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# Performance assessment of wastewater treatment plants with special reference to phenol removal

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## Abstract

This study investigated the performance of five different wastewater treatment plants (WWTPs) in some rural areas in the Eastern Cape, South Africa, through an assessment of the physicochemical qualities of the effluents and their removal capacity for selected phenolic compounds. Water samples were collected on a monthly basis (February–July 2016), and temperature, pH, turbidity, total dissolved solids (TDS), total suspended solids (TSS), electrical conductivity (EC) and dissolved oxygen (DO) were measured *on-site* using standard methods. Phenolic compounds were determined using the solid-phase extraction technique followed by derivatization and gas chromatography–mass spectrometry analysis. Recorded temperatures for all the WWTPs effluents were 29–30 °C in summer, 20–27 °C in autumn and 12–17 °C in winter, and the pH was between 6.8 and 9.1. The treatment processes increased only the DO of the final effluents, while TDS, EC, turbidity and TSS were considerably reduced. Out of the nine phenolic compounds investigated, 2,4-dimethylphenol (2,4-DMP), 4-chloro-3-methylphenol (4-C-3MP), 2-nitrophenol (2-NP) and pentachlorophenol (PCP) were prominent in the influent (17.3–42.1 µg/L) and concentrations of phenolic compounds in the effluents and receiving water bodies were mostly below the tolerable limits allowed internationally for a safe aquatic environment. Generally, the WWTPs removal capacities range between 33 and 96%, however, South Africa standard limits for discharged effluents were occasionally exceeded especially with some WWTPs. This poses health challenges to rural communities that rely heavily on surface water for domestic use.

**Keywords** Final effluent · Phenolic compounds · Physicochemical parameters · Receiving water · Wastewater treatment plants

## Introduction

Efficient wastewater treatment plants (WWTPs) are those that produce effluents that meet the desired standard criteria in chemical and biological quality at the minimum operational cost and maintenance (Sukumaran et al. 2015). The relative ease of removal of contaminants of health concern

such as pathogens, chemical waste including heavy metals in the WWTPs remained a strong yardstick for measuring the efficiency of WWTPs (Coskuner and Ozdemir 2006). In line with this, developing an integrated approach for assessing the efficiency of WWTPs is, therefore, a welcome idea in this direction (Sukumaran et al. 2015).

In South Africa for instance, many water quality studies conducted identified poor operation and maintenance of wastewater and sewage treatment facilities as contributors to pollution of surface water on which many rural communities depend for their domestic, agricultural and other purposes (Mema 2010). Consequently, this has impacted negatively on human health and the environment. Williams (2008) stated that the water quality of a combined sewer is dependent on the different lifestyles of the inhabitants of the area and the temporal flow pattern. Different disease outbreaks: diarrhea, typhoid fever, cholera, reported in some parts of the country (Delmas, Mpumalanga Province, KwaZulu-Natal, Limpopo Transkei and Eastern Cape), have led to the death of children

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(Addendum 2008; Mema 2010). These outbreaks were linked to town water supplies which were suspected to have been contaminated with human feces, because of poor microbiological water quality linked to sewage spills from catchment-based land activities and spillage of untreated sewage.

The monitoring of physicochemical parameters in water resources in order to assess water quality and any potential risk is a necessary tool in the protection of the environment and of public health (Okoh et al. 2007). Temperature, pH, odor, electrical conductivity (EC), total dissolved solids (TDS), turbidity, radioactivity, chloride, phosphate, nitrate nitrogen and biochemical oxygen demand are among physicochemical parameters listed by Tebbut (1998) that need to be tested for in different water sources. In view of their significance to health, these parameters are recognized by international standards for the evaluation of surface water quality (Tebbut 1998; Chigor et al. 2013).

In the Eastern Cape of South Africa, many households have no access to pipe-borne water especially in the rural areas according to the South African Department of Environmental Affairs and Tourism (DEAT) (Chigor et al. 2013). This has made the use of water from alternative sources such as surface water inevitable. However, the inefficiency of most treatment plants in this region has been reported, along with their negative impact on the physicochemical qualities of the receiving water bodies (Igbinosa and Okoh 2009; Odjadjare and Okoh 2010).

Phenol and phenolic compounds are ubiquitous contaminants in the environment. Industrial, domestic and vehicle emissions are the various sources through which these contaminants are introduced to the environment (Kumar et al. 2014). They are produced in the industry for various uses which include plant protection (pentachlorophenol and tetra-chlorophenol), and wood preservation because of their fungicidal and antiseptic properties (Schummer et al. 2009; Kumar et al. 2014). These compounds are dominant chemicals in pharmaceuticals, textiles, plasticizers, pulp and paper, pesticide manufacturing, the wood industry, detergent application and metallurgic industries (Santana et al. 2009). Phenolic compounds, especially chlorophenols, have been reported to be highly toxic, estrogenic, carcinogenic and also inhibit biological activities during wastewater treatment (Guedes and Leitaó 2012).

The potential toxicity and negative impacts of these compounds on the environment have garnered increased public attention in recent years, so much so that some phenolic compounds such as chlorophenols and nitrophenols have been classified as priority pollutants by World Health Organisation (WHO), U.S. Environmental Protection Agency (USEPA) and the European Community (EC) (USEPA 1982; WHO. 1989; EC 2001). These same bodies have classified some of these compounds as endocrine disrupting chemicals (EDC) due to their potential to interact with hormone receptors. The USEPA and EC have set the

minimum permissible level for phenolic compounds in water intended for drinking at 0.5 µg/L for total compounds and 0.1 µg/L for individual compounds, and 5 µg/L for bathing water (Silva et al. 2009; Kumar et al. 2014).

This present study which was carried out between February and July, 2016, focuses on evaluating the performance of five WWTPs operated under different treatment technologies in two Municipalities in the Eastern Cape, South Africa, using the physicochemical qualities of their discharged effluents and their removal capacity for phenolic compounds. Effluent compliance with set standards of physicochemical quality will also be assessed for the purpose of determining their impact on the quality of the receiving water for domestic and agriculture uses, as well as for the survival of the aquatic life.

## Materials and methods

### Study area

The Eastern Cape Province, where this study was conducted, is located on the east coast of South Africa and is the second largest province in the country (area 168,966 km<sup>2</sup>), with the third largest population (about 7 million) and where subsistence agriculture predominates. Out of the five selected WWTPs for this study, four are located in the Amathole District Municipality and they include: WWTPs serving Alice, Adelaide, Bedford and Seymour, while the Berlin WWTP is in the Buffalo District Municipality (Table 1). The estimate of their population was based on the South Africa population census results for 2011.

It is important to note that the final effluent at Bedford was not discharged into any river but was used as irrigation water to serve the Golf Course adjacent to the treatment plant. Sometimes it was collected by construction companies for site work. At Berlin, on the other hand, the final effluent is a tributary of the Nahoon River which flows a long distance before it gets to the final destination. The river could not be assessed for the upstream, but the sample was taken about 600 m away from the discharging point.

