific aquifers in Ubima, Ikwerre Local Government Area, River State, Nigeria

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Geoelectrical logging was conducted in two well locations in Ubima community, Niger Delta, Nigeria. Self-Potential (SP) and Resistivity (Short Normal and Long Normal) logs were employed. Two points each in wells 1 and 2, respectively, were tested to determine the porosity (ϕ), permeability, *k* and the total dissolved solids (TDS) in the aquifer by derivation from electrical conductivity (EC). The results show that for test zones A and B in well 1, porosity and permeability had values of 20.6% and 49 *m*D, respectively, at depth range of 105 to 120 m and values of 19.4% and 34 *m*D, respectively, at depth range of 120 to 140 m. The total dissolved solid (TDS) derived from electrical conductivity range of 869.5 to 877 µmhoscm^{-1} is between 556 and 561 ppm for these zones. For well 2, test zones A and B, the porosity and permeability values observed were 23.7% and 108 *m*D at depth range of 80 to100 m and values of 22.0% and 71 *m*D at depth range of 100 to 120 m, respectively. A TDS value of 400 ppm was observed for both test zones in this well. This implies that screening and casing installations should be done between 90 and 110 m to ensure that a prolific and good water quality aquifer is delineated for borehole water project in this community.

Key words: Geoelectrical logging, well screening, prolific aquifer, Ubima, Niger Delta.

INTRODUCTION

Different kinds of aquifers systems exist within the subsurface of the Niger Delta (Oteri and Atolagbe, 2003; Etu-Efeotor and Akpokojde, 1990; Ekine and Osobonye, 1996; Amajor, 1991). Some contain mineralized water, saline water and fresh water. Post-drilling geoelectrical logging is a useful technique for hydrogeological characterization of these ground water aquifer systems when chemical analyses are not available. This should necessarily be conducted before screening and casing installations for water supply to wells to ensure that a

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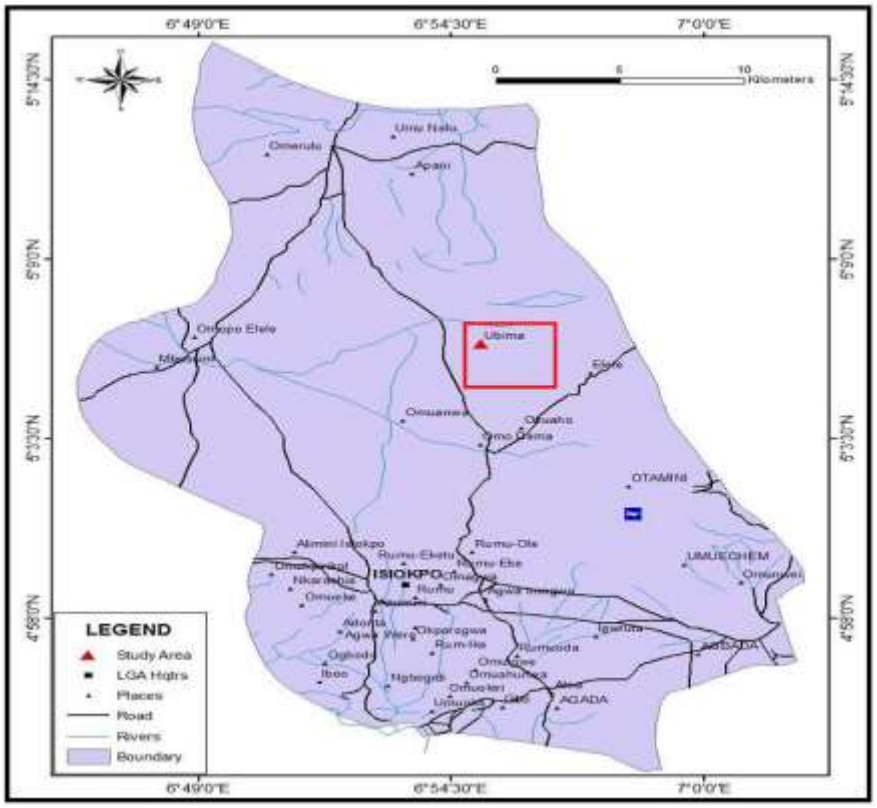

Figure 1. Map of Ikwerre L.G.A. showing the study area– Ubima.

prolific and good water quality aquifer is delineated for any borehole water project.

Etu-Efeotor and Akpokodje (1990) noted that groundwater development and abstraction have been undertaken indiscriminately without regard to the safe yield of aquifers and possibly deterioration of water quality. Most ground water wells are not geoelectrically logged by some drilling outfits before completion and production probably due to the cost implications or lack of expertise. In such cases, the hydrologists rely only on core samples collected at depth intervals while drilling to draw conclusion on the location for screening in the wells. This certainly does not allow for proper delineation of a prolific and good quality aquifer. The result in this case is sometimes, polluted ground water systems with mineralized water, saltwater intrusions and or dry wells after some period of abstraction.

Sequel to the above, geoelectrical logging was conducted in two well locations with drilled depths of about 140 and 120 m, respectively, at Ubima community, Niger Delta, Nigeria. Self-Potential (SP) and Resistivity (Short Normal and Long Normal) Logs were employed. The total dissolved solids (TDS) were determined by the derivation of the electrical conductivity (EC) and ultimately the porosity (ϕ) and permeability and k of the aquifers.

Location of the study area

The study area is Ubima Community located in Ikwerre Local Government Area of Rivers State delimited by Latitude 6°49′0′E and 7°0′0′E and Longitude 4°58′0′N and 5°14′30′N in the Niger Delta (Figure 1).

Geology of the study area

Generally, the Benin Formation constitutes the major aquiferous formation in the study area. It is about 2100 m thick at the basin centre and consists of coarse-medium grained sandstones, thick shales and gravels. The upper section of the Benin Formation is the quaternary deposits which are about 40-150 m thick and comprise sand and silt/clay with the later becoming increasingly more prominent seawards (Etu-Efeotor and Akpokodje, 1990). The Formation consists of predominantly freshwater continental friable sands and gravel that have excellent aquifer properties with occasional intercalations of claystone/shales (Olobaniyi and Oweyemi, 2006).

According to Etu-Efeotor (1981), Etu-Efeotor and Akpokodje (1990), Offodile (2002), Udom et al. (2002), the Benin Formation is highly permeable, prolific and productive and is the most extensively tapped aquifer in the Niger Delta. All the boreholes in the study area are drilled into the Benin Formation.

According to Nwankwoala and Warmate (2014), Ubima is underlain by the Coastal Plain sands, which in this area is overlain by soft-firm silty clay sediments belonging to the Pleistocenic Formation. The general geology of the area essentially reflects the influence of movements of rivers, in the Niger Delta and their search for lines of flow to the sea with consequent deposition of transported sediments. In broad terms, the area may be considered flat. The surface deposits in the area comprise silty-clays. The near surface silty clays are subjected to mild desiccation during the dry season. Substantial seasonal variations in moisture are expected in the area. This could result in some false enhancement of strength in the dry season. The sandy layers underlying the top clay are predominantly medium-to-coarse in grain sizes, fairly well graded and found to exist in various states of compaction (Nwankwoala and Warmate, 2014).

Theory of geoelectrical logging

Spontaneous (self) potential log

Static SP values relate to the chemical activity of the formation water (Ushie, 2001), which also correlates the true resistivity of the formation and the formation of water resistivity. They are used to distinguish lithology such as shaly from sandy formation (Raghumath, 2008). Moore and Rust (1944) stated that electrochemical potential is the primary parameter for the SP log. Electrochemical potential relates to the chemical activities of the formation water by (Schlumberger, 1989):

$$EC = -KLog\frac{a_w}{a_{mf}} \quad (1)$$

Where: EC = electrical conductivity; a_w, = chemical activity of the interstitial water; a_{mf} = chemical activity of the mud filtrate; K = coefficient proportional to the absolute temperature of formation.

Since the chemical activity of the formation water is related to NaCl content and hence the resistivity, Wyllie (1948) proposed that the SP due to the electrochemical activity may also be written as:

$$SP = KLog\frac{R_{mf}}{R_w} \quad (2)$$

Where: K= $0.133T_f + 61$; T_f = formation temperature in degrees Fahrenheit; R_{mf} = resistivity of the mud fluid = 16 Ωm (for NaCl-based mud); R_w = resistivity of the formation water.

Electrical resistivity logs

The resistivity logs used contain both short normal and the long normal. The short normal has a distance of 16" between the potential electrodes, MN, while the long normal has a distance of 64" between the potential electrodes, MN. The apparent resistivity of a medium can be calculated by the formula:

Resistivity, ρ = Resistance, R x Geometry Factor (3)

Where: Resistance, $R = \frac{V}{I}(\text{Ohms})$

Geometry factor is simplified as follows:

Short Normal Resistivity (16") = 4πMN = 12.56 x 16 = 5.02 m
Long Normal Resistivity (64") = 4πMN = 12.56 x 64 =20.0 0m
MN = Separation between the potential electrodes in meters

Archie's (1942) formula relates formation factor, porosity, the resistivity of the rock, and the resistivity of the formation water for clean water-bearing zone as follows:

$$F = \frac{R_o}{R_w} = \frac{R_t}{R_w} = \frac{a}{\phi^m} \quad (4)$$

Where F = formation factor; R_o = resistivity of rock saturated with conducting fluid; R_w = resistivity of formation water; R_t = true resistivity of the formation; *m* = 3 (cementation factor for a high yielding water productive aquifer); *a* = 1 (for unconsolidated sands); ϕ = porosity.

Electrical conductivity (EC) of water estimates the total amount of solids dissolved in water, that is, total dissolved solids (TDS). TDS is measured in ppm (parts per million) or in mg/l. The aquifer quality can be determined by the total dissolved solid (TDS), which is derived from the electrical conductivity values (EC) using Equation 5 (Raghumath, 2008):

$$TDS = 0.64xEC \quad (5)$$

The permeability of the identified aquifers can be estimated using the Jorgensen (1988) equation:

$$k = 84105\left(\frac{\phi^{m+2}}{(1-\phi)^2}\right) \text{ (milliDarcies, mD)} \quad (6)$$

