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The Effect of Hydraulic Loading Rates on Nitrogen Removal by Using a Biological Filter Proposed for Ventilated Improved Pit Latrines

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ABSTRACT: No on-site sanitation system treats both urine and faecal matter in one process. A laboratory scale biological filter was fed with high concentration of urea (4 g N/L) and 17.1 g COD/L to determine if it will be possible to treat liquid that leach from a ventilated improved pit latrine. The HLR in the proposed biological filter system was calculated to be *ca* 36 L/m^2/d, significantly lower than the rates that are typical applied in standard rate biological filters (in the range of 1000 – 4000 L/m^2/d) used to treat domestic wastewater. However, the TKN and COD concentrations in standard rate biological filters are significantly lower, namely *ca* 60 mg N/L and 500 mg COD/L, compared to the typical nitrogen and COD concentrations of faecal sludge, namely 3 - 5 g N/L and 20 – 50 g COD/L, respectively. The biological filter was operated at 13.0, 23.9, 35.7 and 62.3 L/m^2/d, until stable state conditions were obtained. It was possible to remove most of the nitrogen and COD at the applied hydraulic loading rates by a combination of volatilization, nitrification and de-nitrification processes. However, at 62.3 L/m^2/d the column efficiency (1.5 m long column) decreased and ammonia concentration in the effluent increased again. The best performance was achieved at a hydraulic loading rate of 35.7 L/m^2/d, with an average ammonia concentration of 285.5 (± 9.1) mg N/L.

Key words: Ammonia volatilization, Biomass assimilation, De-nitrification, Nitrification,Urea

INTRODUCTION

In many developing countries, pit latrines are the sanitation system mostly used. Depending on the permeability of the soil, liquid will leach from a pit latrine into the surroundings and can contaminate groundwater in the vicinity (Dzwairo *et al.*, 2006). Faecal sludge from pit latrines contains high concentrations of nitrogen and organic matter 2 - 5 g N/L and 20 – 50 g COD/L, respectively (Cofie *et al*., 2006). Due to natural biological processes that occur, the composition of the biodegradable nitrogen and organic compounds in the leachate, will change as it moves through the soil. Research has shown elevated concentrations of nitrogen species in surrounding groundwater (Zingoni *et al*., 2005; Dzwairo *et al.*, 2006) and nitrogen in the form of nitrate will enter the groundwater and pose a serious health risk to the users of the water source (Suthar *et al*., 2009). Nitrate is especially dangerous to babies younger than 6 months and immuno-compromised individuals as it causes a condition called methaemoglobinaemia and is possibly also carcinogenic (Suthar *et al.*, 2009). In very remote areas when the pit latrine is full, the pit is closed and the content is left in the soil, where depending on the soil conditions, it will become stable in time. In permeable soil the liquid will seep away very fast and the pit will be dry, on the other hand, if the soil permeability is low the liquid will be contained for a longer period. Because of this, anaerobic conditions prevail and the excreta decompose quicker than in dry pits. During the anaerobic digestion of the human excreta the BOD is reduced by microbial action, decreasing the pollution potential of organic matter. However, due to the absence of nitrification under anaerobic conditions, the nitrogen will mainly be present in the form of ammonia. This increases the risk of groundwater pollution through seepage. On the other hand in more densely populated areas the pit is emptied once a year with a vacuum tanker. However, this practice is costly because a fleet of vacuum tankers is required, which has to be maintained. Another concern is that a suitable disposal and faecal sludge treatment facility is required. Faecal sludge is frequently treated in anaerobic ponds, which cause odour problems, followed by