### Chemicals and materials

A standard mixture of phenolic compounds (2000 µg/mL), containing phenol (PH) 99%, 2-chlorophenol (2-CP) 99.8%, 2,4-dimethylphenol (2,4-DMP) 100%, 2,4-dichlorophenol (2,4-DCP) 100%, 4-chloro-3-methylphenol (4-C-3MP) 99.7%, 2-nitrophenol (2-NP) 100%, 4-nitrophenol (4-NP) 100%, pentachlorophenol (PCP) 99%, 2,4,6-trichlorophenol (2,4,6-TCP) 100%, was sourced from Accu-standard, Inc., USA. Surrogate standard, 2-fluorobiphenyl (2FB), was bought from Restek, USA, while bis(trimethylsilyl)trifluoroacetamide (BSTFA) and chlorotrimethylsilane used for



**Table 1** Brief description of selected wastewater treatment plants

Plants	Adelaide	Alice	Bedford	Berlin	Seymour
Technology	Activated sludge	Activated sludge	Oxidation pond	Trickling filters	Activated sludge
Design capacity (ML/d)	0.5	0.5–2	0.5–2	0.5–2	0.25
Geographical location	S32°42.343'S 26°18.790'E	32°47.566'S 26° 50.958'E	32°41.15S 26° 05'E	32°50.700'S 27° 37'E	32°47.566'S 26° 50.958'E
Treatment process	Scr → Gr → Se → AS → Sc → Chl	Scr → Gr → Sed → AS → Sc → Chl	Scr → Gr → Ap → Sed	Scr → Gr → Sed → Tf → Sc → Chl	Scr → Gr → Sed → AS → Chl
Population	12,191	15,143	8770	3048	2467
Total area km <sup>2</sup>	40	9.65	14.6	38.22	2.59
Receiving River	Cobra	Tyume	No discharge	Tributary of Nahoon	Kat

Scr. screening, Gr. grit removal, Sed. sedimentation, AS. activated sludge, Sc. secondary clarifier, Chl. chlorination, Ap. aeration pond; Tf. trickling filter

derivatization were purchased from Macherey–Nagel and Darmstadt, respectively, both from Germany. A working mixture of phenolic compounds and surrogate at 100 µg/L was prepared from the stock standard solution in methanol and stored at 4 °C in an amber bottle. Turbidity was measured using Hach instrument 2100P Turbidimeter, while Hanna multi-parameter probe (HI 98195) and Hach DR 900 colorimeter were used to measure other physicochemical parameters *on-site*, which include temperature, pH, dissolved oxygen (DO), electrical conductivity (EC), total dissolved solids (TDS) and total suspended solids (TSS).

Solid-phase extraction cartridges (SPE) C18-U (strata) 1000 mg/6 mL, used for the extraction of wastewater, were from Phenomenex. HPLC-grade dichloromethane, n-hexane, methanol, acetone and glass wool were sourced from Darmstadt, Germany. Double-distilled water was produced in the Department of Chemistry's laboratory, University of Fort Hare. Sodium thiosulfate pentahydrate and anhydrous sodium sulfate (which was heated in a muffle furnace at 450 °C for 2 h) were from Merck, South Africa. Pure nitrogen gas (99.995%) bottled by Afrox gas, South Africa, was used for drying.

### Sample collection and preparation

A monthly sampling campaign was conducted for 6 months between February and July, 2016, for physicochemical parameters and 4 months (April–July) for phenolic compounds. Wastewater samples were collected from two different sampling points at the respective WWTPs; the influent, final effluent and samples from receiving water bodies (upstream and downstream) were collected about 500 m away from the discharging point. Pre-cleaned 1-L amber glass bottles with PTFE-lined polypropylene screw cap were used for sample collection. Sampling bottles were first rinsed with the wastewater samples before composite samples were collected and preserved immediately with 20–40 mg sodium thiosulfate and HCl<sub>(aq)</sub> to a pH of ≤ 2 according to standard procedure. Samples were preserved

in ice and transported to the Chemistry laboratory at the University of Fort Hare, South Africa, where samples were kept at 4 °C in the refrigerator until the time of analysis, which was not more than 3 days after the collection.

### Physicochemical measurement

The selected physicochemical parameters were evaluated *on-site* using a multi-parameter ion specific meter following standard method as contained in America Public Health Association (APHA 2012). The Hach equipment was appropriately examined and calibrated according to manufacturer's instructions before use. Readings were carried out in triplicate, and enough time was allowed for the Hannan multi-parameter probe to stabilize before readings were taken. The reduction capacity of WWTPs for some physicochemical parameters (turbidity, TSS and TDS) was calculated in percentage as the ratio of the difference between average influent values and final effluent value to the average influent value multiplied by 100.

$$100 \times \{(\text{Inf} - \text{Ef})/\text{Inf}\} \quad (1)$$

where Inf = Average influent value and Ef = Average effluent values.

The WWTPs evaluation was based on its compliance with standard practice as contained in South African water quality guidelines for domestic and recreational water (DWA 1996a, b).

### Sample extraction and derivatization

All water samples collected were extracted following the modified method of Sánchez-Avila et al. (2009) and Munch (2000). Wastewater samples were first filtered through glass wool to remove suspended particles. Extraction was carried out with 500-mL samples of the filtrate which was spiked with 25 µL of the standard surrogate (2FB) to monitor



method efficiency. The wastewater samples were passed at the rate of 10 mL/min through the C-18-U cartridges previously conditioned, with 7 mL each of n-hexane, dichloromethane, methanol and 10 mL distilled water at the rate of 3 mL/min. Sample bottles were rinsed into the cartridges and allowed to dry under vacuum for 1 h. Elution was carried out with 7 mL each of a mixture of dichloromethane/ n-hexane (1:1 v/v) and dichloromethane/ acetone (1:1 v/v), and eluents were reduced to about 2 mL using a rotary evaporator and blown to almost dryness before derivatization. The respective residues were re-constituted with 200  $\mu$ L of a mixture of N,O-bis(trimethylsilyl)trifluoroacetamide (BSTFA) and chlorotrimethylsilane (99:1 v/v) in vials and put in the oven at 70 °C for 30 min for complete derivatization.

### Instrumental analysis

Extracted samples were analyzed using gas chromatography–mass spectrometer (GC–MS) Agilent technology 7890B gas chromatograph coupled with mass selective detector (MSD) 5977A. The dimension of the capillary column HP-5 MS was 30 m  $\times$  250  $\mu$ m i.d.  $\times$  0.25  $\mu$ m film thickness. Helium gas at 90-kPa pressure with a constant flow of 1 mL/min was used as the carrier gas. The transfer line and injection port were set at 280 and 300 °C, respectively, and the volume of the sample extract (1  $\mu$ L) was injected by auto-sampler in a splitless mode. The GC oven temperature was initially set at 70 °C held for 1 min before it was increased to 150 and 290 °C at the rate of 14 and 6 °C/min, respectively.