The resistivity of the formation water, R_w can be estimated

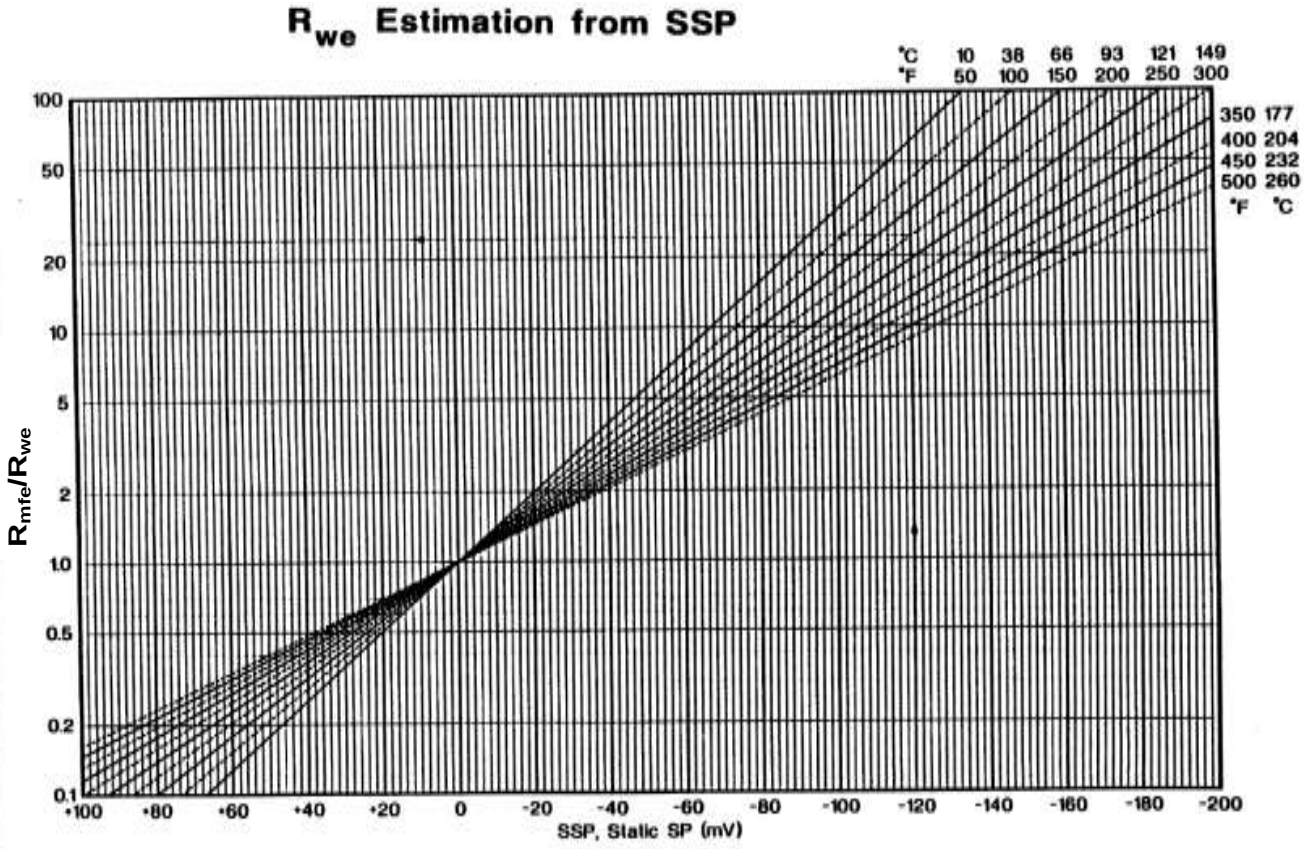

Figure 2. Typical R_{mf}/R_w versus static SP chart (Schlumberger, 1986).

from the Static Spontaneous Potential (SSP) Chart (Figure 2). The wells in this study were at a temperature of 38°C.

METHODOLOGY

Set-up

The Terrameter system used for the logging exercise consist of a basic unit called the SSR-MP-ATS measuring unit with a liquid crystal digital read-out and automatic signal averaging microprocessor, a current control box, and multi electrode probe. The unit has capabilities of measuring spontaneous potential (SP) and resistivity (Short Normal 16" Normal 64" long lateral 18").

Logging

The operations involve the logging of two boreholes, about 10 km apart and drilled to total depths of 140 and 120 m, respectively. Self-potential (SP) resistivity logs (Short Normal 16" and Long Normal 64") were run. The sonde was first lowered step by step towards bottom of the hole. Thereafter, the resistivity and SP measurements were taken when the sonde was being pulled out of the hole. This allowed for a steady sonde and unperturbed logging of electrical resistivity and spontaneous potential in the wellbore.

Digitization of logs

Electrical resistivity (Ωm) and spontaneous potentials (*mV)*) were digitized from the logs only at depth intervals (zones) that are likely to depict good aquifers. These zones indicated by 'two 'sharp kicks' (deviation from a smooth trend) were observed in both wells respectively and indicate two aquiferous layers of different hydrogeological properties.

RESULTS

Figures 3 and 4 show the electrical resistivity and spontaneous potential loggings for both wells, respectively. These show that the depth investigated is made of permeable sandy layer down to the logged depths. Zones A and B were tested in wells 1 and well 2, respectively. The logs show that the electrokinectic streaming potential within regions of 100-120 m for test zone A in well 1 and 80-100 m for test zone A in well 2 are higher than other regions. The digitized values are shown in Table 1.

From Table 2, it can be observed that for test zones A and B in well 1, porosity and permeability have values of 20.6% and 49 *m*D, respectively, at depth range of 105 to

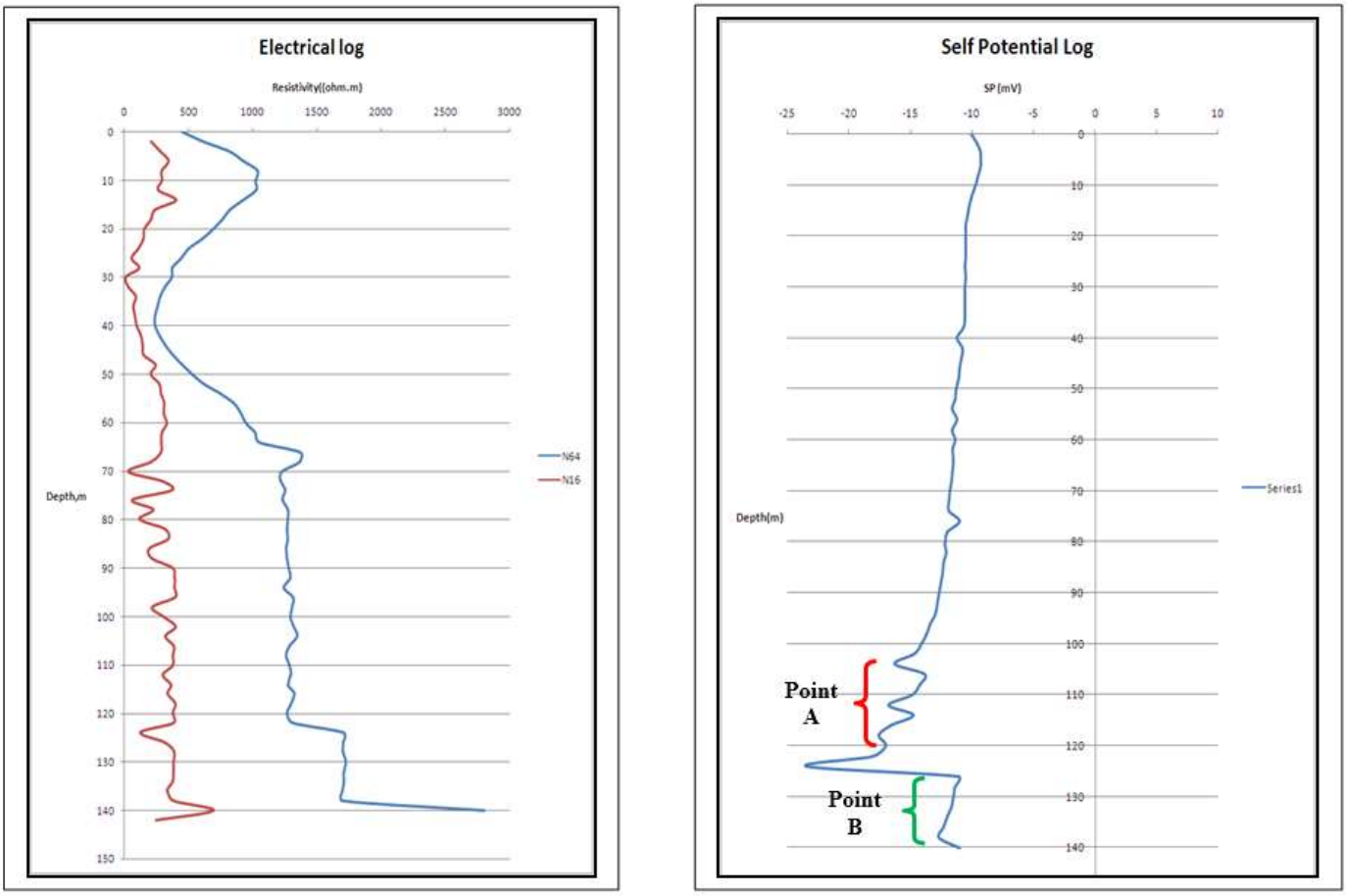

Figure 3. Resistivity and SP logs for Well 1.

120 m and values of 19.4% and 34 *m*D, respectively at depth range of 120 to 140 m. The total dissolved solid (TDS) derived from electrical conductivity range of 869.5 to 877 µmhoscm^{-1} is between 556 and 561 ppm for these zones.

For well 2, test zones A and B, the porosity and permeability values observed were 23.7% and 108 *m*D at depth range of 80 to100 m and values of 22.0% and 71 *m*D at depth range of 100 to 120 m, respectively. A TDS derived from EC value of 625 µmhoscm^{-1} was observed to be 400 ppm for both test zones in this well.

DISCUSSION

The SP logs display a fining upward sequence which depicts a fluvial environment. Two zones (A and B) in each of the two wells were observed to be of hydrogeological interest in this study. The log records indicate the fact that the zones of low SP values (105 to 120 m for well 1 and 80 to 100 m for well 2) are of high porosity and permeability than the other zones. The core samples collected around the two zones (A and B) during the drilling of the two wells consist of friable sands and gravels indicative of good aquifer 'pay zones'. The zones have porosities between 20 and 30% respectively which are consistent with established standards for good aquiferous geological materials (Freeze and Cherry, 1979; Todd, 1980; Driscoll, 1986).

Conclusion and recommendation

The logging details of the borehole as shown identified aquifer 'pay zone' at depth range of about 90 – 125 m. The lithology here comprises sandy and gravely subsurface material. Ultimately, the borehole materials exhibit a productive aquifer thickness greater than 20 m. Screen installation and proper gravel packing of the well annulus, to improve on the efficiency of the well production system should be done at the extensive aquifer zone between 90 and 110 m for a productive

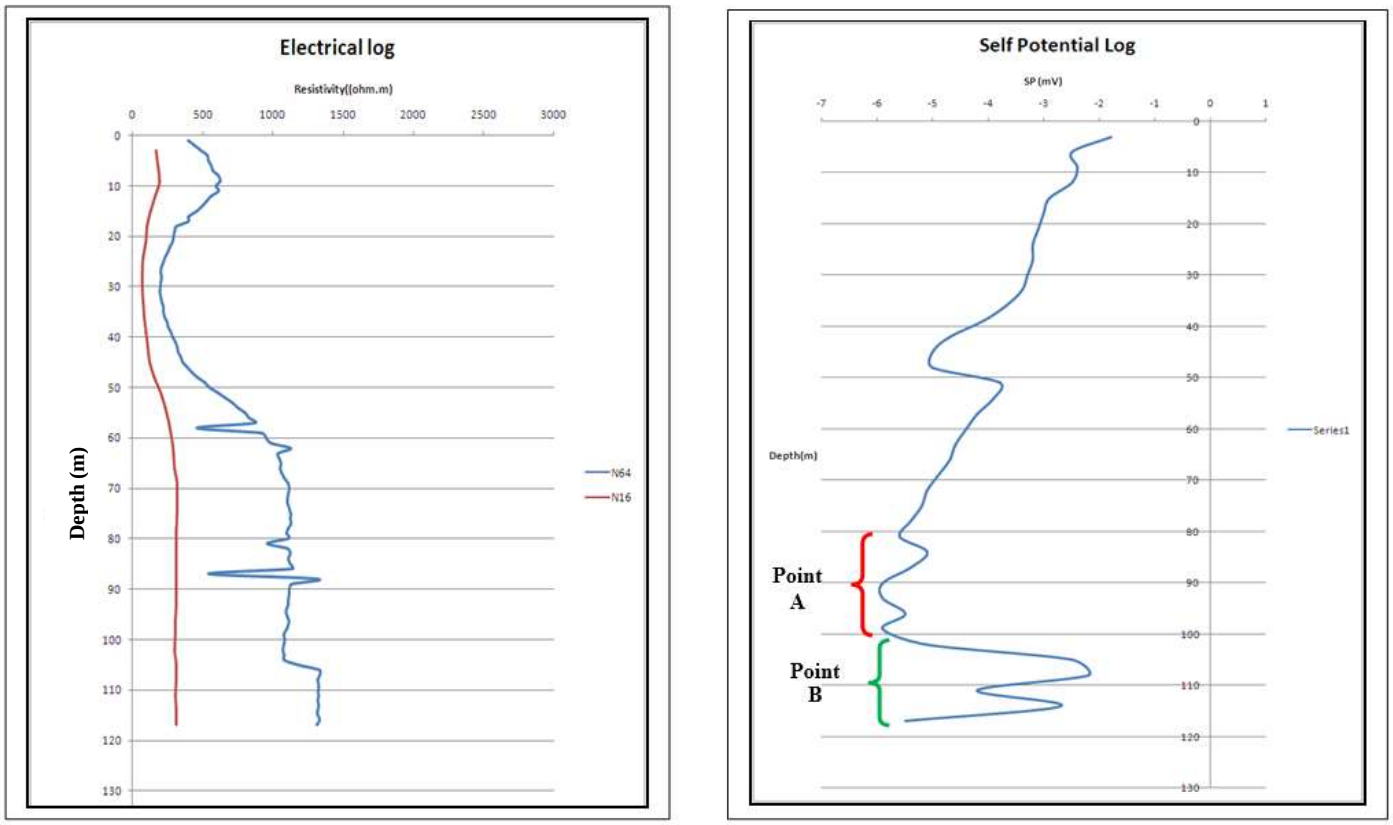

Figure 4. Resistivity and SP logs for Well 2.