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facultative ponds where both aerobic and anaerobic conditions exist. However, high ammonia concentrations suppress algal growth which limits the use of facultative ponds (Strauss *et al*., 2000). Alternative treatment options include sedimentation and thickening, drying beds, as well as wetlands (Ingallinella *et al.*, 2002). However all of the above mentioned treatment options are off-site and has cost implications. On-site treatment options seem to be lacking with no existing treatment for both urine and faecal matter in the same system. An example, where some intervention takes place is the composting pit latrine, where the faecal matter is composed and the urine is separately collected to be used as fertilizer or allowed to soak away (Von Münch and Mayumbelo, 2007). A need exist for a low cost, on-site treatment system, where both faecal matter and urine can be treated. A new modified VIP system is proposed, where the faecal sludge in the pit will be allowed to leach into a biological filter. The filter will be packed with stones. The vent pipe will be fitted with an extraction fan to induce a draft through the filter. Thus, all the liquid that seeps from the top part of the pit will be treated in the biological filter. The aim of this study is to determine whether the biological filter will be able to process the required nitrogen and organic load at the HLRs that will be applied. Faecal sludge comprises of a liquid and solid phase. The liquid phase mainly originate from urine. Between 600 - 1200 m? urine is excreted per person per day (Schouw *et al.*, 2002). The main factors that will have an influence on the amount excreted are the daily fluid intake and the climatic conditions. The number of persons that use a pit latrine during a day will also vary. These variations in the hydraulic load can have an effect the performance of any treatment process (MetCalf and Eddy, 2003). The HLRs for standard rate biological filters are 1000 – 4000 L/m^2/d (MetCalf and Eddy, 2003). To determine the HLR on a pit latrine it was assumed that urine is produced at a rate of 1.5 L/capita/d, as indicated by Ronteltap *et al* (2010) as a global average. Assuming that a pit latrine will serve a family of six persons, and that the surface area of the biological filter underneath the pit will be 0.5 m x 0.5 m, then the HLR in the proposed biological filter system will be *ca* 36 L/m^2/d, significantly lower than the rates that are typical applied in standard rate biological filters used to treat domestic wastewater. However, the TKN and COD concentrations in standard rate biological filters are significantly lower, namely *ca* 60 mg N/L and 500 mg COD/L (MetCalf and Eddy, 2003), compared to the typical nitrogen and COD concentrations of faecal sludge, namely 2 - 5 g N/L and 20 – 50 g COD/L, respectively (Cofie *et al.*, 2006). In a biological filter the micro-organisms attach themselves to the filter media and form a biofilm. For treatment of wastewater to occur the substrate and oxygen have to be transported from the bulk of the liquid to the surface of the biofilm, from where it diffuses through the biofilm to where the substrate is oxidized and finally waste by-products are excreted back into the bulk liquid (Rauch *et al*., 1999; Mudliar *et al*., 2008). One of the factors which have an influence on the efficiency of the process is the HLR. The HLR is the volume of wastewater applied to a unit filter surface area (MetCalf and Eddy, 2003). The HLR will influence the distribution of the biofilm into the filterbed, and detachment of biofilm from the media (Eding *et al*., 2006; Morgenroth and Wilderer, 2000). Furthermore, the HLR will also influence the ability of a biological filter to perform nitrification. This is due to the fact that heterotrophic and autotrophic organisms have to compete for space in the bioflim. The heterotrophic organisms will replace the autotrophic organisms and no nitrification will take place. As the concentration of the organic compounds decrease in the column, the rate of nitrification will increase (Chen *et al.*, 2006). The C/N ratio determines how the heterotrophic and autotrophic organisms will be distributed through the filter (Chen *et al.*, 2006). At high HLRs, the heterotrophic organisms will be pushed deeper in the filter bed to such an extent that nitrification will be hampered (Grady *et al*., 1999).

The purpose of this investigation was to evaluate the effect of different HLRs on the proposed process.

MATERIALS & METHODS

The biological filter consisted of a PVC column, 150 mm in diameter and 1500 mm in length. The filter was packed with stones, 10-20 mm in diameter. The synthetic waste medium was fed intermittently to the biological filter by means of a diaphragm dosing pump (Model: Alldos M205). The pump was controlled by an electronic timer, which switched the pump on for 30 seconds, every hour. The medium was distributed through micro sprayers over the stones onto the biological filter (Fig. 1). Air was supplied by means of a compressor (Model: Fini, 100) through a network of perforated perspex pipes at a rate of 1Nm^3/h.

Fresh medium was prepared daily. Primary settling tank effluent from a domestic waste water treatment plant (COD was *ca* 250 mg/L; TKN *ca* 50 mg N/L; total P *ca* 10 mg P/L) was supplemented with urea, 8.6 g/L and glucose, 16 g/L to give a TKN concentration of 4 000 mg N/L and 17 100 mg COD/L. The pH was buffered to pH 7 with 1.8 g/L potassium di-hydrogen phosphate, and 2.8 g/L disodium hydrogen phosphate. The phosphate salts were also used to ensure sufficient phosphorous to sustain unrestricted microbial growth in the biological filter.

