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Physicochemical quality of an urban municipal wastewater effluent and its impact on the receiving environment

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Abstract The physicochemical qualities of the final effluents of an urban wastewater treatment plant in South Africa were assessed between August 2007 and July 2008 as well as their impact on the receiving watershed. The pH values across all sampling points ranged between 6.8 and 8.3, while the temperature varied from 18°C to 25°C. Electrical conductivity (EC) of the samples was in the range of 29–1,015 μS/cm, and turbidity varied between 2.7 and 35 NTU. Salinity and total dissolved solids (TDS) varied from 0.36 to 35 psu and 16 to 470 mg/L, respectively. The concentrations of the other physicochemical parameters are as follows: chemical oxygen demand (COD, 48–1,180 mg/L); dissolved oxygen (DO, 3.9–6.6 mg/L); nitrate (0.32–6.5 mg NO_3^- as N/L); nitrite (0.06–2.4 mg NO_2^- as N/L); and phosphate (0.29–0.54 mg PO_4^{3-} as P/L). pH, temperature, EC, turbidity, TDS, DO, and nitrate varied significantly with season and sampling point ($P < 0.05$ and $P < 0.01$, respectively), while salinity varied significantly with sampling point ($P < 0.01$) and COD and nitrite varied significantly with season ($P < 0.05$). Although the treated effluent fell within the recommended water quality standard for pH temperature, TDS, nitrate and nitrite, it fell short of stipulated standards for other parameters. The result generally showed a negative impact of the discharged effluent on the receiving watershed and calls for a regular and consistent monitoring program by the relevant authorities to ensure best practices with regard to treatment and discharge of wastewater into the receiving aquatic milieu in South Africa.

Keywords Physicochemical qualities · Wastewater effluent · Receiving watershed

Introduction

Water forms the backbone of the world's economy and it is critical to the development of all spheres of human endeavor (Obi et al. 2006). It is essential for living systems, industrial processes, agricultural production, and domestic uses (Hu 2009). The quality of water available and accessible to a people has tremendous impact on their living standard and well being; hence, global and local efforts are rife at ensuring adequate provision of clean and safe water to the world's growing population. As a semiarid country, South Africa has a peculiar challenge of meeting her ever-increasing water demand occasioned by industrial and population growth. This has inspired

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the government to set up a "Strategic Framework for Water Services" (DWAF 2003) aimed at ensuring basic water supply (at least 25 L of potable water per capita per day) to all South Africans. As laudable as this program might be, it may create its own challenges; increasing water supply will most likely translate to increase in wastewater output. The implication, therefore, is that additional wastewater output without due diligence of the working efficiency of existing wastewater treatment plants might end up compounding an already bad situation.

There are reports in the literature about the inability of existing wastewater treatment plants in South Africa to adequately treat wastewater effluent prior to discharge into the receiving environment (Morrison et al. 2001; Fatoki et al. 2003). This has seriously compromised the quality of receiving water systems by altering the interrelationship and interactions of parameters that govern the stability of the ecosystem. Physicochemical parameters such as temperature, pH, DO, salinity, and nutrient loads have been reported to influence biochemical reactions within water systems. Such changes in the concentration of these parameters are indicative of changes in the condition of the water system (Hacioglu and Dulger 2009), the consequence of such is the compromise of the water quality for beneficial uses.

Wastewater discharges may contain health-compromising pathogens, carcinogenic substances (e.g., heavy metals, trihalomethanes [THM], etc.), and/or chemical substances which may cause adverse environmental impact such as changes in aquatic habitats and species composition, decrease in biodiversity, impaired use of recreational waters and shellfish harvesting areas, and contaminated drinking water (Environment Canada 2001; CCME 2006). All of these impacts lead to a less valuable environment, poor health, a less prosperous economy, and ultimately, a diminished quality of life (Environment Canada 2001).

Many South Africans live in rural areas and lack potable water supply, thus relying on surface waters that are negatively impacted by untreated or inadequately treated wastewater for their daily subsistence (Pearson and Idema 1998; Mackintosh and Colvin 2002). Furthermore, the environmental implications of inadequately treated effluent may take a serious toll on the socioeconomic status of South Africa as a leading tourist destination in the world. To preserve the health of unsuspecting South Africans and maintain the integrity of the environment, it is imperative to regularly and consistently monitor the quality of municipal wastewater effluent prior to discharge into the receiving environment. In this report, we evaluate the physicochemical quality of the final treated effluent of a typical urban wastewater treatment facility in South Africa and its impact on the receiving environment.

Materials and methods

Plant description

The wastewater treatment plant under study is located in East London, an urban settlement in the Buffalo City municipality of the Eastern Cape Province of South Africa, and situated in the geographical coordinates 32.97° S and 27.87° E. The plant receives domestic and industrial sewage. It is an activated sludge treatment plant comprising four screens, a grit channel, two aerobic tanks, six sedimentation tanks, two anaerobic tanks, and two anoxic tanks. Disinfection of effluent is done by chlorination via a water-pressure-operated, wall-mounted, gas chlorinator in a baffled reinforced concrete contact tank. The final effluent is discharged into the Indian Ocean between Nahoon and Eastern Beach at Bats cave. The average daily inflow during the period of study was 32,000 m^3/day, while the plant has a built-in capacity of 40,000 m^3/day.