Table 1. Summary of the test zones, digitized SP values and formation resistivity, mud weight resistivity and formation water resistivity.

Well	Test zones (m)	Depth (m)	Sp value (mv)	$R_o=r_t$ (Ωm)	R_{mf}/r_w Versus static sp	R_{mf} (Ωm)	R_w (Ωm)
1 (Figure 3)	A	105 -120	-15	1300	1.4	16.0	11.4
	B	125 -140	-8.0	2000	1.1	16.0	14.5
2 (Figure 4)	A	80 - 100	-6	1200	1.0	16.0	16
	B	100 -120	-3	1500	1.0	16.0	16

Table 2. Summary of the computation of porosity, permeability, electrical conductivity and total dissolved solid.

Well	Test zones (m)	Depth (m)	Porosity Φ (%)	Permeability K (mD)	EC ($\mu mhoscm^{-1}$)	TDS (ppm)
1 (Figure 3)	A	105 - 120	20.6	49	877	561
	B	125 - 140	19.3	34	869.5	556.5
2 (Figure 4)	A	80 - 100	23.7	108	625	400
	B	100 - 120	22	71	625	400

groundwater well system in Ubima Community.

The TDS observed for the test zones in both wells are all within acceptable World Health Organization palatability for drinking water.

CONFLICT OF INTERESTS

The author has not declared any conflict of interests.

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Full Length Research Paper

Impacts of bacterial pollution on hand-dug well water quality in Enugu, Enugu State, Nigeria

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This study investigated the effect of bacterial pollution on ground water quality in Enugu urban areas. During the study, samples were collected from ten (10) hand-dug wells (W_1-W_{10}c). Five (5) hand dug wells out of ten (10) were used as control. All samples were taken according to methods described by Federal Ministry of Water Resources (2004). Membrane filtration (MF) method was used to enumerate total coliforms (TC), fecal coliforms (FC) and *Escherichia coli*. Biological oxygen demand, dissolve oxygen, chemical oxygen demand, temperature and pH were also determined. Data were statistically analyzed for mean and standard deviation and the results showed that TC, FC and *E. coli* were influenced by distance and season of the year in all the wells. Their values decreased with increasing distance to pollution sources. Statistical analysis shows that significant difference ($p < 0.05$) was observed with changes in distance and seasons. The results of BOD, DO, COD, temperature and pH show that no significant difference ($p > 0.05$) was observed at different distances and seasons of the year. The values of temperature, DO and pH were within the permissible limit. Presence of bacterial in all the wells was strongly influenced by proximity to the pollution sources. Based on the findings, the research recommends that standard treatment should be given to water from the wells before consumption. The result of this study will be of great importance to the general public, as well as the environmental and health planning unit of government.

Key words: Groundwater, bacterial, pollution, siting distance, season.

INTRODUCTION

When water drains into rivers, lakes and ground water, it carries microbial pollutants as it moves through a watershed. Microbial pollution of water has been a growing crisis in environmental and public health and this has not been fully addressed through scientific research and risk assessment (Joan et al., 1998). Ground water has been historically assumed to be safe without treatment to kill microorganisms. It was assumed that passage through soil would filter out pathogens. Diseases such as cholera, typhoid, dysentery and hepatitis have been linked to drinking water contaminated by human waste (Eubank et al., 1995). This phenomenon has proved to be the greatest threat worldwide and accounted for 70-80% of health problem in the developing world. However, majority of waterborne diseases have resulted from use of untreated ground

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water that has become contaminated (Joan et al., 1998). Increase in bacterial pollution of ground water quality has been attributed to pollution from septic tank dug near bore holes and wells (Chitanand et al., 2008). Despite the fact that ground water is one of the major sources of water supply for majority of Nigeria, there is no integrated ground water quality monitoring scheme in Nigeria (Adebola et al., 2013). Hence, there must be a critical need for an integrated, national initiative on the microbial quality of water and on risk assessment as related to public health (Joan et al., 1998). This according to them is that water quality problem-solving should address entire watershed and aquifers.

There are several studies that confirmed microbial pollution of ground water quality in Nigeria. Nola et al. (1998) identified indicators of fecal contamination such as *Pseudomona aeruginosa, Aeromonas hydrophila* in spring and well water. Also, in another study by Potgieter et al. (2006), total coliforms (TC), fecal coliforms (FC), fecal entercocci and *Clostridium perfringens* identified exceeded recommended guidelines limit. The results of ground water quality analysis carried out in Abeokuta, Nigeria showed that ground water is grossly polluted with microorganisms and heavy metals (Joan et al., 1998). The results of physiochemical analysis of ground water quality (Osibanjo, 1994), as cited in Nubi and Ajuonu (2011), showed that total solid and total hardness exceeded allowable limit. Similar results were obtained at solid waste dumpsites in 9 locations in Enugu (Chima et al., 2009). Study revealed that more than 65% of the population in the study area use water from ground water with little or no treatment. Also in Enugu, blood samples of 240 people, comprising children, pregnant women and nursing mothers and men in this area, had increase heavy metals bioaccumulation. And this was linked to water consumption (Ibeto and Okoye, 2010).

The increase in water pollution in Enugu and attendant shortage of water supply (Onuigbo and Madu, 2013; Onuigbo et al., 2013; Chima et al., 2009; Utomi et al., 2012 and Ezenwaji, 2008) has continued to grow unabated. Water supply shortages in Enugu urban area is attributed to poor quality of water supply and increase in population density which place high demand on ground water supply. In Enugu, soak away pits are usually located close to hand dug wells where access to water is guaranteed. Ground water from hand dug wells is the major source of water supply in the study area. Enugu and its surrounding areas are known to have abundance source of underground water resources. Ground water resources abound in Enugu, 9th mile Ngwo, Ajali Owa in Ezeagu and Oji river axis all in Enugu state. Numerous industries such as Nigeria brewery, Guinness, cocoa-cola, paints and plastic industries attracted to these areas were majorly because of availability of ground water resources. The increase in number of industrial location, population density and various socio-economic activities which place high demand on water in this area make aquiferous zone vulnerable to contamination.

Digging of hand dug well in every household in Enugu urban has become a general practice. Water from wells constitutes a major source of microbial pollution. Bacteria such as *Vibrio cholera, Salmonella typhi, Salmonella paratyphi, Shigella* spp., *Enterotoxigenic Escherichia coli, Salmonella* spp., *Campylobacter* spp. are able to travel unrestricted through the subsoil (Sugden, 2006). If time taken for microorganisms to be transferred from soak away pit to the zone of water table is long, the microorganisms will die off before getting to water point and will no longer be health threat. In ground water, some viruses are known to survive for up to 150 days; *E. coli* indicator bacteria, survive for up to 42 days while some *salmonella spp* have been shown to persist for up to 42 days (ARGOSS, 1991). The further water containing bacterial loads has to travel to the zone of aquifer, the more complex its path, the lesser the organisms it will hold. Longer distance and time allow for higher numbers of microorganisms to die off naturally. When contaminants move through soil and fractured rock, they generally follow the flow of the groundwater. Therefore, it is important to always take the groundwater flow pattern into consideration when siting a well.

This study was conducted to assess the impacts of soak away pits on hand dug wells in relation to siting distance and season in Enugu urban area. The results of this work will provide basic information and source of the scientific knowledge on hand dug wells water in Enugu and other similar water where little or no information is available for effective management and sustainability. It will also help to form baseline data for users of hand dug wells water; help researchers and stake holders in water resources to prevent future deterioration of hand dug well water quality, and for future study.

MATERIALS AND METHODS

Study area

Enugu urban is located within latitude 06°21'N and 06°30'N and longitude 07°26'E and 07°37'N. It is situated at the foot of Udi escarpment and covers an approximate area of 145 sq km. Enugu is commonly referred to as coal city (Newcastle of Nigeria) because of abundant of coal deposit in the area. It is also known as civil service and educationally base city; though industrial and commercial activities are now dominating in major parts of the area. Enugu urban area drains by rivers such as Asata, Ekulu, Akwata, Ogbete, Emene and its tributaries; and they are captured by Nyaba River and drain into Cross River basin. Dry and rainy seasons occur between November to March and April to October, respectively. It lies in transitional-savanna region; and this is derived from prolonged cultivation. The population of Enugu urban according to 2006 National Population Census was 722, 644, (National Population Commission), (NPC, 2007). Using 2.8% annual rate of increase, the populations' projections of people that consume water in the sample areas is about 679, 043.

Sample selection

Enugu urban area is made of three local government areas namely,

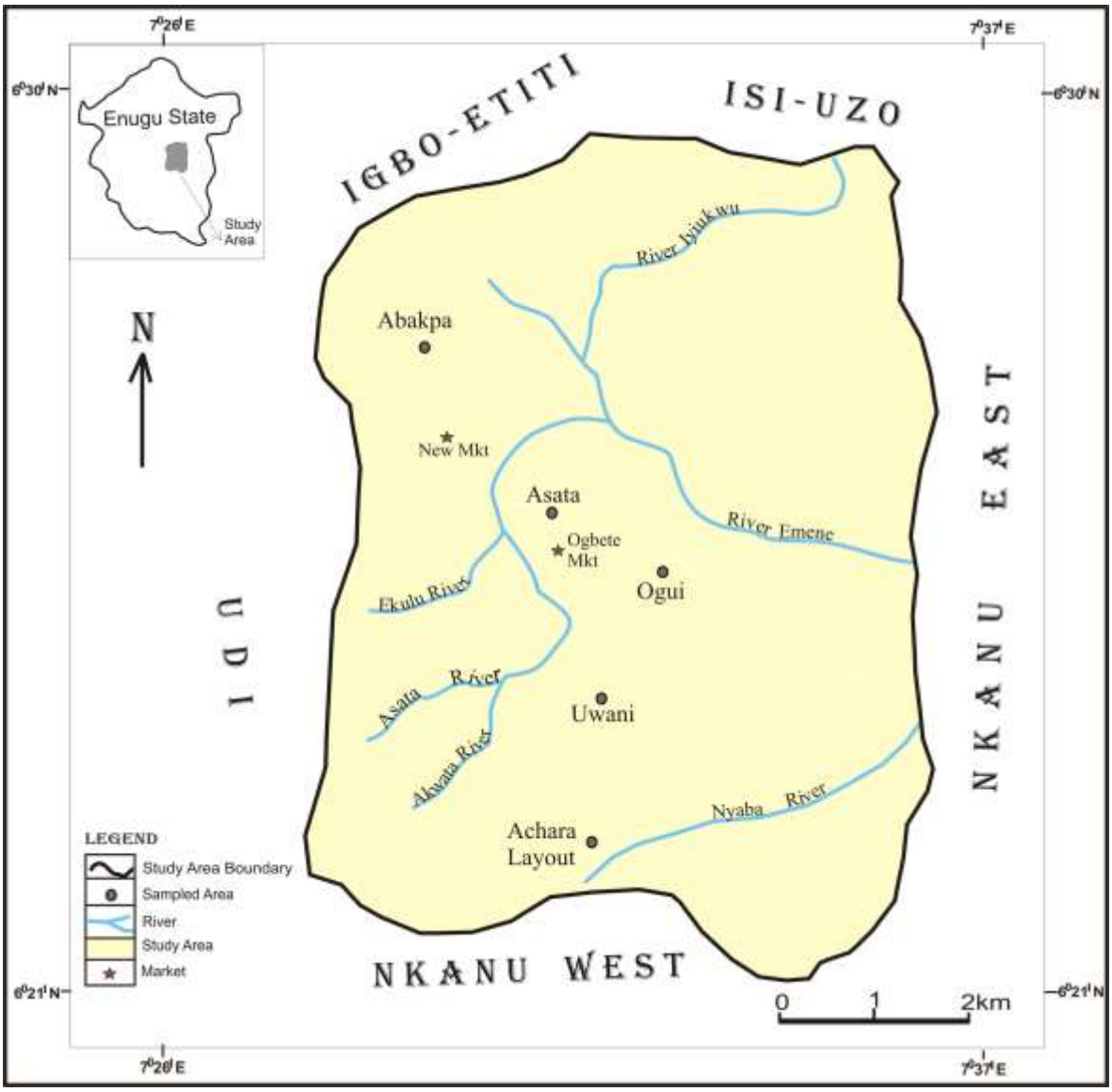

Figure 1. Map of Enugu urban showing sample area (Adapted from Map of Enugu Urban: Ministry of Land and survey, Enugu, 2015).