### Quality assurance and quality control

Fortified procedural blanks and a solvent blank (laboratory reagent blank LRB) were subjected to the same analytical procedure in every batch of 10 samples under analysis to detect any possible background contaminant during extraction. Every sample analyzed was fortified with the surrogate standard for evaluation of method performance, and the replicate analysis of four laboratory fortified blanks (LFBs) gave a standard deviation of below 15%. Instrument sensitivity and reproducibility were always verified by injecting one calibration standard before and after every batch analysis. A calibration standard of phenol mixture and surrogate with a concentration ranging from 0.1 to 15  $\mu$ g/L was analyzed to generate a calibration curve. An external standard calibration was adopted with at least five points for each of the phenol analytes, and concentrations were determined within the linear range of the calibration curve. The correlation coefficient for each of the nine selected phenol compounds varied between 0.995 and 0.999.

### Statistical analysis

The mean values of data obtained were compared using SPSS (IBM SPSS Statistics 20), one-way analysis of variance and Tukey's multiple range tests. Significance for statistical analysis was set at  $p$  values  $<$  0.05).

## Results and discussion

### Variation in physicochemical parameters in WWTPs processes and receiving water bodies

#### Temperature

The sampling periods cut across three weather seasons summer, autumn and winter for all the WWTPs, and the temperature varied as 29–36, 20–27 and 12–17 °C, respectively. The average temperatures for all the treatment plants in the influent, effluent, upstream and downstream were in the range of 21–25, 14–23, 18–22 and 18–25 °C, respectively, and there was no significant difference ( $p >$  0.05) among the respective sampling points. The results of the temperature variation and other physicochemical parameters (pH, TDS, DO and EC) at the five selected WWTPs in the Eastern Cape, South Africa, at different sampling points are presented in Table 2.

A significant difference in temperature was observed across the three seasons ( $p <$  0.05). Temperatures were observed to be generally lower in samples collected at Seymour than other areas; it varied on the average from 14 to 21 °C and a little higher at Bedford (25–35 °C) where sampling are mostly carried out in the afternoon. This might be connected to the different time of sampling during the day. The impact of temperature on a variety of parameters (DO, the toxicity of some chemicals and consequent exposure of living organisms to the toxic substance) makes it a vital tool in water quality determination (Fondriest Environmental 2014a).

The relatively similar temperature range between the sampling points and the receiving water suggests that temperature was independent of WWTP processes and as such, the final effluent may not impact negatively on the receiving water. However, a possible temperature change may have occurred during microbial activities during the secondary treatment, (especially with AS). Such an effect probably could have been neutralized at the secondary clarifier where the treated water was exposed to atmospheric temperature. By inference, the temperature across the sampling points was significantly influenced by seasonal change.

These results are similar to other studies reported by Igbiosa and Okoh (2009) and Odjadjare and Okoh (2010) in South Africa, but those previous studies both observed a significant difference between sampling points (final effluent and receiving water). Although temperature differs among

**Table 2** Physicochemical parameter (temperature, pH, DO, TDS and EC) variance at different points in different WWTPs

WWTP	Influent	Range	Effluent	Range	Upstream	Range	Downstream	Range
Temperature °C								
Adelaide	23 ± 3	15–35	22 ± 3	15–33	22 ± 3	15–34	22 ± 3	16–34
Alice	22 ± 2	16–30	22 ± 3	13–31	22 ± 4	11–30	22 ± 3	14–29
Bedford	25 ± 4	15–36	21 ± 3	13–35	NS	NS	NS	NS
Berlin	23 ± 2	17–28	23 ± 4	15–31	NS	NS	25 ± 3	17–31
Seymour	21 ± 2	17–24	14 ± 7	18–25	18 ± 2	13–23	18 ± 1	16–23
Hydrogen ion concentration (pH)								
Adelaide	7.9 ± 0.2	7.2–8.5	7.8 ± 0.2	7.2–8.7	8.4 ± 0.3	7.7–9.4	8.2 ± 0.2	7.7–9.2
Alice	7.6 ± 0.2	6.8–8.5	7.3 ± 0.4	6.5–8.8	7.6 ± 0.3	6.7–8.7	7.0 ± 0.3	6.8–8.7
Bedford	7.8 ± 0.3	6.9–8.6	7.9 ± 0.3	6.8–9.1	NS	NS	NS	NS
Berlin	7.8 ± 0.2	7.1–8.8	8.3 ± 0.3	7.6–8.9	NS	NS	8 ± 0.2	7.6–8.8
Seymour	7.9 ± 0.3	7.3–8.8	7.4 ± 2.5	7.3–7.5	7.5 ± 0.1	7.3–8.0	7.4 ± 0.2	7.3–8.0
Electrical conductivity (µS/cm)								
Adelaide	530 ± 50.7	357–664	454.3 ± 38.5	357–587	442.1 ± 76.2	143–637	354.4 ± 45.7	143.3–480.3
Alice	366 ± 54.8	218–619	258 ± 18.4	218–318	195 ± 20.2	155–260	217 ± 29.9	143–319
Bedford	437 ± 81.6	258–772	324 ± 22.3	279–413	NS	NS	NS	NS
Berlin	648 ± 84.7	471–1030	646 ± 48.3	557–781	NS	NS	689 ± 61.9	492–827.3
Seymour	373 ± 32.7	290–450	204 ± 103	292–321	153 ± 31.7	23–219	181 ± 41.2	25–312.7
Total dissolved solids (TDS) mg/L								
Adelaide	309 ± 20.3	228–368	265 ± 9.9	229–296	294 ± 56	90–487	210 ± 29.9	90–307
Alice	197 ± 12.3	184–227	147 ± 5.1	136–166	108 ± 4.9	99.5–131	122 ± 12.3	91–171
Bedford	343 ± 70.2	181–650	189 ± 4.1	179–206	NS	NS	NS	NS
Berlin	381 ± 42.5	231–523	385 ± 12.4	352–412	NS	NS	466 ± 35.3	314–531
Seymour	228 ± 30.0	145–288	132 ± 65.9	189–205	104 ± 15	64–142	127 ± 14.0	77–156
Dissolved oxygen (DO) mg/L								
Adelaide	1.6 ± 0.7	0.2–4.5	7.9 ± 0.2	7.2–8.5	10.5 ± 0.7	8.7–13.1	8.9 ± 0.2	8.3–9.9
Alice	1.9 ± 0.90	0.6–6.6	6.8 ± 1.0	4.6 ± 11.2	8.5 ± 0.3	7.5–10.0	7.6 ± 0.6	5.4–9.4
Bedford	1.5 ± 0.4	0.2–3.0	6.8 ± 1.7	6.8–14.1	NS	NS	NS	NS
Berlin	4.4 ± 0.4	3.0–5.3	8.0 ± 0.9	6.0–9.5	NS	NS	8.3 ± 0.3	6.9–9.0
Seymour	3.6 ± 1.1	0.7–6.1	6.8 ± 2.3	6.7–7.0	8.8 ± 0.2	8.3–9.5	8.3 ± 0.5	6.0–9.05

NS no sample

sampling points in this present study, the difference was not statistically significant. The average temperature across the weather seasons also fell within the acceptable limit of  $\leq 25$  °C for domestic water set by South Africa (DWA 1996a) except during summer where temperatures reached their climax and were above 30 °C for all the plants except at Seymour. It is expected that the rise in temperature will normalize within a short time, especially at nights; otherwise, it may have a detrimental effect on the aquatic metabolic rate and some other physicochemical parameters if it persists (Fondriest Environmental 2014a).