Sample collection

Wastewater samples were collected on a monthly basis from the final treated effluent (FE), discharge point (DP), 500 m upstream (UP), and 500 m downstream (DW) of the discharge point between August 2007 and July 2008. Samples were collected in duplicates in 1-L Nalgene bottles previously cleaned by washing in nonionic detergent, rinsed with tap water, later soaked in 10% HNO_3 for 24 h, and finally, rinsed with deionized water prior to usage. During sampling, sample bottles were rinsed three times with sampled water before filling the bottles to the brim at depths of 1 m below the surface of each designated sampling point. Samples were then transported in cooler boxes containing ice packs to the Applied and Environmental Microbiology Research Group (AEMREG) laboratory at the University of Fort Hare, Alice, South Africa for analyses. Samples were processed within 6 h of sample collection.

Physicochemical analysis

All field meters and equipment were checked and appropriately calibrated according to the manufacturers' instructions. pH, temperature, electrical conductivity (EC), salinity, total dissolved solids (TDS), and dissolved oxygen (DO) were all determined on site using the multiparameter ion-specific meter (Hanna-BDH laboratory supplies). Turbidity was also determined on site using a microprocessor turbidity meter (HACH Company, model 2100P). The concentrations of orthophosphate as P, nitrate, nitrite, and chemical oxygen demand (COD) were determined in the laboratory by the standard photometric method (DWAF 1992) using the spectroquant NOVA 60 photometer (Merck Pty Ltd). Samples for COD analyses were digested with a thermoreactor model TR 300 (Merck Pty Ltd) prior to analysis using the spectroquant NOVA 60 photometer.

Statistical analysis

Calculation of means and standard deviations (SD) were performed using Microsoft Excel Office 2007 version. Correlations (paired *t* test) and test of significance (two-way analysis of variance) were performed using SPSS 17.0 version for Windows program (SPSS Inc.). All tests of significance and correlations were considered statistically significant at P values of <0.05 or <0.01.

Results and discussion

Mean seasonal values and SD for the different water quality parameters are given in Table 1. The pH values (6.8–8.3) varied significantly with season ($P < 0.01$) and sampling points ($P < 0.01$) and the interaction effect of both season and sampling point was also significant ($P < 0.01$) on the pH. The seasonal variation was likely caused by the significant difference in pH values observed in spring against those of summer ($P < 0.01$) and autumn ($P < 0.01$) and between autumn and winter ($P < 0.05$), while the variation in pH with sampling point must have been a function of the significant ($P < 0.01$) lower pH values (6.8–7.5) observed in the FE compared to DP, UP, and DW (8.0–8.3) and between DP (8.0) and UP/DW (8.2–8.3; $P < 0.01$). The significant interaction effect of season and sampling point on pH indicates that the variation of pH with season was dependent on the sampling point and the observation is corroborated by the fact that FE and, to some extent, DP were mainly responsible for the observed differences in pH during this study (Table 1).

pH ranges similar to those observed in this study have been reported in the literature for final effluents and their receiving waters (Manios et al. 2006). Conversely, Ogunfowokan et al. (2005) reported lower pH ranges (5.23–6.32) and Akan et al. (2008) reported higher pH values (8.94–10.34) for wastewater effluents and their receiving watersheds in Ile-Ife and Jakara (both in Nigeria), respectively. The composition of wastewater effluent varies from facility to facility according to level of treatment, type of households, businesses, industries, and public facilities discharging into the system (Environment Canada 2001) and this could be an important contributory factor to the observed differences in pH. The pH level of a water system determines its usefulness for a variety of purposes. Very high or low pH has been reported (Morrison et al. 2001; DWAF 1996c) to be toxic to aquatic life and alter the solubility of other chemical pollutants as well as some essential elements in water systems (DWAF 1996c), thereby causing adverse effects on the ecosystem and those who depend on it. The South African target water quality for pH in water for domestic use is 6 to 9 (DWAF 1996b), and the European Union tolerance limit for pH in water for the support of fisheries and aquatic life is also set at 6–9 pH units (Chapman 1996). The pH values observed in this study across all sampled points fell within the recommended standards irrespective of season. This suggests that the effluent may

Table 1 Seasonal distribution of physicochemical parameters of the treated final effluents and its receiving waters

