Performance study of biofilter system for on-site greywater treatment at cottages and small households

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A R T I C L E   I N F O

Article history:
Received 21 December 2016
Received in revised form 25 April 2017
Accepted 27 April 2017
Available online 9 May 2017

Keywords:
Source-separation
Multi-stage approach
Effluent polishing
Soil infiltration

A B S T R A C T

The current contribution of microbial pathogens and nutrient discharge into the environment from inefficient on-site wastewater treatment systems has raised concern in many areas due to the pollution of the nearby water recipient. To overcome this challenge, a novel and more robust proven treatment systems are required. This paper aims to assess the performance of a source separating wastewater management system for the removal of organic matter, total P, total suspended particles and E. coli. The system is a multi-stage approach including – a separate collection of blackwater (BW) and greywater, followed by on-site greywater treatment system in a fixed-film biofilter and finally a soil infiltration system used as a polishing step before discharging into the environment. The separation and collection of BW resulted a notable reduction for chemical oxidation demand (COD), biochemical oxidation demand (BOD), total suspended solids (TSS), nitrogen (N) and phosphorus (P) accounting for 64%, 61%, 75%, 85 and 88%, respectively. The overall removal efficiency of the system for the above-mentioned parameters reached over 90% at the biofilter effluent and more than 95% at the bottom of the constructed infiltration column. For coliform bacteria and E. coli, the overall system reached a reduction of 4–5 log_{10} units of which the major reduction was observed in the infiltration columns. The effluent quality from this source-separating and multi-barrier biofilter treatment system complies with the Norwegian discharge limits. The assessment results reveal that this system can be used in drinking water source catchments with minimum environmental and health related risks.

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1. Introduction

The European water directive regulates treatment and discharge of wastewater based on the eutrophication sensitivity of the watersheds within a particular area. While greater efforts have been taken on the improvement of centralized wastewater systems in urban centers, little attention was so far given to rural areas, which are accounting for a notable fraction of the total wastewater production, especially in the northern countries. Based on statistics from 2015 (Berge and Chaudhary, 2015), 16% of the Norwegian population are not connected to central sewerage systems. Likewise, about one million residents in Finland, 20% of the total population, and over one million vacationists lives in houses that are not connected to the municipal sewer network (Lehtoranta et al., 2014). In addition to the 330,000 on-site systems for rural residents, more than 420,000 recreational houses are currently found in rural Norway (SSB, 2016) with the result that none-severed rural areas contribute with 24% of the wastewater production in the country. In 2014, the estimated nutrient discharges from rural households were approximately 350 tons for phosphorus and 3010 tons for nitrogen (Berge and Chaudhary, 2015) which is a significant fraction to affect the recipient water sources. Similar trend was also observed in Finland (Lehtoranta et al., 2014). In addition to eutrophication effects, on-site wastewater systems may pose a health risk to consumers of drinking water by spreading of pathogens to raw water catchments or by direct contamination of local wells. Hence rural wastewater management needs to be improved in order to sustain or improve the environmental quality and to protect human health.

Especially recreational houses that increased substantially both in number and size as well as in standard of sanitary facilities contributes with an increasing challenges in terms to rural wastewater management (Kaltenborn et al., 2009; Rye and Berg, 2011). Geological conditions on high mountain areas, where a majority of the cottages are located, often limit the applicability of soil infiltra-
tion, which is the dominant type of wastewater treatment (29%) in rural Norway (SSB, 2016). On particular places package treatment plants were installed as an alternative, but these systems were also shown to struggle with the highly varying loading conditions and limited maintenance, resulting into frequent malfunction periods with high discharge of pollutants (Schwermer and Wolfgang, 2016). Novel and more robust wastewater treatment systems need therefore to be developed to handle the increasing environmental pollution from Norwegian recreation homes.