## pH

Another important indicator of water quality, pH, influences biological operations in the treatment plant. The pH values in the influent and the final effluent samples in all the treatment plants varied at 6.8–8.8 and 6.5–9.1, respectively

(Table 2). A similar trend was observed in the receiving water bodies at 6.7–9.4 and 6.8–9.2 for the upstream and downstream, respectively. There was no significant difference in pH across the sampling points, and also among the five treatment plants ( $p > 0.05$ ).

At Adelaide (AS), the pH values at both ends of the receiving water were sometimes as high as 9.0 compared with others just as we observed similar high values in the effluent at the Berlin (TF) plant. The industrial wastes that are being discharged in Berlin may have been responsible for the high pH values in these areas. This suggests the dependence of effluent composition on the type of household, business, industries and the level of treatment received by the wastewater and public facilities discharging into the water system (Environment 2001). The average pH for all WWTPs fell within the permissible limit of 6–9 for domestic and recreational water in South Africa (DWA 1996a, b). These values were also similar to pH values reported by some





researchers in the previous studies for rivers and wastewater effluent in South Africa (Fatoki et al. 2003; Igbinosa and Okoh 2009; Odjadjare and Okoh 2010). This suggests that effluent from WWTPs mainly from domestic wastes may not be acidic after all. However, industrial effluent may bring about a fluctuation in pH characteristics. Decomposition of organic matter interactions of water with rocks and acid rain are other factors that may greatly influence pH (Fondriest Environmental 2013; Choksi et al. 2015). Similar pH fluctuations have been reported by some authors (Ogunfowokan et al. 2005; Akan et al. 2008; Chigor et al. 2013) both in southwest Nigeria and the Eastern Cape, South Africa.

### Electrical conductivity (EC)

EC measures the amount of dissolved ions or total salt content in water and is a useful indicator of salinity (Fondriest Environmental 2014b). The EC of the influent and final effluents varied at 218–1030 and 218–587  $\mu\text{S}/\text{cm}$ , respectively, for all the treatment plants except for TF (Berlin) with appreciably higher effluent values 557–781  $\mu\text{S}/\text{cm}$  (Table 2). This probably suggests that the nature of industrial waste in this area (Berlin) is more of dissolved ions. On the average, there was a significant decrease in EC values across the sampling points and among the treatment plants ( $p < 0.05$ ), as the wastewater passed through the treatment processes but not statistically significant across the seasons. The relative difference in EC observed among WWTPs may suggest different geochemical effects, types of agricultural runoffs, the relative different amount of chlorine disinfectant received during treatment and constituents of the sewage that are being released (Fondriest Environmental 2014b).

The average EC in the final effluent at Adelaide, Alice and Seymour WWTPs that operated AS technology is relatively higher than the EC values in their respective upstream. This may suggest a notable influence on the downstream water quality. The biodegradation of the organic matter load in the wastewater by microorganisms could probably have increased the amount of total dissolved solids (TDS), which has a strong correlation with EC (Table 8). Generally, the final effluent increased EC in the downstream of the receiving water. However, there are possibilities that the upstream water could also have found other pollution point source for dissolved ions apart from effluent discharge. These values are nevertheless, within the acceptable limit recommended by DWAF for domestic water supply of 700  $\mu\text{S}/\text{cm}$  (DWAF 1996b). These values are in agreement with those reported by Igbinosa and Okoh (2009) as well as Chigor et al. (2013) but are lower than those reported by Fatoki et al. (2003) and Odjadjare and Okoh (2010) obtained from Keiskamma River water and from urban areas of South Africa.

### Dissolved oxygen (DO)

The DO in the influent and effluent of all the WWTPs generally varied between 0.2 and 6.6 (average  $1.6 \pm 0.7$ – $4.4 \pm 0.4$  mg/L) and from 4.6 to 14.1 mg/L (mean  $6.8 \pm 1.0$ – $8.0 \pm 0.9$ ), respectively (Table 2). There was no significant difference in DO between the final effluent and the receiving water for any of the WWTPs ( $p > 0.05$ ). The respective DO for upstream and downstream also varied between 7.6 and 10.5 mg/L on the average. This probably suggested a significant reduction in the organic load of the discharged effluent. There was a notable increase in DO as the wastewater passed through the treatment plants suggesting a possibility of increased dissolution of atmospheric oxygen during influent agitation under treatment. The microbial activities and other treatment processes that may have reduced the organic matter in the wastewater would have, as a consequence, increased availability of DO. The green algal growth that is usually observed in the final effluent during storage with OP at Bedford may probably have been as a result of increased DO (which was as high as 14.1 mg/L) compared with other effluents from other plants. Though, this effluent is not released to any water body.

Throughout the sampling periods, the DO in both the upstream and downstream fell within the acceptable standard limits of (5–9 mg/L) for aquaculture in South Africa (DWAF 1996a). The DO for the final effluent of all the selected WWTPs was also above the minimum benchmark of 5 mg/L. DO below 2 mg/L may affect fishes more by impacting negatively on their feeding regime, reproductive behavior, swimming ability and may lead to death if persistent (Fondriest Environmental 2014a). The results for DO are similar to values reported by other authors in South Africa.