Seasons	Sample points	Parameters[a] (mean ± SD)										
		pH	Temperature	EC	Turbidity	Salinity	TDS	COD	DO	NO$_3^-$	NO$_2^-$	PO$_4^{3-}$
Spring	FE	7.1 ± 0.20	20 ± 1.17	749 ± 44	13 ± 2.53	0.36 ± 0.09	372 ± 26	68 ± 0	5.4 ± 0.13	3.8 ± 3.2	2.4 ± 3.8	0.38 ± 0.08
	DP	8.0 ± 0.17	20 ± 0.69	33 ± 9	8.0 ± 2.78	22.9 ± 7.47	16 ± 5	75 ± 0	6.6 ± 0.7	1.28 ± 1.1	0.91 ± 1.1	0.54 ± 0.12
	UP	8.0 ± 0.18	20 ± 1.38	47 ± 1.3	3.3 ± 0.77	34.4 ± 1.0	24 ± 0.58	82 ± 0	6.4 ± 0.36	0.65 ± 0.95	0.7 ± 1.1	0.31 ± 0.1
	DW	8.0 ± 0.16	20 ± 1.24	46 ± 0.52	34 ± 1.26	33.8 ± 1.26	23 ± 0.71	83 ± 0	6.5 ± 0.77	0.32 ± 0.37	0.78 ± 1.22	0.39 ± 0.04
Summer	FE	7.2 ± 0.19	24 ± 0.95	789 ± 172	4.1 ± 1.75	0.38 ± 0.09	367 ± 80	462 ± 599	4.2 ± 0.19	6.5 ± 0.28	0.2 ± 0.15	0.32 ± 0.13
	DP	8.0 ± 0.15	23 ± 1.34	41 ± 8	4.3 ± 1.81	27.8 ± 5.63	20 ± 4	887 ± 1,442	6.1 ± 0.99	3.7 ± 1.8	0.11 ± 0.05	0.34 ± 0.18
	UP	8.3 ± 0.02	21 ± 1.8	48 ± 2.6	2.7 ± 1.30	34.2 ± 0.81	24 ± 0.55	865 ± 1,409	5.7 ± 0.57	2.4 ± 0.67	0.06 ± 0.01	0.29 ± 0.21
	DW	8.2 ± 0.04	21 ± 1.96	47 ± 2.6	34 ± 0.52	33.5 ± 0.52	23 ± 0.42	49 ± 4	6.2 ± 0.80	2.8 ± 2.9	0.07 ± 0.01	0.29 ± 0.20
Autumn	FE	7.5 ± 0.17	25 ± 1.77	1,015 ± 472	3.8 ± 1.10	0.51 ± 0.27	470 ± 232	48 ± 29	3.9 ± 0.99	3.4 ± 3.0	0.23 ± 0.05	0.37 ± 0.31
	DP	8.0 ± 0.18	22 ± 0.67	29 ± 17	6.3 ± 1.74	19.4 ± 12.3	19 ± 4.44	379 ± 445	6.3 ± 0.72	2.3 ± 0.64	0.22 ± 0.18	0.37 ± 0.24
	UP	8.2 ± 0.15	21 ± 1.51	47 ± 0.76	3.8 ± 2.44	33.8 ± 1.55	23 ± 0.88	457 ± 389	6.0 ± 0.49	1.8 ± 0.37	0.10 ± 0.07	0.44 ± 0.39
	DW	8.2 ± 0.18	20 ± 1.52	47 ± 1.84	34 ± 0.36	33.8 ± 0.36	23 ± 0.28	460 ± 382	6.0 ± 0.24	1.3 ± 0.37	0.11 ± 0.05	0.30 ± 0.21
Winter	FE	6.8 ± 0.10	20 ± 2.03	776 ± 42	5.6 ± 0.42	0.42 ± 0.02	387 ± 17	53 ± 31	4.3 ± 0.5	5.6 ± 1.85	0.73 ± 0.40	0.29 ± 0.13
	DP	8.0 ± 0.35	19 ± 0.85	35 ± 12	7.3 ± 3.2	25.4 ± 9.74	19 ± 3.93	1,128 ± 923	5.9 ± 0.99	3.0 ± 0.21	0.2 ± 0.15	0.47 ± 0.44
	UP	8.2 ± 0.13	18 ± 1.15	45 ± 1.33	4.3 ± 3.35	35 ± 0.22	24 ± 0.18	1,180 ± 971	6.1 ± 1.48	3.2 ± 0.64	0.09 ± 0.04	0.42 ± 0.36
	DW	8.3 ± 0.08	18 ± 1.17	45 ± 1.52	35 ± 0.42	34.5 ± 0.42	24 ± 0.3	1,123 ± 919	5.6 ± 0.85	2.3 ± 1.31	0.10 ± 0.05	0.37 ± 0.30
F values[b]		10.32	59.39	2.696	13.973	1.961	4.204	3.616	7.695	9.86	6.276	0.838
P values[c]		0.000**	0.000**	0.049*	0.000**	0.123	0.007*	0.018*	0.000**	0.000**	0.001*	0.477
F values[d]		311.37	31.88	465.104	28.535	546.92	4.072	1.564	41.84	16.151	2.221	0.876
P values[e]		0.000**	0.000**	0.000**	0.000**	0.000**	0.008*	0.207	0.000**	0.000**	0.092	0.457
F values[f]		7.327	3.101	2.863	7.94	1.733	3.712	0.999	1.372	0.517	0.723	0.467
P values[g]		0.000**	0.002*	0.004*	0.000**	0.088	0.000**	0.452	0.207	0.857	0.686	0.893