Source separating sanitation was pointed out as a potential solution to meet the challenges in rural and recreational wastewater management (Jenssen et al., 2016). In this approach, only the so-called greywater originated from kitchen and washing facilities is treated locally while the notably higher polluted blackwater originated from toilets is collected and transported to a centralized treatment or recovery facility. Many different kinds of on-site wastewater and particularly greywater treatment systems have been developed and tested worldwide. Moreover, in countries like the USA and Australia, where regulations for the use of GW have been well established based on issues associated with public health and potential environmental impact, GW treatment for non-potable use is highly encouraged (Oron et al., 2014). However, the bulk of small-scale GW treatment systems currently proposed are either simple filtration systems providing minimal treatment, or are treatment systems, which are not designed to handle the differences in both flow and composition and are therefore not suitable (Gross et al., 2007). Others are complex treatment processes incorporating sedimentation tanks, bioreactors, ultra- and nano-membrane filtration, coagulants, and direct disinfection, which are more costly in terms of energy, operation and maintenance.

A multi-stage process consisting of several partially redundant treatment steps in series offer a relatively high treatment stability despite the variable loading rate. At present only little data are available on such source separating sanitary systems and these are mainly gathered from larger-scale pilot installations in urban regions (Todd et al., 2015). This study performed a comprehensive experiment to assess treatment efficiencies and effluent quality for each particular treatment step in a rural configuration of a source separating sanitary system. Post treatment system using column filtration to mimic soil infiltration trench was carried out to study the application in vulnerable areas and where discharge requirements are very stringent.

2. Methods

2.1. Source separating sanitary system

This study was done with greywater (GW) and blackwater (BW) supplied by a student dormitory with 48 inhabitants. The BW collected with vacuum toilets having a flushing volume of 1.21 and the greywater is collected and pumped separately into two separate stirred storage tanks in the laboratory. More details are given in Todd et al. (2015). For both wastewater fractions (GW, BW), grab samples were taken from the particular stirred storage tank. The concentration in a putative mixed raw sewage (Craw) was calculated considering an average BW fraction of 5.5% on the total wastewater volume as determined by Todd et al. (2015). This calculation was done with help of random variable algebra considering the measured concentrations ranges for greywater (C GW ) and blackwater (C BW ) as normal (COD, BOD, TSS, P) or log-normal (Coliform bacteria) distributed random variables, while a constant value was taken for the volume fraction of blackwater (f BW ) to avoid ratio distribution (Eq. (1))

\[ C_{\text{raw}} (\mu, \sigma) = C_{\text{GW}} (\mu, \sigma) \ast (1 - f_{\text{BW}}) + C_{\text{BW}} (\mu, \sigma) \ast f_{\text{BW}} \]  

(1)

2.2. Greywater treatment system

The study used a greywater treatment GWT system (Ecomotive A02, Ecomotive AS, Runde, Norway) designed for cottages and small households (Heistad, 2008). The GWT system encompasses a sequence of a primary settler, an unsaturated fixed-film biofilter and a secondary clarifier. For the fixed film biofilter lightweight clay aggregates having a diameter of 10–20 mm (LW) (Filtralite, Saint-Gobain Byggevarer AS, Alnabru, Norway) is used. The filter bed has a thickness of 500 mm. After primary settling, the greywater is distributed over the biofilter in intermittent pulses via full cone nozzles as described in Heistad (2008). The dosing pump was controlled by a level switch in the primary settler and a timer giving the pulse intervals. The filter is designed for a nominal load of 6501 d⁻¹, which results into a surface load of 282 mm d⁻¹. The biofilter is supposed to serve for a longer period, but to sustain its efficiency, a resting period of two or three weeks in a year is required.

The GWT system was loaded based the European test protocol for package treatment plants (NS-EN 12566-3:2005 + A2:2013) with a diurnal distribution of hydraulic load (Table 1). Feeding of the GWT was performed with a peristaltic pump (Bredel SPX, Whatson Marlos, Falmouth, UK) and hydraulic load was monitored with a flow meter (Optiflux2000, Krohne, Duisburg, Germany). Grab samples were taken from the effluent of the secondary clarifier. The power consumption was monitored with a power meter connected to the 230 V AC supply of the GWT.

The data from the GWT were collected from April 2013 to May 2016. In total, the system was in operation for 458 days in four continuous periods lasting from 28 to 223 days related to different experiments and performance tests that were conducted with the system. The latter included different sequences with overload, underload and simulated power breaks as outlined more in detail in Table 2.