### Total dissolved solids (TDS)

TDS and EC are two parameters that correlate positively and are also indicators of water salinity. TDS is a measure of ion particles smaller than 2 microns, which includes organic solutes such as hydrocarbon and urea apart from salt ions, and also helps aquatic life to balance their cell density (Fondriest Environmental 2014b). The average TDS in the influent for Adelaide, Alice and Bedford are  $309 \pm 20$ ,  $197 \pm 12$ ,  $343 \pm 70$  mg/L, respectively, while Berlin recorded  $381 \pm 42.5$  mg/L and Seymour  $228 \pm 30$  mg/L (Table 2). It varied generally from 145 to 650 mg/L for the five WWTPs. There was a significant reduction ( $p < 0.05$ ) in TDS as the influents passed through the treatment processes to the final effluent, suggesting a notable influence on the wastewater. This observation was different in TF at Berlin, where a slight increase in the average TDS was observed, from 381 mg/L in the influent to 385 mg/L in the final effluent. The geological difference and variation in influent constituents (especially



with Berlin which receives industrial wastes, Table 1) may be attributed to the significant difference observed in TDS among the plants ( $p < 0.01$ ). The high flow rate of influent observed with Berlin (3000 m<sup>3</sup>/day) and Bedford (3500 m<sup>3</sup>/day) (which harbor the second largest abattoir in the Eastern Cape) may have also increased the amount of dissolved solids in their respective effluents. The average values of TDS downstream of the receiving water bodies were also lower than discharged effluents from the WWTPs except in TF at Berlin. This may suggest other soluble particles along its path as the effluent flows for a longer distance before it finally reaches the receiving water, unlike other effluents. Generally, none of the TDS values in the final effluent and receiving water recorded in this work exceeded the South African stipulated standard of 0–450 mg/L for domestic water (DWA 1996b) or the USEPA maximum contaminant level of 500 mg/L (USEPA 2009). Elevated TDS could be injurious to freshwater animals through the obstruction of the osmoregulation ability of the organism (McCulloch et al. 1993).

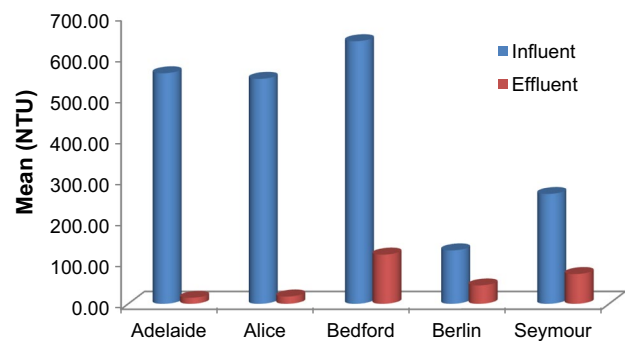
**Turbidity**

Turbidity correlates significantly with microbial growth in water (Fondriest Environmental 2014c). It is one parameter that provides a platform to measure the ability of a WWTP to reduce suspended organic and inorganic loads through its processes. The values of turbidity in the influent, effluent and receiving waters including the percentage reduction in all the WWTPs are presented in Table 3.

The influent values varied between 165–988 NTU, at Adelaide and 169–980 NTU in Alice, while the results in Bedford, Berlin and Seymour were in the range of 165–997 NTU, 30–241 NTU and 149–414 NTU, respectively. The effluent values varied as 1.9–88 NTU for Alice, Adelaide and Berlin plants, and this was reflected in their removal efficiencies of

97, 97 and 83%, respectively (Fig. 1). The effluent values in Seymour (AS) and Bedford (OP) were notably higher (46–183 NTU) and, respectively, recorded 73 and 81% removal. This shows a significant reduction ( $p < 0.01$ ) between turbidity in the influent and the final effluent for all the WWTPs, but there was less influence of seasonal variation ( $p < 0.05$ ).

The nature and chemistry of the wastewater discharge catchment from which Berlin influent is sourced are likely more of dissolved component than the light scattering suspended component measured in turbidity, hence the reason for the lowest values recorded in this area. In addition, this suggestion was evidently supported by the highest values of EC, TDS and DO record at Berlin. The turbidity of the daily inflow was mostly very high at Bedford, Alice and Adelaide probably due to the high levels of anthropogenic activities caused by large populations and high flow rate of influents. Berlin, with a high flow rate of influents occasioned by industrial and domestic waste, was an exception to this trend. Some of the operational conditions of the WWTPs that could have assisted in a deeper interpretation of the results are either not easily accessible or not available



**Fig. 1** Average removal capacities of WWTPs for turbidity

**Table 3** Turbidity (NTU) and total suspended solids (TSS) mg/L at different sampling points of different WWTP

WWTP	Influent	Range	Effluent	Range	Red. (%)	Upstream	Range	Downstream	Range
<b>Turbidity (NTU)</b>									
Adelaide	560.3 ± 134.7	165–988	14.6 ± 5.5	1.9–29.1	97	183.2 ± 56.1	51–417	150.5 ± 55.3	48–417
Alice	546.7 ± 136.2	169–980	17.8 ± 6.9	2.2–38	97	111 ± 23.3	43–189	106.6 ± 23.6	34–181
Bedford	637.7 ± 134.0	165–997	119.1 ± 18.9	50–174	81	NS	NS	NS	NS
Berlin	129.4 ± 36.2	30–241	44.8 ± 21.3	3–88	83	NS	NS	129.6 ± 29.7	58–241
Seymour	266.9 ± 62.4	149–414	72.5 ± 45.2	46–183	73	136.4 ± 41.7	6–286	230.3 ± 72.3	74–551
<b>Total suspended solids (TSS) mg/L</b>									
Adelaide	195.1 ± 27.3	134–303	4.4 ± 1.0	2.3–7	98	48.5–19.9	11–115	41.8 ± 19.1	11–115
Alice	179.9 ± 36.5	106–311	6.7 ± 2.6	3–15.3	96	27.5 ± 9.5	5–49	21.3 ± 6.6	6–41
Bedford	184.9 ± 18.8	122–221	57.4 ± 10.8	22–87	69	NS	NS	NS	NS
Berlin	49.1 ± 17.5	10–91	20.3 ± 11.1	7–43	75	NS	NS	40.2 ± 11.5	17–79
Seymour	79.7 ± 13.0	66–106	15.5 ± 6.4	15–16	87	35.3 ± 13.1	3–69	49.1 ± 14.	17–83

NS no sample



with some WWTPs. This is probably a consequence of the limiting level of operational standard available in some rural areas. The turbidity values for the final effluent reported in this study were similar to those in other works reported in South Africa (Odjadjare and Okoh 2010; Chigor et al. 2013). Figure 1 shows different removal capacities for removal of turbidity in WWTPs.

It is worthy of note that high turbidity values usually observed in the final effluent in OP at Bedford suggested a cumulative effect of algae growth and other microbial activities that prevailed during effluent storage. The lower efficient capacity also observed with AS technology at Seymour as compared to other AS may be attributed to the single tank technology which was used and which limited the numbers of clarification tanks. The turbidity values in the receiving water for all the WWTPs were notably higher than the effluent. This may be as a result of other anthropogenic activities and agricultural runoff which also influenced the turbidity of the river. Highly turbid water tends to aid the growth of pathogens thereby increasing the chances of infection (Obi et al. 2007). The values in the receiving water exceeded both the South African target range of 0 to 1 NTU and the World Health Organization value of  $\leq 5$  NTU (DWA 1996b; WHO 2004). However, the effluent discharged at Adelaide, Alice (using AS) and Berlin (using TF) sometimes fell within the permissible range. This may naturally not have qualified the surface water as fit for domestic use with respect to turbidity because the values could increase the chances of disease transmission through microorganism associated with the particulate matter. A highly turbid effluent reduces disinfection potential of chlorine (DWA 1996b).