FE final effluent, *DP* discharge point, *UP* 500 m upstream DP, *DW* 500 m downstream DP

$*P < 0.05$, $**P < 0.01$; significant variation

[a] Values are expressed in milligrams per liter except in pH, temperature (in degrees Celsius), turbidity (in nephelometric turbidity unit), salinity (in practical salinity unit), and EC (in microsiemens per centimeter)

[b] *F* values for parameters and season

[c] *P* values for parameters and season

[d] *F* values for parameters and sampling point

[e] *P* values for parameters and sampling point

[f] *F* values for combined effect of season and sampling point on parameters

[g] *P* values for combined effect of season and sampling point on parameters

not negatively impact on the usefulness of the receiving watershed for domestic, fishery, and recreational purposes with reference to pH standards.

Temperature is an important water quality parameter due to its influence on other parameters. Temperature affects the solubility and, consequently, the availability of oxygen in water (Akan et al. 2008); it also affects the toxicity of some chemicals in water systems as well as the sensitivity of living organisms to toxic substances (Dojlido and Best 1993; Mayer and Ellersieck 1988). The temperature observed in this study ranged from 18°C to 25°C and varied significantly with season and sampling point ($P < 0.01$). The highest temperature was observed in autumn in FE and the lowest observed in winter in the receiving watershed (UP and DW). Temperature was significantly ($P < 0.01$) higher in the FE and DP compared to other sampling points irrespective of season except in spring where the temperature (20°C) was the same for all sampled points. This explains the significant ($P < 0.05$) interaction effect of season and sampling point on temperature (Table 1) and indicates that temperature was not only a function of season but also dependent on sampling point. Our values for temperature fell within the acceptable limit of no risk (≤25°C) for domestic water uses in South Africa (DWAF, WRC 1995). This observation implies that the discharged effluent was of standard quality with respect to temperature and may not significantly offset the homeostatic balance of the receiving ecosystems; neither will it adversely affect the use of the receiving watershed for domestic purposes.

The values for EC in this study ranged between 29 and 1,015 μS/cm across the sampling points and varied significantly with season ($P < 0.05$) and sampling point ($P < 0.01$). The weak significant variation ($P < 0.49$) in EC with season indicates that season only marginally affected EC values (Table 1), while the strong significant variation ($P < 0.01$) in EC values for FE (749–1,015 μS/cm) compared to other points (29–48 μS/cm) may be responsible for the observed difference in EC with sampling point. The significantly higher EC values consistently observed at FE compared to other sampled points suggests that chlorine concentration contributed to the high EC levels at FE (Mamba et al. 2009). The similar EC values observed upstream (UP) and downstream (DW) of the discharge point (DP) showed that the effluent quality normalized with that of the receiving watershed 500 m downstream (Table 1) and generally alludes to the self-cleaning capacity of the receiving watershed as expected of a massive ocean. EC is a measure of dissolved ions in water systems; it has also been reported to be a useful and easy indicator of salinity or total salt content of water systems (Oluyemi et al. 2006; Morrison et al. 2001). Although EC concentrations for FE (749–1,015 μS/cm) fell short of the target water quality limit (70 mS/m or 700 μS/cm) of no risk for domestic water uses (DWAF 1996b), EC values for the receiving watershed (29–48 μS/cm) largely fell within the acceptable limits and suggest that the receiving watershed is safe and fit for domestic uses with respect to EC.

The turbidity of the water systems under study (Table 1) varied from 2.7 NTU (UP, summer) to 35 NTU (DW, winter). The values were similar to those observed by Igbinosa and Okoh (2009) but relatively higher than those reported by Fatoki et al. (2003). Turbidity throughout the study fell short of the target water quality limit (0–1 NTU) of no risk for domestic water uses in South Africa (DWAF 1996b), implying that the water system under study is not suitable for domestic uses with reference to turbidity. Turbidity, however, fell within acceptable limits by the World Health Organization (WHO 2004) standard (≤5 NTU) for effluents to be discharged into the environment in spring (UP), summer (FE, DP, and UP), autumn (FE and UP), and winter (UP; Table 1). Turbidity is a measure of suspended particles (inorganic and/or organic matters) in water systems and usually correlates significantly with microbial load; hence, high turbidity will more often than not support the growth of pathogens and increase the chances of infection (Obi et al. 2007). The presence of suspended particles in a water body could also render it unfit for full-contact recreational uses (DWAF 1996a). There was significant variation in turbidity with season ($P < 0.05$) and sampling point ($P < 0.05$) in this study. The significant difference in turbidity in spring compared to those of summer and autumn ($P < 0.01$) and winter ($P < 0.05$) might be responsible for the seasonal variation (Table 1). The relatively higher turbidity values in spring could be attributed to surface runoff and erosion occasioned by rainfall, carrying soil and silt into the water system (Morokov 1987). The significantly ($P < 0.01$) higher turbidity values observed at DW in relation to other sampling points may be responsible for the variation in turbidity with sampling point (Table 1) and suggests that factors other than effluent quality contributed to the turbidity of the receiving water downstream. The relatively high turbidity levels observed at FE gives cause for concern as high turbidity is reported to affect the effectiveness of chlorination as a means of disinfection (Obi et al. 2007) and increase chances of THM precursor formation in the effluent (Fatoki et al. 2003). THM is a carcinogenic compound formed as a by-product of chlorine and organic matter reaction in water systems and has serious health implications for aquatic life and humans exposed to it (Environment Canada 2001).