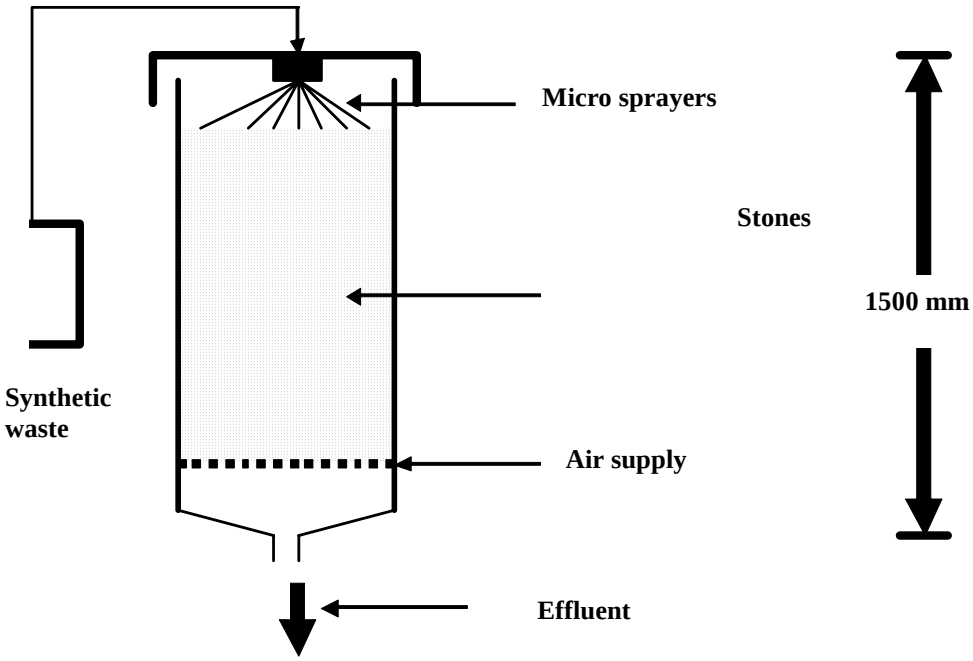

Fig. 1. Schematic layout of the biological filter

The biological filter was operated at the following dosing rates: 230 mL/d; 425 mL/d, 630 mL/d and 1100 mL/d. This is equal to hydraulic loading rates (HLRs) of 13.0 L/m^2/d; 23.9 L/m^2/d; 35.7 L/m^2/d and 62.3 L/m^2/d, respectively. When any operational parameters were changed the reactor was operated (under the new set of parameters) until stable state conditions were obtained. Stable state conditions were based on stable performance of the biological filter with regards to ammonia, nitrite and nitrate concentrations (three to four weeks of comparable results) in the effluent. The pH (Orion Model 410A) and conductivity (Hach Model Sension 6) of the effluent from the biological filter, as well as the volume throughput were determined daily in all the trials. The effluent from the different trials were also analysed at least once a week for COD, TKN, nitrate, nitrite and ammonia according to Standard Methods (APHA, 2005). When the reactor was operated at 23.9, 35.7 and 62.3 L/m^2/d at sample was also taken from the middle of the reactor once a week and analysed for the same parameters. At 13.0 L/m^2/d too little effluent was produced and sampling in the middle of the reactor was not done.

All chemicals used were of analytical grade and obtained from major retailers.

The estimated interval of the mean (t-estimate) was calculated at 95 % confidence level for the different parameters in the effluent from the different trials, at stable state conditions. Data Analysis Plus™ 2.12 an add-in for Microsoft Excell was used to perform the data analysis.The results are reported as $\bar{x} \pm 1.96 \frac{\sigma}{\sqrt{n}}$ (where $\bar{x}$ is the estimated mean, σ is the standard deviation and *n* the number of samples (Keller and Warrack, 2000).