### Total suspended solids (TSS)

The TSS values at Adelaide, Alice and Bedford in the influent varied from 134 to 303, 106 to 311, 122 to 221 mg/L, respectively, while in Berlin, it varied between 10 and 91 mg/L, and in Seymour between 66 and 106 mg/L (Table 3). Relatively low TSS values were observed in the influent at Berlin (TF) and Seymour (AS). Generally, there was a significant removal of TSS in the final effluent of all the WWTPs (Fig. 2). Adelaide and Alice operating AS recorded over 95% reduction, while Seymour (AS), Berlin (TF) and Bedford (OP) had 87, 75 and 69%, respectively. The high TSS values both upstream and downstream of the receiving water varied between 3 and 115 mg/L for all the WWTPs. This also suggests other anthropogenic activities along the course of the river and surface runoff that may be occasioned by rain.

The variations of TSS values across the treatment processes were statistically significant ( $p < 0.01$ ). However, the final effluent showed no additional impact on the receiving water. TSS is a parameter that indicates water clarity and usually is constituted by inorganic materials but may

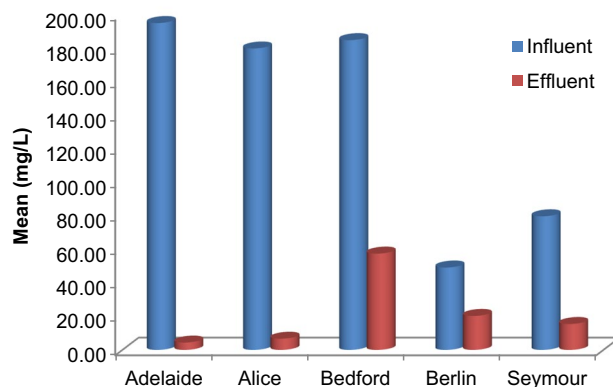


Fig. 2 Average removal capacities of WWTPs for TSS

include bacterial and algae solids found in the water column larger than 2 microns (Fondriest Environmental 2014c). The WWTP effluent values fall within the purview of the South African tolerable limit for TSS in domestic water (25 mg/L), but the same cannot be said of effluent from WWTP in Bedford (OP) and the respective receiving waters.

### Occurrence and fate of phenolic compounds in different WWTPs at Adelaide, Alice, Bedford and Berlin in Eastern Cape, South Africa

The mean concentrations of detected phenolic compounds at different sampling points and receiving water body, the frequency of detection and the percentage reduction at Adelaide, Alice, Bedford and Berlin are presented in Tables 4, 5, 6 and 7, respectively. The wastewater from these areas is largely domestic wastes and agricultural runoffs as there was no reported industrial waste except in Berlin. For the period of sampling, all the nine phenolic compounds were detected in the influent except phenol in Alice and Berlin. However, 4-C-3MP, 2,4DCP, 2-NP, PCP and 2,4,6-TCP were the compounds found in abundant at varying concentrations in all the WWTPs. The most abundant compound across the treatment plants except Berlin was 4-C-3MP with a mean concentration of  $42.1 \pm 17.7$   $\mu\text{g/L}$ ,  $35.3 \pm 10.6$  and  $22.5 \pm 11.1$   $\mu\text{g/L}$  in Adelaide, Alice and Bedford, respectively, followed by 2-NP. At Berlin, 2-NP with an average concentration of  $40.3 \pm 21.2$  was the most abundant. Others were of minor importance in the influent with concentrations usually below 10  $\mu\text{g/L}$ . There was a notable reduction in the compounds in the final effluent suggesting high removal capacity of the WWTP. The average concentration in the final effluent varied between not detected (ND) and 5.3  $\mu\text{g/L}$ . However, this concentration is usually exceeded in OP at Bedford. The removal capacities of all the treatment plant, which varied generally between 33 and 96%, suggest adsorption of the compounds on settling particles as we found a

**Table 4** Concentration of phenolic compounds in the wastewater from WWTP in Adelaide (Mean  $\pm$  SE  $\mu\text{g/L}$ )

Compounds	Influent	Range	FD (%)	Effluent	Range	Red. (%)	Upstream	Range	Downstream	Range
PH	3.2 $\pm$ 2.1	ND–9.5	75	1.8 $\pm$ 1.0	ND–3.9	42	1.5 $\pm$ 0.6	ND–2.5	1.2 $\pm$ 0.6	ND–2.4
2-CP	6.7 $\pm$ 1.4	3.8–10.2	100	2.5 $\pm$ 0.5	1.0–3.6	63	2.4 $\pm$ 0.6	0.8–3.9	2.2 $\pm$ 0.5	0.8–3.2
2,4-DMP	7.5 $\pm$ 1.8	4.2–12.5	100	2.7 $\pm$ 0.3	2.2–3.2	82	4.0 $\pm$ 0.6	3.2–4.8	2.5 $\pm$ 0.6	1.6–3.4
4-C-3MP	42.1 $\pm$ 17.7	9.8–83.0	100	2.5 $\pm$ 0.7	ND–3.9	96	2.8 $\pm$ 0.7	ND–4.0	2.1 $\pm$ 1.1	ND–4.3
2,4-DCP	13.8 $\pm$ 8.3	2.1–37.3	100	2.3 $\pm$ 0.5	1.6–3.0	92	0.8 $\pm$ 0.4	ND–1.7	1.1 $\pm$ 0.4	ND–1.7
2-NP	18.0 $\pm$ 7.1	6.5–36.8	100	3.7 $\pm$ 1.5	1.8–8.3	79	2.9 $\pm$ 0.4	1.7–3.6	3.2 $\pm$ 0.8	1.3–5.1
2,4,6-TCP	9.3 $\pm$ 2.0	5.2–13.7	100	6.2 $\pm$ 4.3	1.0–19.1	34	4.3 $\pm$ 2.2	1.1 $\pm$ 10.7	4.7 $\pm$ 2.9	0.9–13.1
4-NP	7.4 $\pm$ 3.2	1.3–14.0	100	0.9 $\pm$ 0.2	ND–1.1	91	1.6 $\pm$ 0.9	0.5–4.1	6.4 $\pm$ 3.6	1.3–11.5
PCP	12.8 $\pm$ 1.9	10.4–18.5	100	2.4 $\pm$ 0.7	ND–3.7	86	2.4 $\pm$ 0.7	ND–3.1	4.5 $\pm$ 1.6	ND–6.7

PH phenol, 2-CP 2-Chlorophenol, 2,4-DMP 2,4-Dimethylphenol, 4-C-3MP 4-Chloro-3-methylphenol, 2,4-DCP 2,4-Dichlorophenol, 2-NP 2-Nitro-phenol, 2,4,6-TCP 2,4,6-Trichlorophenol, 4-NP 4-Nitrophenol, PCP Pentachlorophenol, Red. Reduction, FD Frequency of detection, ND not detected (below limit of detection 0.102–0.586  $\mu\text{g/L}$ ), SE standard error