Salinity in this study ranged from 0.36 psu (FE, spring) to 35 psu (UP, winter). The values for salinity in FE (0.36–0.51 psu) and DP (19.4–27 psu) fell short of the acceptable limits (33–35 psu) of no risk for all biological activities in the marine ecosystems (SANCOR 1984; Whitfield and Bate 2007) and may adversely affect the aquatic biota of the receiving watershed. However, salinity levels (33–35 psu) at UP and DW fell within the acceptable limits, indicating the self-recovery capacity of the ocean. Salinity varied significantly with sampling point ($P < 0.01$) but not with season during this study. Salinity at FE was consistently and significantly ($P < 0.01$) lower than values recorded in the receiving watershed (DP, UP, and DW) which may be responsible for the observed difference (Table 1). Salinity is the saltiness of a water body and high salt content in effluents discharged into a receiving watershed could cause serious ecological disturbance that may result in adverse effects on the aquatic biota (Morrison et al. 2001; Oluyemi et al. 2006).

TDS values in this study varied between 16 mg/L (DP, spring) and 470 mg/L (FE, autumn). The values fell within acceptable limits (≤2,000 mg/L) for effluents discharged into surface waters by the WHO standards (Akan et al. 2008). It also fell within acceptable limits (0–450 mg/L) for South African water systems applied in domestic uses (DWAF 1996b) except in autumn when the TDS value (470 mg/L) in the final effluent (FE) exceeded the target water quality limit of no risk (Table 1). TDS like EC is a measure of salinity in water systems. The relevance of this parameter to water quality is similar to those discussed under EC and salinity. In addition, TDS as a measure of salinity is an important agricultural water quality parameter with respect to soil salinity. Salinity of soil has been reported to be related to and often determined by the salinity of the irrigation water (FAO 1992), while plant growth, crop yield, and quality of produce are affected by the TDS concentration in irrigation water (FAO 1992). This is worthy of note as the effluent from the wastewater facility under study is used as water resource for a fishpond as well as to irrigate a nearby golf course. TDS varied significantly with season ($P < 0.05$) and sampling point ($P < 0.01$). The significant difference ($P < 0.05$) in TDS values observed in autumn compared to those of other seasons may be responsible for the observed seasonal variation, while the relatively high TDS concentration (367–470 mg/L) observed in FE compared to other sampled points (16–24 mg/L) is likely the reason for the observed difference in TDS with sampling point (Table 1). The TDS values at FE during this study were higher than those reported by Igbinosa and Okoh (2009); conversely, Akan et al. (2008) reported higher TDS values (2,210–2,655 mg/L) for the receiving watershed compared to those (16–36 mg/L) observed in this study.

COD is a measure of the amount of oxygen required by a strong oxidant (e.g., H_2SO_4) to break down both organic and inorganic matters in a water system (Akan et al. 2008). Elevated levels of COD in water systems lead to drastic oxygen depletion which adversely affects the aquatic biota (Fatoki et al. 2003). COD concentrations in this study ranged between 48 and 1,180 mg/L with the highest value recorded upstream of the urban effluent discharge (UP) in winter and the lowest value observed at FE in autumn. The values fell short of the acceptable target limit (30 mg/L) recommended by the South African government for effluents to be discharged into surface waters (Government Gazette 1984) and suggests that the effluent may negatively impact on the receiving environment. COD concentrations, however, fell within acceptable limits ($\leq$1,000 mg/L) of no risk by the WHO standard for effluents to be discharged into surface waters (Akan et al. 2008) except in winter where COD values for DP, UP, and DW were higher than the recommended limit. COD significantly varied with season ($P <$ 0.05) and sampling point ($P < 0.01$) and values were generally highest in winter followed by summer, autumn, and spring, respectively. The higher COD values in winter compared to other seasons could be attributed to the lesser rate of organic matter breakdown (occasioned by lower microbial activity) during the cold (winter) season compared to the warmer seasons (Tomida et al. 1999). The higher COD values observed in the receiving watershed (UP, DW, and DP, respectively) compared to FE suggested that unidentified sources contributed more COD to the watershed than the final effluent. Several authors have reported the pollution of surface water bodies by nonpoint sources such as domestic, municipal, and/or agricultural runoffs (Hacioglu and Dulger 2009; Pradhan et al. 2009; Shirodkar et al. 2009). Contrary to the observation of this study, Morrison et al. (2001) reported higher COD values for final effluents compared to the receiving watershed in their study of the Keiskammahoek sewage treatment facility and its receiving river.