Enugu East, Enugu North and Enugu South. Five (5) sample areas were randomly selected for the study (Figure 1). Two sample areas were selected from two local government areas, while one area was selected from one local government area. The sample areas are Asata, Ogui (Enugu North), Uwani, Achara Layout (Enugu South) and Abakpa in Enugu East. The choice of selecting the areas was because more than 80% of water supply in the area is source from ground water sources (hand-dug wells). A total of ten (10) hand-dug wells were selected for the study. Two hand dug wells were selected from each of the five sample areas. Five (5) hand-dug wells dug within 30 m away from soak away pits were selected, while five (5) wells dug more than 30 m away from pollution sources were selected as control. The choice of distance was based on the standard distance for digging of soak away pit away from hand dug wells as described by Aguwamba (2001). The standard distance according to him shows that soak away pit should be dug 30 m away from hand dug well or borehole, 15 m for any surface waters meant for domestic uses and 3 m for dwelling or property lines. Also, the Sphere Project (2011), recommended 30 m as a minimum standard for the lateral distance between on-site sanitation systems and water sources. This was used to determine the influence of distance of siting soak away pits and bacterial pollution of hand dug wells in Enugu urban.

Sample collection

Samples were taken from 10 hand-dug wells whose depth varied from 30 to 50 m (98-164 ft) during rainy (June and July) and dry seasons (January and February). This was used to determine seasonal variations of bacterial pollution during the two seasons. They were taken in the morning before users start drawing water from the wells. This was to avoid contamination from other sources and also to make sure that water drawn in the previous day was

Table 1. Distribution of indicators organisms (100 ml) in samples collected during rainy and dry seasons.

Well	Distance (meter)	Rainy season			Dry season			Mean	
		Total coliform count	Fecal coliform	*E. coli*	Total coliform count	Fecal coliform	*E. coli*	Rainy season	Dry season
W_1	26	210	105	70	78	50	15	128.3	47.6
W_2	28	113	83	64	59	47	25	86.6	43.7
W_3	29	100	75	57	47	11	9	77.3	22.3
W_4	22	221	120	108	82	59	31	150	57.3
W_5	29	105	78	42	100	31	12	75	47.7
W_6c	72	55	35	30	2	Nil	Nil	40	0.7
W_7c	63	67	50	45	2	1	0.5	54	1.2
W_8c	85	40	23	12	Nil	Nil	Nil	25	0.0
W_9c	48	60	55	50	3	2	1	55	2
$W_{10}c$	45	88	65	58	12	4	2.5	70	6.2
WHO		3/100 ml	0/100 ml	0/100 ml	3/100 ml	0/100 ml	0/100 ml	-	-

W_1 - $W_{10}c$ = Well samples (W_6c - $W_{10}c$ = wells used as control).

replenished in the wells. A total of twenty (20) samples were taken in all; two (2) samples from each well. Ten (10) samples were used to determine bacterial loads, while ten (10) samples were used for physicochemical characteristics of hand dug well water quality. Samples for bacterial analysis were collected with white bottles that were sterilized in an autoclave at 121°C for 15 min (Sharma, 2009), while samples for physicochemical analysis were collected in clean sterile screw capped polypropylene bottles.

Sample collection from hand dug well was done according to guidelines outlined by Federal Ministry of Water Resources, (2004). A capped sterile bottle covered with a sterile handkerchief was used, with a piece of string, and a clean weight attached to the sample bottle. A 50 m length of clean string was rolled around a stick and tied to the sample bottle. The sample bottle was lowered down, and was not allowed to touch the sides of the well. The bottle was completely immersed in water and submerged below the water, without hitting the bottom or disturbing any sediment. The string was unwind immediately the bottle was filled up. For bacterial analysis, some water was discarded to provide for air space and it was then capped with stopper. All samples were properly labeled with marker, and they had information on i.) sample site/well, ii.) date of sample collection, iii.) time of sample collection and iv.) purpose of sample collection. Samples were put in ice packed plastic boxes. The boxes protected samples from sunlight, prevented breakage of sample bottles. Samples were transported to Water Quality Laboratory and analyses were done within two hours of samples collection.

Sample analysis

Samples were subjected to bacterial analysis at Water Quality Laboratory, Federal Ministry of Water Resources, Enugu. Chemicals and reagents used were of analytical grade. Membrane filtration (MF) method was used to enumerate total coliform, fecal coliform and *E. coli*, and they were expressed in Cfu/100 ml (Sharma, 2009). The types of bacterial were determined by serial dilution and plating of samples on differential culture media. The isolates were identified and characterized accordingly. Temperature and pH were determined *in situ* using a portable pH digital meter. Biological oxygen demand (BOD), chemical oxygen demand (COD) and dissolved oxygen (DO) were analyzed using digital titrator.

Statistical analysis

Data were analyzed for mean and standard deviation. Difference in parameter was tested for statistical difference at $p< 0.05$ using student's t-test. All the analyses were done using the statistical package service solution (SPSS) version 21 (SPSS Inc., Chicago Il., USA). Values were compared with water quality standard set by World Health Organization (WHO) and Nigerian Standard for Drinking Water Quality (NSDWQ).

RESULTS AND DISCUSSION

The results of bacterial analysis of sample collected from ten (10) hand-dug wells (W_1-W_{10}) are shown in Table 1. The results showed that water sample from W_1 dug at a distance of 26 m away from soak away pit recorded values of 210/100 ml (TC), 105/100 ml (FC) and 70/100 ml (*E. coli*). The values were lower than result of sample collected from W_4 dug at a decreased distance of 22 m. The results showed that TC, FC and *E. coli* have values of 221/100, 120/100 and 108/100 ml, respectively. These have shown that distance and season were the determining factors of bacterial pollution of ground water quality (Table 1).

Also, the results of sample from W_2, W_3 and W_5 have similar lower values because of increasing distance. The least values of 57/100 ml (*E. coli*) were obtained at W_3 at a distance of 29 m. The values were lower than values of 108/100 ml of *E. coli* obtained at W_4 at a distance of 22 m. The results of sample from W_3 and W_5 dug at the same distance have similar values indicating that distance from pollution sources was a determining factor of bacterial pollution of ground water quality in Enugu. The results showed that TC, FC and *E. coli* have values of 100/100, 75/100 and 57/100 ml; and 105/100, 78/100 and 42/100 ml, respectively. The highest values of TC

(88/100 ml), FC (65/100 ml) and *E. coli* (58/100 ml) obtained at control of $W_{10}c$ showed that it was less than the lowest values of TC (100/100 ml), FC (75/100 ml) and *E. coli* (57/100 ml) obtained at W_3 dug within 29 m away from soak away pits.

This result was in accordance with earlier work done by Idika et al. (2005), Opara et al. (2011) and Chitanand et al. (2008). Their results showed that location of wells very close to soak away pits led to bacterial pollution of wells. Results obtained by Opara et al. (2011) indicated that more than 90% of wells located within 27 m away from soak away pits were microbiologically polluted. Chitanand et al. (2008), confirmed that proximity of contaminating sources determine the presence and transportation of TC into ground water. This corroborated that distance of locating soak away pits influenced bacterial pollution of hand dug wells located close to it. Table 1 shows that TC, FC and *E. coli* have increased and decreased values in W_1-W_5 dug less than 30 m away from soak away pits, and W_6c - $W_{10}c$ (control) dug more than 30 m away from soak away pits, respectively. This indicated that the further the distance to pollution sources, the lesser the amount of bacterial pollution to hand dug wells (Adekunle et al., 2007). These results were in consistent with results obtained by Adekunle et al. (2007) in Igbora, Nigeria. Their results showed that proximity of hand dug wells to pollution sources and season of the year influenced the concentration of TC and FC.

The results of samples collected during rainy season recorded higher values than results of samples collected during dry season. The mean values of W_1-W_5 during rainy season decreased from 128.3, 86.6, 77.3, 150 and 75; to 47.6, 43.7, 22.3, 57.3 and 47.7 during dry season respectively. The same decreasing values were observed in the control (W_6c-$W_{10}c$). The increase in values of bacterial pollution of ground water quality during rainy season was similar and in line with research work done by Awe et al. (2012) and Adekunle et al. (2007). Their work confirmed increase in bacterial pollution during rainy season than dry season. Increase rainfall resulted to increase in water table and this act as a good medium of microbial transfer from soak away pits to hand dug wells close to it. Also, increase use of water for servicing of toilets during rainy season when compare to dry season was also observed during the study.

All these resulted to increase in bacterial pollution of wells during rainy season as was proved by Hill et al. (2006). Decreasing values of 221/100 to 82/100 ml (TC) was recorded from sample collected from W_4 during rainy and dry seasons, respectively, at the same distance of 22 m. The values of TC from W_3 and W_5 at the same distance of 29 m decreased from 100/100 to 47/100 ml and from 105/100 to 100/100 ml in the rainy and dry seasons, respectively. The decreasing values have shown that bacterial pollution of ground water quality was higher during the rainy season than the dry season. The results were consistent and similar to earlier work done by Adekunle et al. (2007) in Igbora, Oyo state, Nigeria.

The results of bacterial analysis showed that W_8c and W_6c were free from microorganisms during dry season. The two wells met the standard requirement of 3/100 (TC), 0/100 (FC) and 0/100 ml (*E. coli*) set by NSDWQ and WHO for drinking, irrigation and commercial activities. W_8c and W_6c in the control showed that there were negligible values of TC, FC and *E. coli* and FC and *E. coli* during dry season, respectively. Interestingly, the two wells were safe from bacterial pollution and this qualified them to be used for domestic, agricultural and commercial activities during dry season only. The values of TC and *E. coli* from the two wells (W_8c and W_6c) were in contrast with the values obtained by Onuigbo and Madu (2013) in surface water of Emene River in Enugu. Results of W_1-W_5, and W_7c, W_9c, and $W_{10}c$ in all the seasons exceeded the maximum allowable limit. Their values were similar and in accordance with values of TC and *E. coli* obtained by Onuigbo and Madu (2013) in Enugu, Nigeria.

The results of physicochemical analysis of water sample collected from W_1-$W_{10}c$ during the study are shown in Table 2. The results showed that the values of temperature, pH and dissolved oxygen which ranged between 10±0.55 to 30±2.67°C, 6.5±0.12 to 8.5±0.25 and 3±1.11 to 8±2.55, respectively, were within the maximum allowable limit as recommended by NSDWQ and WHO. The values of temperature, pH, DO and COD have shown that the physicochemical quality of ground water in the study area was not influence by distance to pollution sources and season of the year. Statistical analysis showed that no significant difference ($p>0.05$) was observed at distance and season of the year. Therefore, the use of distance and season of the year as determinant factors of physiochemical pollution of ground water was not reliable. The results of temperature and pH obtained from this study were similar and consistent with the results obtained by Adefemi (2003) and Ipinmoroti and Oshodi (1993) in Ado Ekiti and Akure, Nigeria, respectively. It was only BOD that has substantial increase in W_1 and W_4; and this may be attributed to other factors not factored in the study.