RESULTS & DISCUSSION

Although air was supplied at a constant rate of 1 N/m^3/h, the low dosing rates resulted in liquid evaporated from the reactor. The average volume of effluent produced at a dosing rate 1100 mL/d (62.3 L/m^2/d) was 856 ± 52 mL/d. That represented a 22 ± 5 % loss in liquid. This loss of water was not very high and could be a result of water retained in the filter and water vapour formed as a result of the intermittent dosing conditions (30 seconds every hour). At a dosing rate of 630 mL/d (35.7 L/m^2/d) the effluent was produced at a rate of 398 ± 19 mL/d. The liquid loss was higher namely 34 ± 6 %. The liquid loss at a dosing rate of 425 L/d (23.9 L/m^2/d) was slightly higher, namely 38 ± 6% (effluent was produced at a rate of 264 ± 26 mL/d). A very large amount of liquid was lost at a dosing rate of 230 L/d (13.0 L/m^2/d) (effluent was produced at a rate of 47 ± 12 mL/d), namely 80 ± 5 %.

The evaporation resulted in a concentration of the effluent from the reactor at the different dosing rates, which was confirmed by the conductivity measure-

ments and concentrations of the different nitrogen species in the effluent from the reactor. This concentration effect was especially evident during the trial at the low HLR of 13.0 L/m^2/d. The reactor took approximately 43 days to become stable. During the first 43 days the average conductivity was 54.9 ± 14.8 mS/cm, while the average ammonia concentration was 4815 ± 1380 mg N/L. Thereafter the reactor was stable and the conductivity was 20.3 ± 2.5 mS/cm and the ammonia concentration reduced to 1390.0 ± 178.0 mg N/L.

The trial conducted at a HLR of 23.9 L/m^2/d was very stable from the start. The average conductivity of the effluent was 15.4 ± 1.8 mS/cm and the ammonia concentration was 1104.8 ± 197.0 mS/cm. Although the liquid loss at a HLR of 35.7 L/m^2/d was only slightly less than that at a HLR of 23.9 L/m^2/d, the performance was better. The ammonia concentration was 285.5 ± 9.1 mg N/L and the conductivity was 7.6 ± 0.4 mS/cm. The highest HLR tested, namely 62.3 L/m^2/d, produced an effluent with a conductivity and an ammonia concentration of 9.1 ± 0.4 mS/cm and 582.8 ± 61.2 mg N/L, respectively.

The concentration of the various nitrogen species, COD and pH values at the different HLRs are presented in Table 1. At all four HLRs the TKN concentration in the effluent was significantly reduced from approximately 4 000 mg N/L to 1499.2 ± 422.0 mg N/L; 1187.1 ± 180.9 mg N/L; 415.5 ± 133.4 mg N/L and 636.4 ± 43.8 mg N/L at 13.0, 23.9, 35.7 and 62.3 L/m^2/d, respectively, at stable state. The TKN concentration in the effluent was mainly in the form of ammonia, as urea is enzymatically hydrolysed to ammonia (Udert *et al*., 2003). The lowest ammonia concentration was obtained at a HLR of 35.7 L/m^2/d, namely 285.5 ± 9.1 mg N/L. Once the urea was oxidized to ammonia, the ammonia was probably removed by a combination of processes, namely volatilization, assimilation into the biomass and biological nitrification.

The hydrolysis of urea proceeds according to the equation below (Udert *et al*., 2003):

$$(NH_2)_2CO + 2H_2O \rightarrow NH_3 + NH_4^+ + CO_2 + HCO_3^-$$

The production of hydroxyl ions will increase the pH of the liquid. pH values up *ca* 9.0 was observed in samples taken from the middle of the reactor during the trails. The same observations were made by Maurer *et al*. (2006) when they showed that the pH of urine, which contains 84 % urea (Fowler, 2007) increased to 9.0 after storage. At pH values larger then 9.0, volatilization of ammonia will occur (MetCalf and Eddy, 2003).

$$\% NH_3 = \frac{100}{1 + [H^+]k_b / k_w}$$

The percentage ammonia that will volatilize can be calculated from the following equation (MetCalf and Eddy, 2003):

k_b = ionization constant for ammonia
k_w = ion product of water

$$\% NH_3 = \frac{100}{1 + [H^+]k_b / k_w}$$

The highest pH value observed in the middle of the reactor during the trials was 8.8.

Solving the above equation for pH value of 8.8 and 20 C, indicates that 20 % of the ammonia can be removed by air stripping. Thus, if the concentration of nitrogen in the feed was 4000 mg NL, the amount of ammonia that could be removed by volatilization would be 792.3 mg N/L.