The DO levels in this study varied from 3.9 to 6.6 mg/L across the sampled points and were similar to those reported previously (Oluyemi et al. 2006; Akan et al. 2007). DO varied significantly with season ($P < 0.01$) and sampling point ($P <$ 0.01). The significantly higher DO values recorded in spring versus summer and autumn ($P < 0.05$) and winter ($P < 0.01$) may be responsible for the observed seasonal variation (Table 1), while the observed difference in DO with sampling points must have been occasioned by the significantly ($P < 0.01$) lower DO values (3.9–5.4 mg/L) observed in FE compared to the other sampling points (5.7–6.6; Table 1). This indicates that the nutrient load of the final effluent was generally higher than those of the receiving watershed (Akan et al. 2008; CCME 2006) and implies that the treated effluent is a contributing source of nutrient to the receiving watershed. The DO levels in this study fell short of the acceptable limit ($\geq$5 mg/L) of no risk for the support of aquatic life (Fatoki et al. 2003) in the final effluent except in spring 2007 where FE was compliant with the stipulated standard (Table 1). DO levels in the receiving watershed were, however, within the recommended standard throughout the period of study, indicating that the receiving watershed supports the survival of the aquatic biota. Dissolved oxygen is essential in maintaining the oxygen balance in an aquatic ecosystem; low dissolved oxygen level in water system is reported to have adverse effects on the aquatic life (Fatoki et al. 2003). It affects the survival of fish by increasing their susceptibility to disease, hampering swimming ability, altering feeding, migration, reproductive behavior, and ultimately, leads to death of aquatic life (Environment Canada 2001).

Nitrate concentration in this study varied between 0.32 mg NO_3^- as N/L and 6.5 mg NO_3^- as N/L and generally fell short of the acceptable safety limit (1.5 mg NO_3^- as N/L) for effluent to be discharged into surface waters in South Africa (Government Gazette 1984). The new South African target water quality standard for nitrate considers the effect of this compound on the health of infants and pregnant women and thus set the safety limit for domestic water supply at 6 mg NO_3^- as N/L (DWAF 1996b). Based on this new standard, the nitrate concentrations in this study were mostly within acceptable limits (Table 1) and suggest that the water system under study is fit and safe for domestic applications. Nitrate concentration, however, slightly exceeded the safety limit in the final effluent during summer (6.5 mg NO_3^- as N/L; Table 1). Nitrate concentrations varied significantly ($P < 0.01$) with season and sampling point. The significant difference in nitrate concentrations recorded in spring against summer ($P < 0.01$) and winter ($P < 0.05$) and for autumn against summer ($P < 0.05$) and winter ($P < 0.05$) may be responsible for the observed seasonal variation. The significantly ($P < 0.01$) higher nitrate values (3.4–6.5 mg NO_3^- as N/L) in the FE compared to other sampled points (0.32–3.7) is likely the cause of the observed difference in nitrate with sampling point (Table 1). The observation suggests that the final effluent was a significant contributor of nitrate to the receiving watershed, in agreement with the report of

Morrison et al. (2001) but contrary to the observation of Ogunfowokan et al. (2005). Nitrates are inorganic sources of nitrogen that support the growth and development of living organisms at appropriate concentrations. However, high nitrate levels may result in excessive nutrient enrichment in water systems (eutrophication) leading to loss of diversity in the aquatic biota and overall ecosystem degradation through algal blooms, excessive plant growth, oxygen depletion, and reduced sunlight penetration (CCME 2006). It has also been reported that nitrate concentration above 45 mg/L may result in anemia in infants and pregnant women and formation of carcinogenic nitrosamines (Akan et al. 2007).

Nitrite like nitrate is a source of nutrient that could have adverse effects on aquatic ecosystems at elevated concentrations. Their effects on water systems are generally similar to those described for nitrate. The South African limit (0–6 NO_2^- as N/L) of no adverse effect for nitrite in domestic water supply is the same as in nitrate (DWAF 1996b) and suggests that the entire water system under study was fit and safe for domestic uses based on their nitrite concentrations (0.06–2.4 NO_2^- as N/L). The nitrite levels recorded in the entire water system in spring and in the final effluent in winter, however, fell short of the South African standard (<0.5 NO_2^- as N/L) for the preservation of the aquatic ecosystem (DWAF 1996c) and, therefore, put the aquatic ecosystem at risk of eutrophication. The nitrite levels during the other seasons do not pose any serious threat to the integrity of the aquatic ecosystem by reason of this standard. Nitrite significantly varied with season ($P < 0.05$) but not with sampling point (Table 1). Nitrite concentration was highest in spring followed by winter, autumn, and summer, respectively. The significant ($P < 0.05$) difference in nitrite concentration in spring compared to other seasons may be responsible for the observed seasonal variation and suggests that surface runoff and erosion occasioned by rainfall during this (spring) season may be a significant factor in the observation (Morokov 1987). Although nitrite did not vary significantly with sampling point, the nitrite concentration downstream (DW) generally reflected nitrite levels in the final effluent throughout the sampling period (Table 1) and suggests that the final effluent was the major contributor of nitrite to the receiving watershed.