The values of BOD in W_1-W_5, and W_7c, W_9c and $W_{10}c$ exceeded maximum permissible limit of 10 mg/l during rainy season. It was only values obtained at W_5 (10±0.67), and values obtained in control of W_6c (10±0.32 and 8±2.11), W_7c (10±1.10) and W_8c (8±0.21 and 8±0.01) during dry season, rainy and dry season, dry season and rainy and dry seasons, respectively that fell within the standard limit. This revealed that there was no significance difference ($p>0.05$) in values of variables irrespective of season of the year and distance to pollution sources. The values of chemical oxygen demand also confirmed that distance to pollution sources and season of the year did not influence the pattern of physiochemical pollution of 10 hand dug wells used in this study. The values which ranged between

Table 2. Mean spatial variation of physiochemical analysis of water samples.

Well	Distance (m)	Rainy Season					Dry Season				
		Temperature (°C)	pH	BOD (mg/l)	DO (mg/l)	COD (mg/l)	Temperature (°C)	pH	BOD (mg/l)	DO (mg/l)	COD (mg/l)
W_1	26	21±0.12	6.7±1.01	80±0.78	5±0.22	58±1.01	27±1.02	6.7±0.34	50±1.23	7±1.20	42±2.34
W_2	28	25.8±0.2	7.2±0.11	29±0.32	6±0.55	40±0.45	21±1.12	6.5±0.12	20±0.41	8±0.54	38±0.21
W_3	29	16±1.26	7.8±1.82	22±1.75	3±1.11	37±0.33	23.5±0.2	6.9±0.24	19±0.01	5±0.22	29±0.23
W_4	22	28±2.05	6.6±1.27	95±2.08	6±0.02	60±2.21	20±2.10	7.5±2.11	34±1.07	7±2.3	53±0.12
W_5	29	23±2.34	8.5±0.25	12±1.22	4±2.20	30±0.10	23±0.55	6.8±2.22	10±0.67	6±0.12	27±0.21
W_6c	72	13±0.45	6.9±0.56	10±0.32	4±1.02	25±2.11	15±0.15	7.4±0.10	8±2.11	6±1.22	28±0.05
W_7c	63	20±2.12	7.4±2.05	12±0.55	5±1.12	28±0.21	22±1.11	6.5±0.05	10±1.10	8±1.01	24±0.01
W_8c	85	11±0.07	7.5±0.28	8±0.21	8±2.55	27±2.22	10±0.55	7±2.70	8±0.01	7±0.10	37±2.11
W_9c	48	21±0.87	6.8±0.05	30±1.70	6±0.37	40±0.78	23±0.71	7.7±0.34	21±0.45	7±1.21	27±2.10
$W_{10}c$	45	30±2.67	6.7±2.30	31±0.03	3.4±0.71	48±1.10	22±2.01	6.5±2.29	101±3.40	6±0.75	25±2.77
NSDWQ		30	6.5-8.5	10	<8	20	30	6.5-8.5	10	<8	20
WHO		30	6.5-9.2	10	<10	20	30	6.5-9.2	10	<10	20

25±2.11 and 60±2.21 as it is in Table 2, exceeded the allowable limit in all the seasons.

Spatial distribution of bacterial in the ten hand dug wells (W_1-$W_{10}c$) used in the study showed that there was strong presence of bacterial in all the wells (Table 3). The table shows more concentration of bacterial in the wells dug within 30 m (W_1-W_5) away from soak away pits than those dug 30 m (W_6c-$W_{10}c$) away from pollution sources. The results are confirmation of influence of distance to bacterial pollution of hand dug wells in Enugu urban area. It was only *E. coli* that was found in all the wells, *Salmonella thyphi* and *Vibrio* cholera were found in all the wells except W_6c, W_7c and $W_{10}c$ respectively. *Klebsiella aerogenes, Micrococcus luteus, Enterobacter aerogenes* and *Shigella flexneri* were only present in three wells. The presence of *E. coli* in all the wells was an indication of fecal pollution by soak away pits. The results revealed that waters from wells used in this study were not safe for domestic and commercial uses and this calls for treatment before it will be used for domestic, agricultural and commercial purposes. The presence of bacterial in water from wells used for various purposes in the study area was an indication of possible outbreak of cholera, dysentery, typhoid and yellow fever, if consumption of water from hand dug wells was not discontinued. These diseases according to Jahata et al. (2009), have continued to be major cause of human mortality and morbidity. The presence of bacteria indicate contamination of hand dug well water with fecal waste that may contain other harmful or disease causing organisms, such as viruses, parasites. Drinking water contaminated with these organisms will cause stomach and intestinal illness including diarrhea and nausea, and even lead to death. These effects may be more severe and possibly life threatening for babies, children, the elderly or people with immune deficiencies or other illnesses. Waterborne diseases are caused by pathogenic microorganisms that are transmitted in contaminated fresh water. Infection generally results during bathing, washing, drinking, in the preparation of food, or the consumption of foods that are infected.

Conclusion

The results of this work showed that distance and season of the year strongly influenced the bacterial pollution of hand-dug well water quality used for socio-economic activities in Enugu urban area. Distance and season are two determinant factors of bacterial pollution of hand dug wells by soak away pits in the area. Statistical analysis showed that significant difference ($p<0.05$) was observed at distance and seasons of the year. Results showed that water samples collected from hand dug wells have increased values of TC, FC

Table 3. Spatial distribution of bacterial species in wells used for the study.

S/N	Bacteria	W_1	W_2	W_3	W_4	W_5	W_6c	W_7c	W_8c	W_9c	$W_{10}c$
1	*Escherichia coli*	+	+	+	+	+	+	+	+	+	+
2.	*Vibrio cholera*	+	+	+	+	+	+	+	+	+	-
3.	*Salmonella typhi*	+	+	+	+	+	-	-	+	+	+
4.	*Salmonella paratyphi*	+	+	+	+	+	+	-	+	-	-
5.	*Stereptococcus feacalis*	+	+	+	-	+	-	+	+	-	-
6.	*Enterococcus faecalis*	+	-	+	-	+	-	-	+	+	-
7.	*Pseudomonas aeruginosa*	+	-	-	+	+	-	-	-	+	+
8.	*Shigella flexneri*	-	+	-	+	-	+	+	-	-	+
9.	*Staphylococcus aureus*	+	+	-	+	-	-	-	-	+	+
10.	*Proteus mirabilis*	-	+	-	+	-	+	-	-	-	+
11.	*Vibrio parahaemolyticus*	+	-	-	+	-	-	+	-	+	-
12.	*Enterobacter aerogenes*	-	+	-	-	-	+	-	-	+	-
13.	*Klebsiella aerogenes*	-	-	+	+	+	-	-	-	-	-
14.	*Micrococcus luteus*	-	+	-	-	+	-	+	-	-	-
15.	*Shigella flexneri*	-	-	+	+	-	-	-	-	-	+

(+) Positive; (-) negative.

and *E. coli* at wells dug closer to soak away pits than wells dug further away from pollution sources; and during rainy season than during dry season. Contrastingly, the results of physiochemical quality of ground water were not influenced by distance to soak away pits and season of the year. Analysis showed that no significant difference ($p>0.05$) was observed in distance and season of the year. The results of this research work have shown that hand dug wells in Asata, Ogui, Uwani, Achara Layout and Abakpa areas are no longer potable for domestic consumption.

Policy implications of the study

The following recommendations if implemented will ensure that potable water is available for the residents of the study area:

1. Treatment: Water from wells in the study area should be adequately treated so as to meet NSDWQ and WHO permissible limit.
2. Alternative sources of water supply: This should be urgently provided by government, non-governmental organization and other stake holders in water resources. Alternative sources of water supply such as bore-hole, piped water should be established in flash point areas so as to curtail this ugly trend.
3. Review: There should be upward review of standard distance for digging of soak away pits away from hand dug wells since most of the wells dug 30 m away from soak away pits as recommended distance have high presence of microorganism and were confirmed to be polluted and unhealthy for domestic and other socio-economic uses. Data obtained from the review should be used to set new specification and standards for siting of soak away pits away from hand dug well water resources in a localized areas.
4. Awareness: This should be created in the areas to inform users of water abstracted from hand dug wells on the increased bacterial pollution of well waters used in the area. Users of ground water should be alerted on the lurking dangers of continued use of water from hand dug wells in the study area. This should be done through women organizations, town hall meeting, focused group discussion, posters, hand bills, public and private schools, hospitals and market places.
5. Geology: Ground water flow and the structure of underlying rocks should be thoroughly investigated before selecting site for digging of hand dug wells and soak away pits in Enugu. The configuration and structure of geological formation of Enugu urban area should be investigated so as to spot out areas with aquifer formations. Areas with abundance ground water resources should be mapped out and detailed geologic studies carried out and local specification used in area with localized rock and soil structure. This is due to the fact that the structure of underlying rocks are determining factors of direction of ground water flow, the medium of transport of microorganisms. Some bacteria and virus have potentials to travel unrestricted under the subsoil. Better assessment of ground water flow condition will enable identification of dominant contaminants sources.
6. Structural modification of hand-dug wells: Wall of the hand-dug wells where ground water seeps into the well should be protected with minute sized stones. The stones will help to trap and sieve out microorganisms from entering into well water.
7. Specifications and guidelines: Land developers such as engineers, surveyors, architects, estate managers,

quantity surveyors and allied workers should be given specifications and guidelines. This will enable them to build houses and its components that will be in conformity with local, regional and international regulations on standard distance of siting of soak away pits away from ground and surface water resources, dwelling places, recreational centers and property lines in all its ramification.

8. Enactment of enabling laws: Enabling laws should be made by legislatures to prevent outbreak of disease that emerge from water pollution and to protect people's life and fragile environmental resources. It will also help to enhance water resources utilization and management, promote and protect urban and regional town planning functionality and ecological and aesthetic functions of the environment.

9. Routine inspection and law enforcement: Sanitary inspection officers should be employed to carry out routine inspection of housing and housing patterns that are built in violation of environmental and building standards. Housing patterns that are not in compliance with standards should be redesigned in order to comply with the standards. Health challenges associated with consumption of ground waters that are polluted with bacterial invasion will be reduced. Also they should enforce laws and regulations, prosecute law violators so that hygienic and sanitary conditions of dwellers will be optimally enhanced.

CONFLICT OF INTERESTS

The authors have not declared any conflict of interests.

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Full Length Research Paper

Kinetic analysis of anaerobic sequencing batch reactor for the treatment of tannery wastewater

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A pilot scale anaerobic sequencing batch reactor (ASBR) was operated at different organic loading rate (1.03, 1.23, 1.52 and 2.21 kg.m^{-3}.d^{-1}) in order to determine the chemical oxygen demand (COD) removal and methane production kinetic models. The system was operated at mesophilic temperature. The wastewater was fed using submersible pump in every twenty four hours and agitated with hydraulic pump for fifteen minutes in every one hour. The COD removal efficiencies was found to be between 69-85% and the methane yield was also between 0.17±0.2 and 0.30±0.02 m^3/kg COD removed. In the kinetic studies, modified Stover-Kincannon and second-order models were found to be the most appropriate model for ASBR treating tannery wastewater than first order model. The saturation value constant and maximum COD removal rate found in Stover-Kincannon model were 5.57 and 5.56 kg of COD m^{-3}.d^{-1}, respectively. The kinetic studies of volumetric methane production showed that Michaelis-Menten model was found to be capable of predicting the volumetric methane production in ASBR that treat tanney wastewater.

Key words: Anaerobic sequencing batch reactor (ASBR), chemical oxygen demand (COD) removal, Michaelis-Menten, second order, Stover-Kincannon.