**Table 5** Concentration of phenolic compounds in the wastewater from WWTP in Alice (Mean  $\pm$  SE  $\mu\text{g/L}$ )

Compounds	Influent	Range	FD (%)	Effluent	Range	Red. (%)	Upstream	Range	Downstream	Range
PH	ND			ND			1.0 $\pm$ 0.5	ND–1.9	1.3 $\pm$ 0.7	ND–2.7
2-CP	3.6 $\pm$ 1.3	1.8–6.1	100	1.1 $\pm$ 0.3	0.2–1.7	60	2.0 $\pm$ 0.3	1.2–2.8	2.6 $\pm$ 0.4	1.6–3.2
2,4-DMP	22.7 $\pm$ 12.0	3.3–44.6	100	1.1 $\pm$ 0.5	0.2–2.4	93	1.3 $\pm$ 0.6	0.2–2.8	1.6 $\pm$ 0.7	0.5–3.1
4-C-3MP	35.3 $\pm$ 10.6	22.1–56.1	100	2.7 $\pm$ 0.6	ND–3.9	92	3.6 $\pm$ 1.1	ND–5.6	4.3 $\pm$ 0.8	ND–5.8
2,4-DCP	4.0 $\pm$ 1.4	2.3–5.7	100	1.1 $\pm$ 0.1	0.9–1.3	74	1.1 $\pm$ 0.1	0.9–1.1	1.5 $\pm$ 0.1	1.3–1.6
2-NP	11.6 $\pm$ 3.7	7.0–18.6	100	4.0 $\pm$ 0.7	2.8–5.9	55	3.8 $\pm$ 0.8	2.0–5.7	6.7 $\pm$ 2.2	3.3–13.2
2,4,6-TCP	25.4 $\pm$ 7.3	ND–34.3	75	1.6 $\pm$ 0.5	0.6–3.0	87	2.3 $\pm$ 0.4	1.4–3.3	4.9 $\pm$ 2.3	1.3–11.5
4-NP	4.2 $\pm$ 0.4	3.4–4.9	100	0.8 $\pm$ 0.2	ND–1.0	82	2.28 $\pm$ 1.0	1.6–5.0	2.7 $\pm$ 1.2	1.3–5.5
PCP	20.3 $\pm$ 8.9	6.1–36.7	100	5.3 $\pm$ 1.3	ND–7.5	74	2.4 $\pm$ 0.5	1.5–3.7	4.9 $\pm$ 8.3	ND–8.3

PH phenol, 2-CP 2-chlorophenol, 2,4-DMP 2,4-dimethylphenol, 4-C-3MP 4-chloro-3-methylphenol, 2,4-DCP 2,4-dichlorophenol, 2-NP 2-nitrophenol, 2,4,6-TCP 2,4,6-trichlorophenol, 4-NP 4-nitrophenol, PCP pentachlorophenol, RED. reduction, FD frequency of detection, SE standard error, ND not detected (below limit of detection 0.102–0.586  $\mu\text{g/L}$ )

**Table 6** Concentration of phenolic compounds in the wastewater from WWTP in Bedford (Mean  $\pm$  SE  $\mu\text{g/L}$ )

Compounds	Influent	Range	FD (%)	Effluent	Range	Reduction (%)
PH	2.2 $\pm$ 1.6	ND–5.5	75	1.1 $\pm$ 0.6	ND–2.4	49
2-CP	4.5 $\pm$ 0.6	ND–5.6	75	1.6 $\pm$ 0.7	ND–2.7	65
2,4-DMP	8.2 $\pm$ 1.5	5.6–12.2	100	2.7 $\pm$ 0.4	1.8–3.3	75
4-C-3MP	22.5 $\pm$ 11.1	6.4–55.1	100	2.4 $\pm$ 1.6	ND–5.5	92
2,4-DCP	7.0 $\pm$ 3.6	1.8–15.2	100	1.5 $\pm$ 0.6	ND–2.4	78
2-NP	17.3 $\pm$ 5.6	5.7–32.3	100	2.9 $\pm$ 0.9	ND–4.8	87
2,4,6-TCP	7.6 $\pm$ 0.7	6.3–9.4	100	6.8 $\pm$ 4.4	0.9–19.5	11
4-NP	6.5 $\pm$ 2.1	2.9–11.7	100	8.7 $\pm$ 5.9	1.7–22.6	–0.5
PCP	8.2 $\pm$ 1.3	4.9–10.6	100	5.3 $\pm$ 1.2	ND–6.5	68

PH phenol, 2-CP 2-chlorophenol, 2,4-DMP 2,4-dimethylphenol, 4-C-3MP 4-chloro-3-methylphenol, 2,4-DCP 2,4-dichlorophenol, 2-NP 2-nitrophenol, 2,4,6-TCP 2,4,6-trichlorophenol, 4-NP 4-nitrophenol, PCP pentachlorophenol, RED. reduction, FD frequency of detection., ND not detected (below limit of detection 0.102–0.586  $\mu\text{g/L}$ ), SE standard error

strong correlation between TSS, turbidity and removal of phenolic compounds (especially the ones in abundance); see Table 8. Apart from adsorption on settling particles, other possible removal pathways for organic pollutants in WWTPs

may include: biological or chemical degradation, transformation and volatilization during aeration. However, some other authors argued that the removal of majority of these compounds is through biodegradation (Zhao et al. 2009;

**Table 7** Concentration of phenolic compounds in the wastewater from WWTP in Berlin (mean  $\pm$  SE  $\mu\text{g/L}$ )

Compounds	Influent	Range	FD (%)	Effluent	Range	Red (%)	Downstream	Range
PH	1.5 $\pm$ 0.5	ND–2.1	67	ND		100	ND	
2-CP	2.0 $\pm$ 1.2	0.2–4.3	100	1.4 $\pm$ 0.5	0.4–2.2	33	1.7 $\pm$ 0.8	0.3–2.9
2,4-DMP	11.5 $\pm$ 6.5	1.1–23.3	100	1.0 $\pm$ 0.8	0.1–2.5	91	1.0 $\pm$ 0.7	0.2–2.4
4-C-3MP	18.6 $\pm$ 9.6	2.8–36.0	100	2.6 $\pm$ 1.0	1.1–4.5	86	2.3 $\pm$ 0.7	1.5–3.7
2,4-DCP	2.8 $\pm$ 1.5	1.0–5.8	100	1.3 $\pm$ 0.0	ND–1.3	85	0.5 $\pm$ 0.3	0.1–1.1
2-NP	40.3 $\pm$ 21.1	1.7–74.5	100	2.4 $\pm$ 0.9	1.1–4.1	94	2.8 $\pm$ 0.8	1.3–4.1
2,4,6-TCP	12.0 $\pm$ 6.5	2.5–24.4	100	2.1 $\pm$ 0.6	1.2–3.2	83	2.2 $\pm$ 0.6	1.1–3.2
4-NP	2.5 $\pm$ 1.4	0.8–5.1	100	0.8 $\pm$ 0.3	0.3–1.3	67	1.3 $\pm$ 0.5	0.3–1.8
PCP	31.8 $\pm$ 21.7	1.7–73.9	100	3.1 $\pm$ 1.2	0.8–4.8	90	3.3 $\pm$ 1.0	1.3–4.5