Orthophosphate (as P) levels in this study varied from 0.29 mg PO_4^{3-} as P/L to 0.54 mg PO_4^{3-} as P/L across seasons and sampling points. The P levels observed in this study exceeded the South African target limit of 5 μg/L (0.005 mg PO_4^{3-} as P/L) for P in water systems that will reduce the growth of algae and other plants and suggests that the water is polluted and pose serious threat to the aquatic biota in particular and the ecosystem in general. P did not vary significantly with season or sampling point. The higher P levels sometimes observed in the receiving watershed compared to the final effluent suggests that there were other unidentified sources of P in the water system. This could be as a result of agricultural, municipal, or domestic runoffs (nonpoint sources) that flowed into the receiving watershed from diverse sources in the catchment area under study (Correll 1998). Similar P levels as observed in this study had been previously reported (Morrison et al. 2001; Fatoki et al. 2003); higher P levels were, however, reported by other workers (Ogunfowokan et al. 2005; Akan et al. 2008). Phosphates are reported to be the most important growth-limiting factor in eutrophication and results in a number of undesirable ecological effects in the water system (CCME 2006). Common sources of phosphate in water systems are domestic wastes (e.g., phosphate-based detergents) and agro-allied chemicals such as fertilizers (Ogunfowokan et al. 2005).

Conventional approaches to water quality assessment are based on comparison of experimentally determined parameter values with existing guidelines. While this methodology is appropriate for checking legal compliance and allows proper identification of contamination sources, it does not give a holistic picture of the spatial and temporal trend of the overall quality of the water system (Boyacioglu 2007). Due to the complex nature of the physical, chemical, biological, and socioeconomic processes that govern the water system, researchers are exploring ways to better understand the interrelationships and interactions of the components involved in these processes under various circumstances. Such understanding promises to further our capacity to preserve and manage our water systems. Several authors (Shyamala et al.

2008; Pradhan et al. 2009; Shirodkar et al. 2009) have used correlation as a tool to elucidate the interrelationship between and among water quality parameters as well as to trace the possible sources of contamination in a complex environment. Furthermore, conventional water quality assessments could involve as many as 20 parameters to adjudge a water system fit for use or otherwise. This could be very expensive especially for developing countries such as South Africa and could limit water quality evaluation in such countries. Correlation among other tools can also be used to identify parameters that are representative of others in order to cut down on the number of parameters that might be critical to adjudging the quality of a water system (Boyacioglu 2007). In this section, we employ correlation as a tool to elucidate the interactions and interrelationships between water quality parameters and their usefulness in identifying possible sources of pollution.

The correlation matrix of the various physicochemical parameters is given in Table 2. There was significant positive correlation between and among pH, salinity, and DO ($P < 0.01$), while these parameters negatively correlated with EC, TDS, and nitrate ($P < 0.01$) and with nitrite ($P < 0.05$). The positive correlation between pH and salinity is generally indicative of the higher pH concentration of the more saline-receiving watershed compared to the less saline effluent (Table 1). The positive correlation between DO and salinity indicated that DO concentration increased with increasing salinity, suggesting that the more saline-receiving watershed is better oxygenated compared to the less saline effluent. The better oxygenation of the watershed must be sequel to the wind-induced turbulence and mixing of the marine water near the seashore where watershed samples (DP, UP, and DW) were collected (Shirodkar et al. 2009). The results revealed that the wastewater effluent was the main contributor of low dissolved oxygen to the watershed; however, it is worthy of note that the receiving water quickly returned to DO levels similar to those observed upstream after flowing about 500 m downstream from the point of effluent discharge (Table 1), indicating its self-cleaning capacity.

There are several reports in the literature suggesting that EC and TDS were good and easy indicators of salinity (Oluyemi et al. 2006; Akan et al. 2008); results from this study, however, reveals that this may not always be the case. While the near-perfect correlation between EC and TDS suggests that these two parameters could very well represent one another in the determination of water quality irrespective of external and internal influences, their inverse relationship with salinity suggest that they may not always be good indicators of salinity. Our study showed that, in the final effluent where chloride ions are dominant compared to sodium and other ions in the receiving waters (results not shown), EC and TDS values were significantly higher compared to salinity. Furthermore, if EC and TDS were very good indicators of salinity, it would be expected that the introduction of effluent high in EC and TDS

Table 2 Correlation matrix of physicochemical variables in treated final effluents and the receiving watershed