INTRODUCTION

Tanning is almost a wet process that uses about 30 to 40 L of water/kg of hides or skin processed and also discharges about 90% of the consumed water as wastewater (IFC, 2007). The wastewaters, which are discharged without proper treatment, would contaminate surface and ground water as well as soils. A tanning industry can cause groundwater pollution of about 7 to 8 km radius (Mondal et al., 2005). Currently, there are more than 30 tanneries under operation in Ethiopia. These tanning industries generate 11,312 m^3 wastewater per day and disposed to the surrounding water bodies without proper treatments (LIDI, 2010). However, it is characterized by a high load of pollutants which require proper treatment before it would be discharged into the receiving water body. The treatment systems adopted by most industries are frequently considered as

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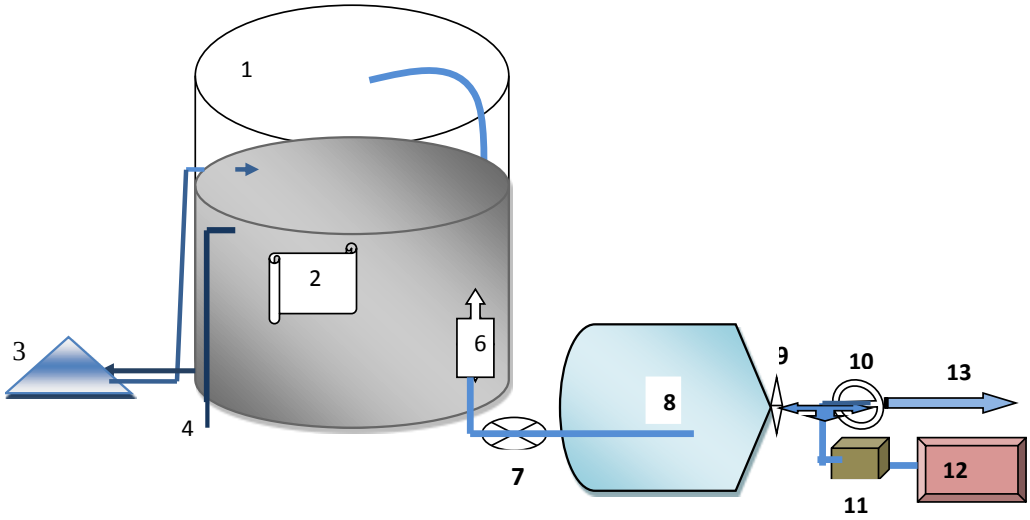


Figure 1. Schematic diagram of the pilot scale anaerobic sequencing batch reactor. ASBR (1); control panel (2); mixing chamber (3); feeding pipe (4); gas pipe (5); gas flow meter (6); moisture trap unit (7); biogas storage bag (8): gas valve (9); gas blower (10); sulfur scrubber (11); generator (12) and gas line to the kitchen (13).

regulatory obligation that increase capital and operational costs and ultimately yield negative economic returns. Compliance to environmental legislations should not necessary lead to the creation of additional costs. It should instead provide a secondary source of income. Anaerobic treatment is considered as sustainable method of reducing pollution from domestic, agricultural and industrial operations (Seghezzo et al., 1998; William and David, 1999). It consumes little energy as no aeration is needed and produces renewable energy in the form of biogas and nutrient rich digestate (Kaparaju and Rintala, 2011). Beside the supply of energy and manure, the anaerobic technology provides the opportunity for the reduction of greenhouse gas emission and mitigation of global warming through substitution of fossil fuel for energy production and chemical fertilizer (Pathak et al., 2009). The total output in CO_2 equivalents might be reduced from 2.4 kg CO_2-eq/kg COD removed for fully aerobic treatment to 1.0 kg for primarily anaerobic processes (Insam and Wett, 2008). In fully aerobic system, up to 1.4 kg CO_2/COD removed is generated from electric power generation, whereas the carbon dioxide generated in the anaerobic processes is greenhouse gas neutral (Insam and Wett, 2008; Kaparaju and Rintala, 2011).

The anaerobic sequencing batch reactor (ASBR) operates in a cyclic batch mode with four distinct phases per cycle. The four phases are: filling, reacting, settling and release (Timur and Oèzturkm, 1999; Zhang et al., 1997). The ASBR systems has been successfully applied in laboratory and pilot scales for treatment of high strength wastewaters including landfill leachate, slaughterhouse wastewater, municipal sludge and dairy wastewaters, and brewery wastewater (Xiangwen et al., 2008).

Mathematical description of biological treatment processes has been performed using process kinetics model. The understanding of process kinetics is important for the design of specific unit and operation of treatment systems, predicting system stability, effluent quality and waste stabilization. The knowledge on kinetics model leads to optimization of performance, a more stable operation and a better control of wastewater treatment operations. There are numerous mathematical models in the literature for describing anaerobic processes such as first order model, second-order, Grau model, Modified Stover-Kincannon, Sundstorm model, Contois model, Chen, Michaelis-Menten type kinetic model, etc (Jafarzadeh et al., 2009; Sandhya and Swaminathan, 2006; Buyukkamaci and Filibeli, 2002). In this study, different mathematical models such as first order model, second order Grau model, Modified Stover-Kincannon, Michaelis-Menten type kinetic model were applied to data obtained from the ASBR operation and kinetic coefficient for the reactors were calculated.

MATERIALS AND METHODS

Experimental set up

The pilot-scale anaerobic SBR was constructed using concrete materials in a cylindrical in shape with a dimension of 4 m in height and 4 m in diameter. Figure 1 shows the schematic diagram of the pilot scale ASBR and the accessories. The total volume was 100 m^3

Table 1. Characteristics of tannery wastewater used at five different OLR.

Parameter	Phase I	Phase II	Phase III	Phase IV
pH	9.64 ±0.46	9.20 ±0.33	9.28 ± 0.311	9.09±0.49
E.C. (mS)	8.76±0.40	8.46±0.39	8.08±0.38	8.43±0.72
TDS (g/l)	7.49±0.36	7.24±0.44	6.81±0.42	7.26±0.68
Salinity (g/l)	9.26±0.57	9.19±0.38	8.91±0.33	9.07±0.71
COD (mg/l)	4221±359	4265± 215	4586± 292	4458± 396
TN (mg/l)	451±47.5	517±112	492.5±89.9	458±58.6
NH_4^+-N (mg/l)	231±45	270±66	255±58	248±44.46
Total phosphorus (mg/l)	22.2±6.8	18 ±4.5	19.3 ±4.16	23.5 ±6.5
Sulfide (mg/l)	93±22.27	126±38.9	123.5±33.8	117.5±29.4
Sulfate (mg/l)	470±75.	390±76.9	520±99.13	469±69

of this 80 m^3 as working volume and the remaining volume for head space. The internal part of the digester was insulated with plastic foam and covered with geo-membrane. Stainless steel tubes were installed 30 cm above the bottom of the digester surface for the circulation of hot water. The top of the digester has two holes. One of the holes was fitted with PVC pipe for biogas outlet to gas flow meter and to the gas storage bag. The other one was fitted with stainless steel tubing extended to the bottom of the digester for hot water circulation. The ASBR was operated under mesophilic condition (31°C) and hot water heated with solar panel was used to maintain the temperature. The wastewater in the ASBR was mixed in every hour using hydraulic pump for fifteen minutes during the day time of the operation.

Operation of the ASBR

The performance of the pilot scale ASBR was evaluated at four different OLRs. The performance of the ASBR was monitored by measuring COD removal efficiency and the biogas production and quality. During the first phase, the reactor was operated at the organic loading rate (OLR) of 1.03 $kg.m^{-3}.d^{-1}$ and constant hydraulic retention time of 4 days. In the second phase, the OLR was increased from 1.03 to 1.23 $kg.m^{-3}.d^{-1}$ by increasing the volume of wastewater from 20 to 23 m^3 and HRT was 3.5 days. In the third phase, the reactor was operated at OLR of 1.52 $kg.m^{-3}.d^{-1}$ and HRT of 3 days by increasing the volume of the inlet wastewater from 23 to 26.5 m^3. Finally, it was operated at OLR of 2.21 $kg.m^{-3}.d^{-1}$ and 2 days by increasing the volume of wastewater from 26.5 and 40 m^3.

Physico-chemical analysis

The characteristics of influent and effluents were determined in terms of chemical oxygen demand (COD), total nitrogen (TN), ammonium-nitrogen (NH_4^+-N), nitrate-nitrogen (NO_3^--N), sulphides (S^{2-}), sulphate (SO_4^{2-}), total phosphorous (TP) and orthophosphate (PO_4^{3-}) colorimetrically using spectrophotometer (DR/2010 HACH, Loveland, USA) according to HACH instructions. Total solid and volatile solid were also measured according to the methods described in standard methods of water and wastewater (APHA, 1998). pH of tannery wastewater was measured using a pH meter (CON, 2700). TDS, EC and salinity were measured using TDS/EC/salinity meter. Sodium, potassium, calcium, magnesium, chromium, copper, iron and lead were determined using an atomic absorption spectrometer (novAA, 400P). Total nitrogen content of the wetland plant samples was determined by Kjeldahl method. Percent of removal efficiency (%) for each parameter was determined by the following equation:

$$\mathrm{RE}(\%) = 100 * \left(\frac{S_{In} - S_{out}}{S_{In}}\right)$$

RESULTS AND DISCUSSION

Characteristics of raw tannery wastewater

Tannery wastewater is characterized mostly in terms of the levels of pH, salinity, organic matter (COD), nitrogenous compounds (TN and NH_4^+), suspended solids (SS) and total dissolved solids (TDS), chromium and sulfides (Jahan et al., 2014). However, these parameters vary significantly from tannery to tannery depending on the size of the tannery, chemicals used for a specific process, amount of water used and type of final product produced.

The mean characteristics of raw wastewaters used in the study are presented in Table 1.

The wastewater was characterized as alkaline with pH value ranging from 9.09±0.49 to 9.64±0.46. It also contain high level of electrical conductivity (8.08±0.38 to 8.76±0.34 mS), total dissolved solids (6.81±0.42 to 7.49±0.36 g/l) and salinity content (8.77±0.72 to 9.26±0.51 g/l). This due to the chemicals used in the soaking and beam house operation. It contained high organic matter and nitrogenous compounds with COD ranges from 4221±359 to 4586± 292 mg/l. The influent had high total nitrogen (TN), NH_4^+-N and sulfate values ranges from 451±47.5 to 517±112 mg/l, 231±45 to 270±66 mg/l and 390±76.9 to 520±99.13 mg/l, respectively; likewise, sulfide and phosphate concentrations ranged from 92.9±23.27 to 127±43.3 mg/l and 18±4.5 to 23.5±6.5 mg/l, respectively. Seyoum (2004) reported higher concentration of TN, ammonium and COD in tannery wastewater.

Table 2. Summary of the performance of anaerobic sequencing batch reactor.

Parameters	Phase I	Phase II	Phase III	Phase IV
OLR ($kg.m^{-3}.day^{-1}$)	1.03±0.09	1.23±0.06	1.52±0.1	2.21±0.23
HRT (day)	4	3.5	3	2
COD removal (%)	81±2.1	79±2.3	76±1.6	69±1.7
COD out (mg/l)	791±149.5	898.9± 122	1101.4±123	1358.3±170
Biogas production ($m^3.day^{-1}$)	26.2±1.6	28.1±1.8	31.8±2.7	36.7±2.8
Methane (%)	70±1.6	68±1.7	64 ±3.0	55±1.9

Performance of the ASBR

The COD removal efficiency and biogas production, methane yield and content of the ASBR are summarized in Table 2. During the first phase of the operation, the COD removal efficiency varied in the range of 78-84%. The average COD removal efficiency and mass removal rate in the single feeding mode were 81±2% and 791±149.5 mg/l, respectively.