PH phenol, 2-CP 2-chlorophenol, 2,4-DMP 2,4-dimethylphenol, 4-C-3MP 4-chloro-3-methylphenol, 2,4-DCP 2,4-dichlorophenol, 2-NP 2-nitro-phenol, 2,4,6-TCP 2,4,6-trichlorophenol, 4-NP 4-nitrophenol, PCP pentachlorophenol, Red. reduction, FD frequency of detection, ND not detected (below limit of detection 0.102–0.586  $\mu\text{g/L}$ ), SE standard error

Gao et al. 2014). There was no significant difference in the concentrations of any of the compounds ( $p > 0.05$ ) between the final effluent and the receiving water bodies, suggesting no significant impact by the effluent on the river. The concentration of 4-NP detected in the final effluent in Bedford was higher than what was obtained in the influent suggesting that some compounds may have escaped degradation by microorganisms or converted to biomass (sludge); hence, they may be released with the effluent (Wang et al. 2005).

The concentration of phenolic compounds both in the final effluent and receiving water for all the treatment plants was below the tolerable limits of 5  $\mu\text{g/L}$  set by USEPA and the European Union for bathing water (domestic use) (Petrovic et al. 2009). It is worthy to note that the concentrations of 2,4,6-TCP, 2-NP and 4-NP were occasionally above this limit in the effluent and receiving water. The prominent phenolic compounds detected in these areas were similar to the compounds detected in Cape Town, South Africa, which include 2-NP, 4-C-3MP, PCP and 2,4-DMP (Olujimi et al. 2012). Michalowicz and Duda (2007) identified nitrophenols and methyl-phenols as common products of vehicle emissions that find their way to the environment. The relatively low amounts of phenolic compounds detected in this work can in no way be compared to a similar investigation carried out in Cape Town by Olujimi et al. (2012) from sewers receiving effluents from industries that use these chemicals. The phenolic compounds reported in the final effluent and downstream of the investigated WWTPs in Cape Town were in the range of 30–200  $\mu\text{g/L}$ . However, the high population reported in these areas (133,000–900,000) would have increased the anthropogenic activities in the area apart from the presence of industries. This suggests that these compounds are more likely to be introduced to surface water from industries that use these chemicals than from domestic and agricultural runoff. It could, therefore, be inferred that these meagre amounts in the influent and receiving water bodies could probably be the result of decomposition or

leachate from the use of products containing these compounds, for example, plastics, dyes, drugs, pesticides, paper and petrochemical products (Zhou et al. 2005). Pesticides containing phenolic compounds and vehicle emissions are probably the commonest contributors of these compounds in these areas (Michalowicz and Duda 2007). Sim et al. (2009), also suggests that phenolic compounds might occur naturally through degradation of lignin, tannin and humic substances. This investigation, therefore, conclude that the relative amounts of phenolic compounds that are generated in these parts of the rural areas were not above the removal capacity of the existing WWTPs.

## Conclusion

The five selected WWTPs processes from different technologies impacted considerably on the raw wastewater to produce a final effluent that met acceptable South Africa standards in terms of EC, DO, TDS, turbidity and TSS. However, standard limits for discharged effluents were occasionally exceeded especially with some WWTPs. The discharged effluent may not have impacted on the total suspended solids (TSS) in the receiving water; nevertheless, there is evidence of the negative impact of some physicochemical properties measured in terms of TDS, EC and turbidity depending on the geological difference and the constituents of the treated effluent. This poses health challenges to rural communities that rely heavily on surface water for domestic use. However, the four phenolic compounds 4-C-3MP, 2-NP, 2,4-DCP and PCP mostly detected in the wastewater were notably reduced by the WWTPs. This is not enough reason to overestimate the performance of the plants judging by the meagre amounts that were detected entering the WWTPs. Despite that, the study also revealed evidence of phenolic contaminants in some receiving water. It is, therefore, pertinent for concerned authority responsible for WWTPs and

**Table 8** Pearson correlations of physicochemical properties with phenolic compounds

	pH	TEMP	E.C	DO	TDS	TUR	TSS	PH	2-CP	2,4-DMP	4-C-3MP	2,4-DCP	2-NP	2,4,6-TCP	4-NP	PCP
pH	1															
TEMP	-0.198	1														
E.C	0.230	-0.077	1													
DO	0.113	-0.364	-0.415	1												
TDS	0.218	-0.095	0.973**	-0.402	1											
TUR	0.126	0.052	0.237	-0.293	0.158	1										
TSS	0.153	-0.034	-0.178	-0.181	-0.258	0.872**	1									
PH	-0.256	0.303	-0.618	-0.135	-0.578	-0.189	0.562	1								
2-CP	-0.042	0.156	0.103	-0.298	0.130	0.846**	0.708*	-0.161	1							
2,4-DMP	-0.027	0.249	-0.319	0.316	-0.361	-0.192	-0.083	0.028	-0.201	1						
4-C-3MP	0.146	0.105	0.055	-0.543	0.036	0.555	0.705*	0.707	0.607*	-0.218	1					
2,4-DCP	0.226	0.186	-0.256	-0.177	-0.222	-0.237	-0.021	0.641	-0.355	0.268	0.389	1				
2-NP	-0.228	-0.104	-0.172	-0.050	-0.165	0.573	0.408	0.604	0.501	-0.007	0.531	0.049	1			
2,4,6-TCP	-0.154	0.109	-0.376	0.305	-0.362	-0.057	0.066	-0.112	0.116	0.765**	-0.323	-0.201	-0.071	1		
4-NP	0.007	0.224	-0.248	-0.028	-0.269	-0.005	0.219	-0.183	0.199	0.356	-0.100	-0.249	-0.394	0.673*	1	
PCP	-0.242	0.121	-0.074	-0.369	-0.060	0.760**	0.648*	0.787*	0.735**	-0.245	0.630*	-0.081	0.744**	-0.027	-0.148	1

\*\*\*Correlation is significant at the 0.01 level (2-tailed)

\*Correlation is significant at the 0.05 level (2-tailed)

environmental safety in South Africa to fashion out a routine monitoring of the performance of these WWTPs and ensure compliance with standard practices. With this, abnormal fluctuations can easily be detected and addressed promptly for a safe environment for aquatic life and domestic use.

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