Table 1.The performance of the reactor at the different hydraulic loading rates

	13.0 L/m^2/d	**23.9 L/m^2/d**	**35.7 L/m^2/d**	**62.3 L/m^2/d**
TKN, mg N/L	1499.2 (± 422.0)	1187.1 (± 180.9)	415.5 (± 133.4)	636.6 (± 43.8)
Ammonia, mg N/L	1390.0 (± 178.0)	1104.8 (± 197.0)	285.5 (± 9.1)	582.8 (± 61.2)
COD, mg COD/L	1152.5 (± 651.2)	1176.0 (± 230.9)	2300.3 (± 1757.2)	2957.5 (± 637.1)
Nitrite, mg N/L	15.4 (± 10.2)	17.3 (± 14.0)	14.7 (± 12.1)	15.3 (± 6.5)
Nitrate, mg N/L	1720.0 (± 273.8)	982.8 (± 224.5)	24.0 (± 23.2)	1.4 (± 1.1)
pH	6.1 (± 0.2)	7.5 (± 0.2)	7.8 (± 0.1)	7.7 (± 0.1)

Some of the nitrogen was also used for the assimilation of biomass. Microorganisms require nitrogen for growth. The nitrogen requirements can be calculated from stoichiometric reaction equations developed by using the concept of half-reactions (Grady and Lim, 1980). For heterotrophic, aerobic growth with glucose as carbon source and electron donor the three half reactions will be:

$$R_d = \frac{1}{24} C_6H_{12}O_6 + \frac{1}{4} H_2O \rightarrow \frac{1}{4} CO_2 + H^+ + e^- \quad (1)$$

$$R_c = \frac{1}{20} C_5H_7NO_2 + \frac{9}{20} H_2O \rightarrow \frac{1}{5} CO_2 + \frac{1}{20} HCO_3^- + \frac{1}{20} NH_4^+ + H^+ + e^- \quad (2)$$

$$R_a = \frac{1}{2} H_2O \rightarrow \frac{1}{4} O_2 + H^+ + e^- \quad (3)$$

Thus for glucose where Y_{NH} = 0.79 e^-. e biomass/ e^-. e substrate, the overall stoichiometric reaction equation for bacterial growth is:

$$R_0 = R_d - 0.79R_c - 0.21R_a \quad (4)$$

R_0 = the overall stoichiometric reaction
R_d = Reaction for electron donor
R_c = Reaction for bacterial cell synthesis
R_a = Reaction for electron acceptor
By applying this equation (4) the amount of ammonia required per e^-. e substrate (expressed as N) can be calculated as follows:

$$0.79 \times \frac{1}{20} \times 14 = 0.553 \text{ g N} \quad (5)$$

As 1 e^-. e substrate is equivalent to 8 g COD (Grady and Lim, 1980)

$$\frac{COD}{N} = \frac{8 \text{ g COD}}{0.553 \text{ g N}} = 14.47 \quad (6)$$

Since 16 g/L glucose (concentration of the glucose used in the synthetic medium) has a COD value of 17.1 g/L, the amount of N required for growth will be 1.1818 g/L. The amount of nitrogen required for synthesis of new biomass could therefore account for 29 % of the initial nitrogen loss in the system.

From the above explanations it is safe to assume that the *ca* 2000 mg N/L of nitrogen removed could be accounted for by the volatilization of ammonia and incorporation into the biomass. The rest of the ammonia was probably removed by biological nitrification. The nitrification capacity at the different HLRs was calculated according to the following formulae and the results are shown in Table 2:

$$\Delta N_{NH_3} = N_tQ - N_sQ - N_vQ - N_eQ_e \quad (7)$$

ΔN_{NH_3} = mass of ammonia nitrified per day, mg/d
N_t = TKN concentration in the feed, mg N/L
N_s = concentration of the nitrogen that is assimilated to form new biomass in mg N/L
N_v = concentration of the nitrogen that was volati ized in mg N/L.
N_e =TKN concentration in the effluent in mg N/L.
Q = dosing rate in L/d
Q_e = flow rate of the effluent in L/d.
The mass nitrites and nitrates that denitrified were calculated according to the following formulae:

$$\Delta N_{NO_X} = \Delta N_{NH_3} - Q_eN_{NO_X} \quad (8)$$

Where

ΔN_{NO_X} = mass of nitrites and nitrates denitrified per day in mg/d

N_{NO_X} = sum of the nitrite and nitrate concentrations in mg N/L

As the HLR increased, the mass of nitrogen applied increased and the biofilm in the reactor responded by increasing the rate of ammonia oxidation accordingly. The total ammonia nitrified at the different HLRs was 378.7 ± 61.2 mg N/d, 473.3.9 ± 98.1 mg N/d, 1097.0 ± 187.1 mg N/d and 1518.2 ± 185.5 mg N/d at HLRs of 13.0, 23.9, 35.7 and 62.3 L/m^2/d, respectively. As nitrifying organisms are strictly aerobic (MetCalf and Eddy, 2003) the results also implied that the oxygen supplied was enough to sustain nitrification. These results are in accordance to the model describes by Grady *et al.* (1999) which predicts behaviour of attached growth processes, in packed towers. The authors postulated that as the HLR increase, the resistant to mass transfer from the bulk of the liquid to the biofilm decrease and the rate of substrate removal increase. However, according to Grady and Lim (1980) at a very low flow rate the effectiveness increases faster than the mass application rate, which results in better substrate removal in the filter. However, there exists an optimum flow rate, where after the mass application rate will increase

faster than the effectiveness, which implies that a longer tower would be required to achieve the same degree of substrate removal. This could explain the better performance of the biological filter at the HLR of 13 L/m^2/d compared to that at 23.9 L/m^2/d.

Another consequence of increasing the HLR is that the biofilm is distributed deeper into the filter bed and that a longer reactor is required to achieve the same degree of treatment (Grady *et al*., 1999). A similar effect was observed at the reactor trial operated at a HLR of 62.3 L/m^2/d where the nitrification rate in the upper part of the filter was 1034.0 ± 276.0 mg N/d, in the same range as the trail operated at 35.7 L/m^2/d, namely 1082.7 ± 181.7 mg N/d. However, in the bottom part of the reactor the nitrification rate was 484.1 ± 178.5 mg N/d at the HLR of 62.3 L/m^2/d, where only 14.3 mg N/d was nitrified at 35.7 L/m^2/d. Another possible explanation for this observation is that the ammonia became limited, as all the urea was not hydrolysed in the upper part of the column. The results give in Table 3 show the difference between the TKN and ammonia concentration in the middle of the reactor at a HLR of 62.3 L/m^2/d. The difference was larger than the differences between the two parameters at lower HLRs (23.9 and 35.7 m^2/d). At 62.3 L/m^2/d, the difference was 516 mg N/L, while the differences were 201 mg N/L and 68.7 mg N/L at 23.9 and 35.7 L/m^2/d respectively. This indicated that all the urea was not oxidized to ammonia in the upper reaches of the reactor, but

Table 2.The mass nitrogen removal rate at the different hydraulic loading rates

HLR L/m^2/d	Nitrification rate ΔN_{NH_3}, mg/d			De-nitrification rate ΔN_{NO_X}, mg/d
	Upper part	Bottom part	Total NH_3-N oxidized	
13.0	ND	ND	378.7 (±65.9)	270.9 (± 97.0)
23.9	350.2 (± 258.7)	123.1 (± 85.1)	473.3 (± 98.1)	145.7 (± 116.1)
35.7	1082.7 (± 181.7)	14.3 (± 9.9)	1097.0 (± 187.1)	1084.2 (± 195.8)
62.3	1034.0 (± 276.0)	484.1 (± 178.5)	1518. 1 (± 185.5)	1502.9 (± 188.4)

ND – not determined because the volume of effluent produced was too little to sample and analysed both in the effluent and the middle

Table 3.The composition in the middle of the reactor at 23.9, 35.7 and 62.3 L/m^2/d

	23.9 L/m^2/d	35.7 L/m^2/d	62.3 L/m^2/d
TKN, mg N/L	1556.1 (± 334.4)	461.0 (± 171.1)	1290.4 (± 226.0)
Ammonia, mg N/L	1355.1 (± 263.9)	392.3 (± 171.9)	774.4 (±128.0)
COD, mg COD/L	2010.0 (± 780.2)	4025.4 (± 1586.0)	4717.5 (±1616.7)
Nitrite, mg N/L	21.7 (± 20.0)	0.1 (± 0.0)	5.3 (± 2.6)
Nitrate, mg N/L	334.5 (± 145.7)	8.8 (± 5.7)	2.1 (± 1.6)
pH	8.5 (± 0.2)	7.8 (± 0.5)	8.5 (± 0.5)

was only oxidized in the lower part of the reactor, as the difference between TKN and ammonia concentration in the effluent was only 54.6 mg N/L.