In the second phase, the COD removal efficiency varied in the range of 76 to 83% with average removal efficiency of 79±2.3%, while the average concentration of COD was 898.9±122 mg/l in the final effluent. In the third phase, the average COD removal efficiency decreased to 74-79% with average COD concentration of 1101.4±123 mg/l. In the final phase, the COD removal efficiency varied between 67 and 72%. The average removal COD removal efficiency and effluent concentration were 69±1.7% with 1358.3±170 mg/l, respectively. The anaerobic digester showed significant variation in COD removal efficiency with variation of organic loading rate (ANOVA, $P<0.05$). The results of this study indicate that COD removal efficiency was highest in the first phase of operation and lowest in the final phase of operation. The final phase showed residual COD 31% from the influent. The COD removal efficiencies obtained in this study were higher than the results recorded by Song et al. (2003) from the treatment of tannery wastewater (COD removal of 60 to 75%) using upflow anaerobic fixed biofilm reactor at varying organic loading rate of 0.16 to 3.14 kg $m^{-3}d^{-1}$ and HRT of 16 days to 1 day. The results obtained in phase II (79±2.3%) are comparable with results reported by Lefebvre et al. (2006) for the treatment of tannery soak liquor (COD removal of 78%) at a HRT of 5 days and an OLR of 0.5 kg $m^{-3}d^{-1}$ using up flow anaerobic sludge blanket bed reactor. Other comparable mean COD removal (78.2%) was reported by EI-Sheikh et al. (2011) in the treatment of tannery wastewater using two stage UASB reactors. On the other hand, Banu and Kaliappan (2007) found slightly higher COD removal efficiency (86% at OLR of 2.74 kg $m^{-3}d^{-1}$ and HRT of 60 h; 88% at OLR of 3.22 kg $m^{-3}d^{-1}$ for 70 h) in the treatment of tannery wastewater using hybrid upflow anaerobic sludge blanket reactor.

Kinetic model substrate removal

First order substrate removal model

The hydrolysis of organic pollutants was described by first order kinetics model. The mass balance equation for the substrate in the anaerobic system can be described as follows:

$$-\frac{\mathrm{ds}}{\mathrm{dt}} = \frac{\mathrm{Q}}{\mathrm{V}}(\mathrm{S}_{o-}\mathrm{S}_{\mathrm{e}}) - K1Se$$

where, So is substrate concentration in the influent (mg/l): Se is substrate concentration in the effluent; Q is flow rate of influent to reactor (l/d); V is effective volume of the reactor and K1 is first-order kinetic constant (per day). Under steady state conditions, $(-\frac{\mathrm{ds}}{\mathrm{dt}})$=0 and the above equation can be represented in the following form:

$$\frac{\mathrm{S}_{0-}Se}{\theta_{\mathrm{H}}} = K1Se$$

where,θ_{H} = hydraulic retention time.

The value of first-order kinetic constant can be obtained by plotting $\frac{\mathrm{S}_{0-}Se}{\theta_{\mathrm{H}}}$ against Se as given in the above equation. The slope of the line in the plot would represent the value of K_1. The first order model for COD removal is drawn in Figure 2. The value of the k was obtained from the slope of the line that was plotted (So-Se)/HRT versus Se. The graph fit a straight line with regression coefficient of (R^2= 0.83).

This indicates that about 17% of the total variations were not explained in the first order regression model. The k value was determined as 0.99 per day. Isik and Sponza (2005) obtained comparable (k=0.93) value with this study from the treatment simulated textile wastewater using UASBR system

Modified Stover-Kincannon Model

The Stover-Kincannon was developed first for rotating biofilm reactor (Sandhye and Swaminathan, 2006). It is

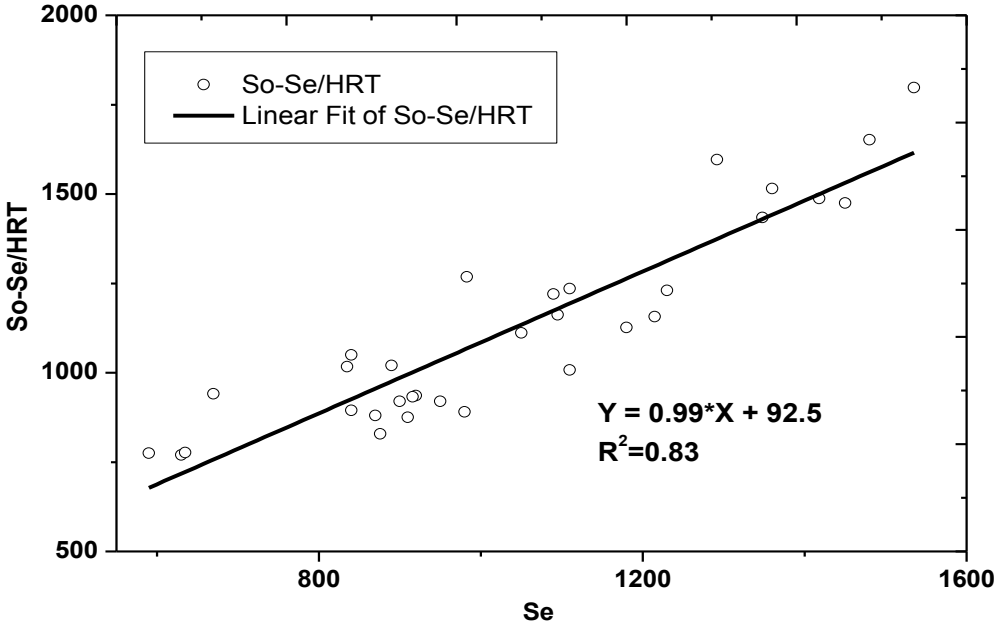


Figure 2. Substrate removal first order model for ASBR.

assumed that the organic loading rate can be correlated with substrate utilization rate using mono-molecular kinetics. The substrate removal rate is defined in two different forms as shown below:

$$\frac{\mathrm{ds}}{\mathrm{dt}} = \frac{\mathrm{Q}}{\mathrm{V}}(\mathrm{S}_{i-}\mathrm{S}_{\mathrm{e}})$$

$$\frac{\mathrm{ds}}{\mathrm{dt}} = \frac{\mathrm{U}_{max}\left(\frac{\mathrm{QS}_i}{\mathrm{V}}\right)}{\mathrm{K}_{\mathrm{B}} + \left(\frac{\mathrm{QS}_i}{\mathrm{V}}\right)}$$

This can be linearized as:

$$\left(\frac{ds}{dt}\right)^{-1} = \frac{V}{q(Si - Se)} = \frac{KB\,V}{Umax\,QSi} + \frac{1}{Umax}$$

Where ds/dt is the substrate removal rate (g/L-day), U_{max} is the maximum utilization rate constant (g/L-day), K_B is saturation value constant (g/L-day), Q is the flow rate (L/day) and V is the effective volume of reactor (L). Since dS/dt approaches U_{max} as the organic loading rate, qSi/V approaches infinity. Figure 3 illustrates the graph drawn between the reciprocal of mass loading removal rate (V/(Q(So-Si) with the reciprocal of OLR to derive the values of Umax and K_B for the anaerobic sequencing batch reactor treating tannery wastewater. The graph fit a straight line with regression coefficient of (R^2= 0.99). This indicates that only less than 1% of the total variations were not explained in the regression model. Hence, the regression coefficient supports strongly the validity of the linearized Stover-Kincannon model. It can be conclude that modified Stover-Kincannon model can be used to describe the performance of mesophilic ASBR treating tannery wastewater in this study. The maximum value for COD removal rate (U_{max}) and saturation constant (K_B) were determined as 5.56 and 5.78 kg of COD m^{-3} d^{-1}, respectively.

The predicted U_{max} was higher than the maximum loading rate (2.21 kg m^{-3} d^{-1}) used in this study. This revealed that ASBR has higher potential in withstanding high strength tannery wastewater. Moreover, the closeness of U_{max} and K_B values obtained indicate that increasing organic loading rates would lead to reduction in the processes efficiency. Senturk et al. (2010) and Ahn and Forster (2000) also made similar conclusion. Table 3 shows the comparison of the value of U_{max} and K_B reported for various types of substrate (wastewater) using the same type of modified Stover-Kincannon model.

The U_{max} and K_B found in this current work were higher than Priya et al. (2009) and Ahn and Forster (2002). They were almost comparable with the values obtained by Kapdan (2005) and Isik and Sponza (2005). On the other hand, the value of both Umax and K_B in this work were lower than that of Senturk et al. (2010), Wanga et al. (2009), Yilmaz et al. (2008) and Sandhye and Swaminathan (2006). The highest U_{max} were obtained for milk permeate waste water in AMBBR system followed by paper mill wastewater, while the lowest was for formaldhyde containing wastewater followed by corrugated paper wastewater. On the other hand, the

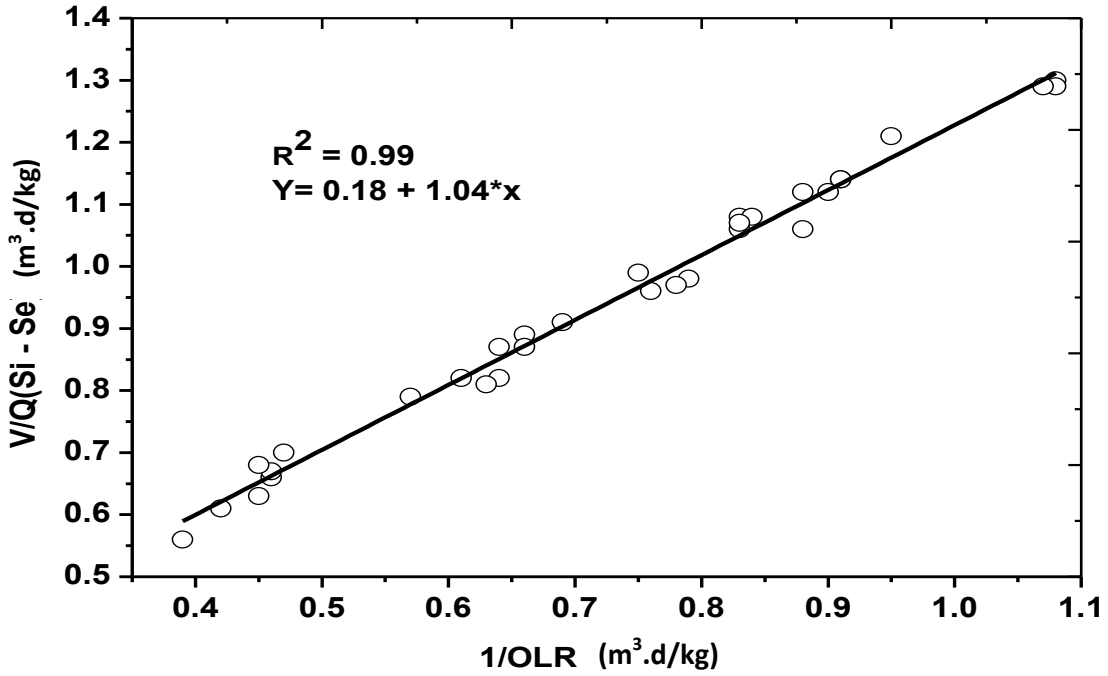


Figure 3. Modified Stover-Kincannon model for ASBR.

Table 3. Comparison of the kinetic parameters obtained for the various substrates.

Types of wastewater	Reactor	U_{max}	K_B	References
Tannery	ASBR	5.56	5.78	This study
Food processing wastewater	Anaerobic contact	22.92	23.59	Senturk et al. (2010)
Formaldhyde containing wastewater	UAFB	3.4	4.6	Priya et al. (2009)
Milk permeate wastewater	AMBBR	89.3	102.3	Wanga et al. (2009)
Paper mill wastewater	AF	86.21	104.15	Yilmaz et al.(2008)
Textile wastewater	UAFB	31.69	45.37	Sandhye and Swaminathan (2006)
Simulated Textile wastewater	UASBR	7.5	8.2	Isik and Sponza (2005)
Synthetic saline wastewater	UASBR	5.3	7.05	Kapdan (2005)
Corrugated paper wastewater	AF	3.86	0.80	Ahn and Forster (2002)

highest K_B was found for paper mill wastewater followed by milk permeates wastewater, while the lowest was from corrugated paper wastewater. The variation of U_{max} and K_B values among different researchers might be attributed to the variation of characteristics wastewater, reactor configuration and microorganisms used in the studies (Priya et al., 2009).