Variables	pH	Temperature	EC	Turbidity	Salinity	TDS	DO	COD	Nitrate	Nitrite	Phosphate
pH	1.0										
Temp	−0.372	1.0									
EC	−0.894**	0.556*	1.0								
Turbidity	−0.506*	−0.523*	0.300	1.0							
Salinity	0.939**	−0.553*	−0.930**	−0.523*	1.0						
TDS	−0.908**	0.534*	0.999**	0.322	−0.935**	1.0					
DO	0.732**	−0.537*	−0.905**	−0.028	0.796**	−0.899**	1.0				
COD	0.494	−0.341	−0.420	−0.164	0.418	−0.425	0.136	1.0			
Nitrate	−0.704**	0.417	0.701**	0.228	−0.738**	0.705**	−0.767**	0.082	1.0		
Nitrite	−0.576*	−0.120	0.377	0.781**	−0.459	0.401	−0.015	−0.507*	0.034	1.0	
Phosphate	0.172	−0.293	−0.233	0.433	0.086	−0.236	0.350	0.136	−0.295	0.170	1.0

EC electrical conductivity, *TDS* total dissolved solids, *DO* dissolved oxygen

$^{*}P = 0.05$, $^{**}P = 0.01$; significant correlation (two-tailed) levels into the saline-receiving watershed would lead to an increase in EC and TDS levels with a concomitant increase in salinity, but the reverse was actually the case in this study (Table 1). This, therefore, implies that the type of dissolved ions present in a water system will to a large extent determine whether or not EC and/or TDS would be good surrogates of salinity.

The significant negative correlation of salinity with nitrate and nitrite also points to the less saline municipal effluent as the source of these nutrients in the watershed. This could further be explained by the consistent higher concentrations of nitrate and nitrite observed in the effluent compared to other sampled points throughout this study (Table 1). The significant positive correlation between and among temperature, EC, and TDS ($P < 0.05$) and their (EC and TDS) negative correlation with salinity, pH, and DO ($P < 0.01$; Table 2) generally showed that the less saline effluent had higher temperatures compared to the more saline-receiving watershed during this study (Table 1). The inverse relationship between turbidity and salinity suggests that the less saline effluent may be a source of turbidity in the watershed. However, the fact that turbidity did not correlate significantly with other prominent parameters in the effluent (e.g., EC and TDS) suggests that there may be other source(s) of turbidity in the receiving watershed apart from the effluent. This other source(s) may be responsible for the elevated levels of turbidity observed at DW compared to FE and other sampled sites (Table 1).

COD and orthophosphate did not correlate significantly with other parameters, suggesting diffuse origins of these parameters (COD and orthophosphate) in the watershed. The insignificant negative correlation of COD with EC and TDS and its insignificant positive correlation with salinity, pH, and DO, however, suggest that COD was introduced into the watershed by an unidentified source upstream of the effluent discharge. This observation is corroborated by the elevated levels of COD observed upstream compared to other sampling points, especially FE and DW (Table 1). The slightly higher COD concentration observed in DP during summer and autumn is most likely a result of additional COD from the municipal effluent (FE) to the upstream water. A cursory look at Tables 1 and 2 suggest that orthophosphate followed a similar trend as COD and indicates a common source.

In general, for reasons mentioned earlier with respect to the complex nature of the processes that govern water systems, it is difficult to compare the activities of one water system to another due to their uniqueness. For example, contrary to the observation of this study, Shirodkar et al. (2009) reported significant negative correlation between salinity and DO in the coastal waters of Mangalore in India. The authors explained that the incursion of the less saline riverine water compared to the more saline marine water was responsible for this observation. In a similar vein, Pradhan et al. (2009) reported positive correlation between pH and the nitrogenous nutrients (nitrate and nitrite) contrary to the observation of this study. Nutrient incursion was also cited as responsible for this observation. Consistent with our observation, Igbinosa and Okoh (2009) reported significant positive correlation among pH, DO, and salinity ($P < 0.05$) and between EC and TDS ($P < 0.01$), while they reported significant negative correlation between pH and nitrate ($P < 0.01$) and between temperature and DO ($P < 0.01$). Contrary to our observation, however, the authors (Igbinosa and Okoh 2009) reported significant positive correlation for salinity with EC and TDS ($P < 0.01$) and for pH with TDS and EC ($P < 0.05$). The most stable relationship common to all the studies was seen between EC and TDS. This is an indication that external influence has little or no effect on these parameters and that they both represent each other very well, thus suggesting that either of the two parameters can be used to measure water quality in the stead of the other where limited resources is an issue.

Conclusion

The primary objective of this study was to evaluate the physicochemical qualities of the final effluent of an urban wastewater treatment facility in South Africa as a surrogate index of its capacity to remove selected pollutants from the wastewater influent prior to discharge into the receiving environment. While the treated effluent met the recommended water quality standard for pH temperature, TDS, NO_2^-, and NO_3^-, it fell short of stipulated standards for EC, turbidity, salinity, COD, DO, and PO_4^{3-}. The result generally showed a negative impact of the discharged effluent on the receiving watershed and calls for a regular and consistent monitoring program by the relevant authorities to ensure best practices with regard to treatment and discharge of wastewater into the receiving aquatic milieu.

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