The higher concentrations of the TKN and ammonia in the effluent indicated that the length of the reactor indeed became limited at a HLR of 62.3 L/m^2/d (Table 1). The extent to which the biofilm will be pushed deeper into the filter bed will be determined by the organic loading rate. At the different HLRs of 13.0, 23.9, 35.7 and 62.3 L/m^2/d the organic loading rates were 0.15, 0.27, 0.41 and 0.71 kg BOD/m^3/d, respectively and the percentage nitrification achieved was *ca* 79 %, 55 %, 89 %, and 76 % respectively. MetCalf and Eddy (2003) reported that for rock media nitrification is only 50 % at an organic loading rate of 0.22 kgBOD_5/m^3/d and further recommends an organic loading rate of less than 0.08 kg BOD_5/m^3/d to achieve 90 % nitrification. These were significantly better than the typical organic loading rates that are required to achieve nitrification in standard rate trickling filters with rock media.

The de-nitrification rate increased as the HLR increased resulting in relatively low nitrite and nitrate concentrations in the effluent at HLRs of 35.7 and 62.3 L/m^2/d (Table 1). De-nitrification can occur in a biofilm even if aerobic conditions exist in the biological filter given that the biofilm is thick enough to maintain anaerobic conditions within (Biesterfeld *et al*., 2003). At the lower HLRs the de-nitrification rate was less complete, with nitrate concentrations of 1720.0 ± 273.8 mg N/L and 982.8 ± 224.5 mg N/L at 13.0 and 23.9 L/m^2/d respectively. This was probably due to the low flow rate and high oxygen in the biological filter possibly leading to an insufficient anaerobic layer in the bio-film. Another requirement for de-nitrification is that enough carbon is available. Grady and Lim (1980) observed that the COD:NO_3-N ratio in packed towers are higher than that required for suspended growth processes and reported that COD:NO_3-N ratios > 4.5 mg COD:mg NO_3-N is required to obtain more than 90 % removal of nitrogen (Grady and Lim, 1980). The COD:NO_3-N ratio in this investigation was 4.3, which was slightly lower but in the range of the above mentioned ratio and should therefore not be the major cause of weak de-nitrification rate at the lower HLRs of 13.0 and 23.9 L/m^2/d. A possible additional factor that could have an influence on the de-nitrification rate at 13.0 L/m^2/d is the low pH value of 6.1. Dinçer and Kargi (2000) reported an optimum pH value in the range 7 – 8 and Glass and Silverstein (1999) indicated that de-nitrification rate decreases as the pH decreases below 7. Evidence of toxicity to de-nitrifying bacteria is available at very high concentrations of nitrate, which are in the same range as the nitrate concentrations observed at the low HLR of 13.0 L/m^2/d (Table 1). Glass and Silverstein (1998) investigated the effect of pH on the de-nitrification rate in water that contains high nitrate concentrations. These authors found that with an initial nitrate concentration of 1350 mg N/L and the pH d" 7.0, de-nitrification was completely inhibited. These findings are similar to the observations made during this investigation.

The hydraulic loading rate also influences the rate at which microorganisms slough from the stones. At higher HLRs the shear stress increase, which results in an increase in the detachment rate of the microorganisms (Morgenroth and Wilderer, 2000). The gradual increase in the total COD in the effluent of the biological filter at the different HLRs confirmed this behaviour during this investigation. The total COD concentration in the effluent increased from 1152.5 ± 651.5 mg/L at 13.0 L/m^2/d to 2957.5 mg/L at 62.3 L/m^2/d.

CONCLUSION

The results from this investigation proofed that it will be possible to treat liquid with a composition similar to that of feacal sludge, namely high in nitrogen and organic matter, in a biological filter at the low HLRs of between 13.0 and 62.3 L/m^2/d. However, at 62.3 L/m^2/d the length of the filter, namely 1.5 m became limiting to the overall performance of the filter. The best results were obtained at a HLR of 35.7 L/m^2/d, which is equivalent to a pit latrine that serves a family of six persons.

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