Grau second-order model

The second order model was employed to the experimental results for ASBR system treating tannery wastewater. The general equation of Grau second order kinetics model is shown below:

$$\frac{ds}{dt} = k_2 X \left(\frac{S}{S_o}\right)^2$$

Where, ds/dt is the substrate removal rate (g/L-day), k_2 is kinetic constant (g COD/gVS-day), X is the concentration of microorganisms (gVS/L), S is substrate concentration at any time, and S_0 is the concentration of initial substrate (g/l).

Integrating the above equation within the boundary conditions (S=Si to Se and t=0 toθ_H), and then linearized,

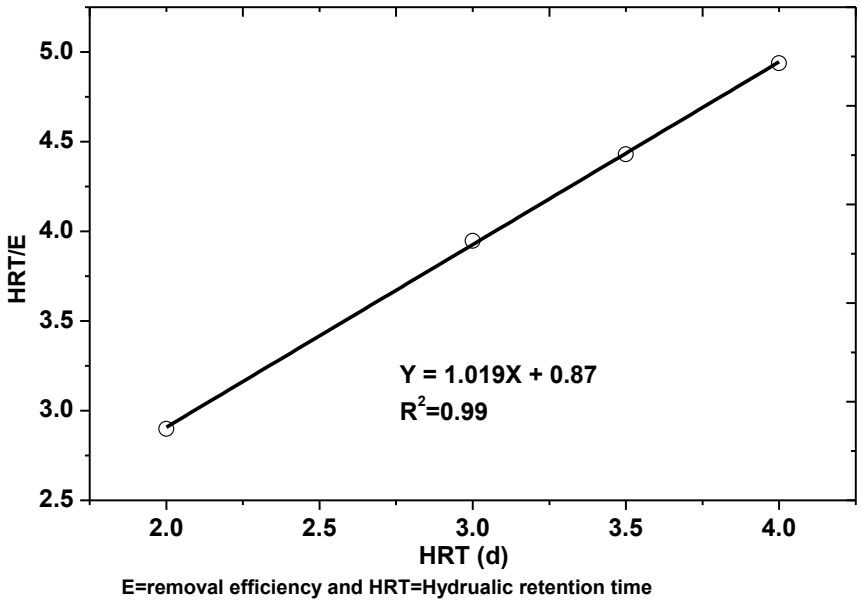


Figure 4. Linear plots of the Grau second-order kinetic model.

the following equation is obtained.

$$\frac{S_0\theta_H}{S_0 - S} = \theta_H + \frac{S_0}{k_2 X}$$

Holding the term "S_0/k_2X" in the above equation constant leads to

$$\frac{S_0\theta_H}{S_0 - S} = a + b\theta_H$$

The term $(S_0\text{-}S_i)/S_0$ is the substrate removal efficiency (E) and it can be used in the equation as shown below:

$$\frac{\theta_H}{E} = a + b\theta_H$$

If V/Q(Si-Se) is plotted against 1/OLR, K_B/U_{max} is the slope and $1/U_{max}$ is the intercept point of the line. Figure 4 shows the plot of the Grau second-order multi-component model for ASBR.

The value of a, and b were determined from the intercept and slope of the straight lines. The values of a and b were obtained to be 0.87 and 1.019, respectively, with high correlation coefficient (R^2=0.99). This confirms the validity of the application of this model for ASBR treating tannery wastewater. Hence, the formula for predicting substrate concentration in the effluent can be given as:

$$S = S_0\left(1 - \frac{\theta}{0.87 + 1.019\theta}\right)$$

These results show that both modified Stover-Kincannon model and Grau second-order can be applied successfully for modeling of the experimental results of ASBR treating tannery wastewater with high correlation coefficient (R^2=0.99). On the other hand, the first order model appeared to be less successful (R^2=0.83) on predicting substrate removal from tannery wastewater in ASBR system.

Kinetics of methane production

The volumetric methane production rate can be obtained through the expression:

$$r_{CH_4} = \frac{q_{CH_4}}{V}$$

where qCH_4 is the daily methane production (m^3/d) and V is the reactor working volume (m^3).

Figure 5 illustrate the graphical estimation of the concentration of non-biodegradable organic matter (TCOD) based on the relationship between ln (TCOD) and 1/(HRT) (Wang et al., 2009). By the least-square fitting of ln (TCOD) and 1/HRT, an intercept of 0.531 g TCOD L^{-1} (regression coefficient 0.95) was calculated, which corresponds to an infinite HRT. Thus, this can be considered to be the amount of non-biodegradable substrate (Borja et al., 2003; Raposo et al., 2004; Rincon et al., 2006; Wang et al., 2009).

Senturk et al. (2010) and Rincon et al. (2006) found

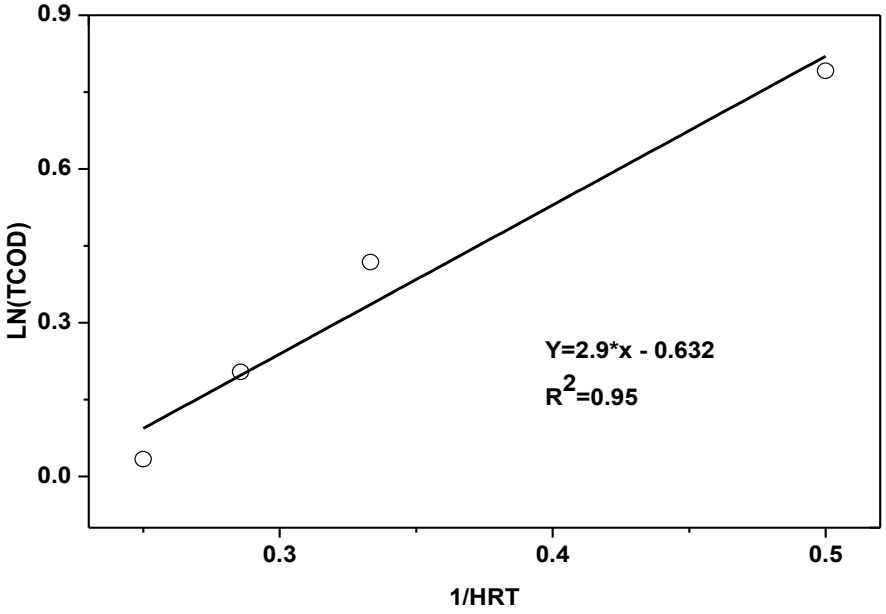
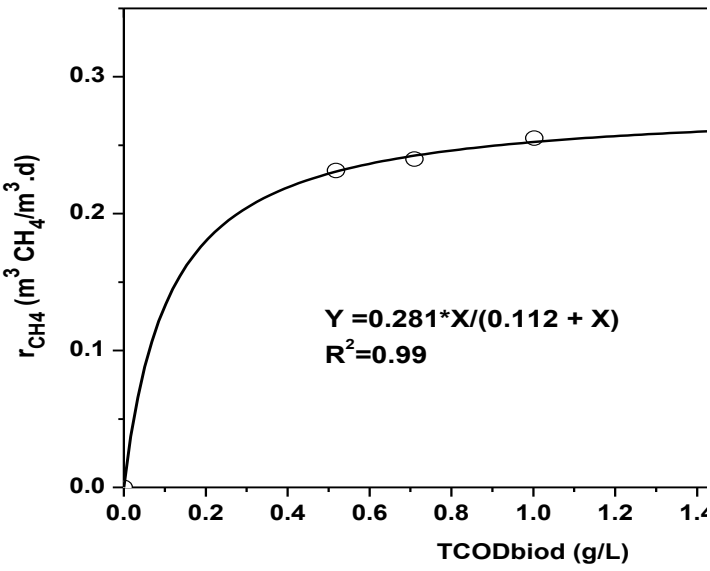


Figure 5. Estimation of the fraction of non-biodegradable soluble organic matter contained in the wastewater used in the study.

Figure 6. Variation of the volumetric methane production rates (r_{CH_4}) as a function of the biodegradable TCOD concentration.

lower non-biodegradable substrate (290 and 92 mg/l) in reactors treating wastewater generated from food processing and protein production from chickpea, respectively. On the other hand, Borja et al. (2003) and Wang et al. (2009) determined higher valu



es of non-biodegradable substrate in the treatment of wastewater generated from Olive Pomace and diary industry, respectively.

The observed values of r_{CH_4} (methane production) plotted as a function of biodegrdabel organic matter concentration (TCOD biod) are shown in Figure 6. As shown in the figure, the observed methane production

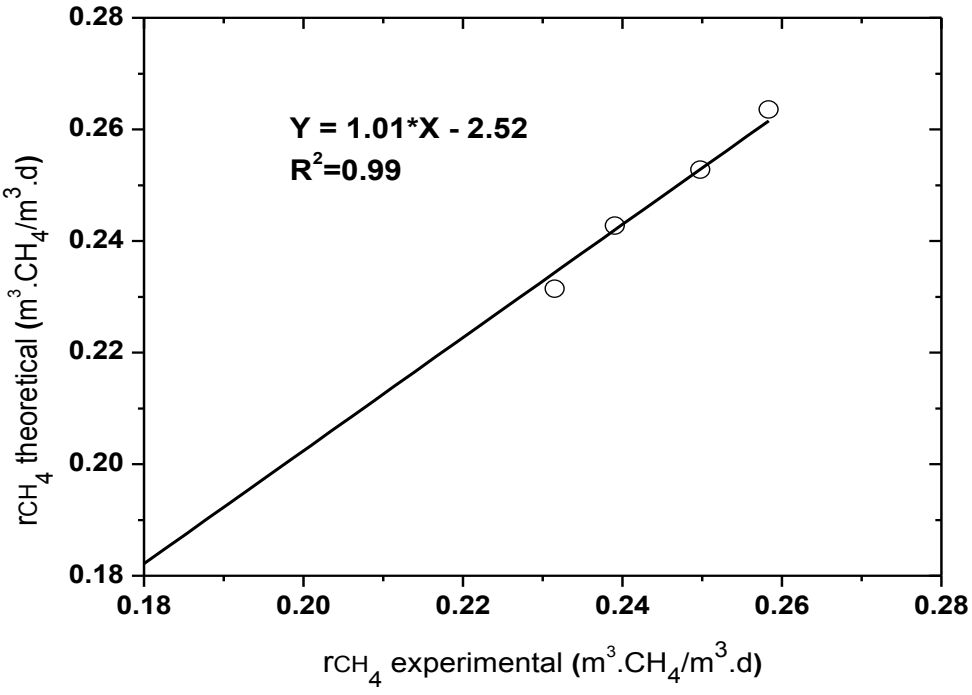


Figure 7. Comparsion between the experimental methane proudction rates and the theoretical value predicted.

values fit Michaelis-Menten type kinetic model, which is a hyperbolic function. By using the origin 8 software, the following kinetic equation was obtained:

$$r_{CH_4} = 0.281TCODb/(TCODb + 0.112)$$

The theortical r_{CH_4}values could be determined using the above equation for the reactor used in this study. The predicted thoertical methane production values were plotted against the observed methane production values. As shown in Figure 7, a linear regression line with a slope of 1.01 and a regression coefficient of the graph 0.99 were obtained. This indicted that the proposed model is capable of predicting the behavior of the ASBR treating tanney wastewater.

Conclusion

The results of this study showed that tannery wastewater could be treated efficiently in anaerobic sequencing batch reactor at different HRT (4, 3.5, 3 and 2 days). The tannery wastewater treatment performance of the ASBR was evaluated at different organic loading rate of 1.03, 1.23, 1.52 and 2.21 kg $m^{-3}.d^{-1}$ and the kinetic analysis for the reactor was performed using the data found in the experiment. The results of the system showed that COD removal efficiency was 69 to 85% and the methane yield was 0.17±0.2 to 0.30±0.02 m^3/kg COD removed. Modified Stover-Kincannon and second-order models were found to be the most suitable model for ASBR (R^2 = 0.99). Michaelis-Menten model was also found to be capable of predicting the volumetric methane production of ASBR that treat tanney wastewater.

CONFLICT OF INTERESTS

The authors have not declared any conflict of interests.

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