2.3. Infiltration trench as a polishing step for the GWT effluent

To gather more data on the recommended post polishing in an infiltration trench, a column experiment was established (Reiakvam, 2016). During this period, the GWT was operated with nominal load. The experiment encompassed two parallel columns having a diameter of 600 mm. Each column represents a discharge point in an infiltration trench with a single-hole in the perforated disposal pipe that is placed on the top of the infiltration trench in the actual disposal system. The infiltration material used in this experiment consists of 150 mm drainage layer of 11–22 mm crushed granite stone at the bottom and sequentially overlaid by 150 mm of 0.2–1.0 mm fine sand dominated by silicon dioxide in the form of quartz and 150 mm of 2–4 mm LWA (Filtralite, Saint-Gobain Byggevarer AS, Alnabru, Norway). Single geotextile cover separated the layers and the trench is covered with 200 mm of till soil (sandy loam) at the top to mimic backfill (Fig. 1). Each of the infiltration columns was loaded with GWT effluent with peristaltic pumps at an actual flow rate of 2.51h⁻¹. The infiltration took place via a pipe having 6 mm inner diameter to the center of the column on the top of the LWA layer, giving a total filtration depth of 450 mm (Fig. 2). Loading of the

Table 1: Diurnal distribution of greywater into the GWTP.

<table>
<thead>
<tr>
<th>Time frame</th>
<th>Volume fraction (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0:00–0:00</td>
<td>no load</td>
</tr>
<tr>
<td>07:00–09:00</td>
<td>40</td>
</tr>
<tr>
<td>09:00–12:00</td>
<td>15</td>
</tr>
<tr>
<td>12:00–19:00</td>
<td>no load</td>
</tr>
<tr>
<td>19:00–21:00</td>
<td>30</td>
</tr>
<tr>
<td>21:00–0:00</td>
<td>15</td>
</tr>
</tbody>
</table>
Table 2

<table>
<thead>
<tr>
<th>Loading sequence</th>
<th>Hydraulic load (L d⁻¹)</th>
<th>number of periods</th>
<th>total length</th>
<th>number of samples</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nominal load (100%)</td>
<td>650</td>
<td>8</td>
<td>435 days</td>
<td>50</td>
</tr>
<tr>
<td>Overload (150%)</td>
<td>975</td>
<td>2</td>
<td>10 days</td>
<td>7</td>
</tr>
<tr>
<td>Underload (50%)</td>
<td>325</td>
<td>1</td>
<td>12 days</td>
<td>3</td>
</tr>
<tr>
<td>Power break</td>
<td>650</td>
<td>4</td>
<td>8 days</td>
<td>8</td>
</tr>
<tr>
<td>Loading breaks</td>
<td>no load</td>
<td>4</td>
<td>647 days</td>
<td>4*</td>
</tr>
</tbody>
</table>

* Samples were taken within the first 3 days after restarting load.

columns coincided with the operation periods of the biofilter in the GWTP. The latter were determined to have a total length 15 h d⁻¹ at the nominal load of 650 L d⁻¹. This implies that the filter has a long resting time, which allows sufficient time for drainage. This will prolong the lifetime of the infiltration system. Considering these figures, each of the columns reached a hydraulic load of 37.5 L d⁻¹ corresponding to 150 mm d⁻¹ over the whole column cross section area (Reikalvam, 2016).

Sampling from the infiltration columns was carried out in three 3-day sampling periods. The first period (P1) started 20 days, the second sampling period (P2) after 41 days and the third sampling period (P3) 118 days after the infiltration started. Analysis was done, based on grab samples taken from the center of the bottom plate via a drainage pipe having 15 mm diameter. For P1 and P2 the drainage pipe was open to the atmosphere representing deep unsaturated zone. For P3, the out let pipe is bent upwards with a water lock of 50 mm in order to simulate a ground water level at the bottom of the column, which keep the same pressure on the top of the ground water table in the actual field.

2.4. Lab analysis and mass load calculations

Grab samples were taken from inlet, GWTP outlet and final effluent under the normal loading and different stress events. BOD₅ was analysed with a manometry respirometric method (OxiTop, WTW, Weilheim, Germany). For COD, total phosphorus (P), total nitrogen (N) spectrophotometric test kits (Hach-Lange, Berlin, Germany) were used. Total suspended solids (TSS) were determined with 1.2 µm glass fiber filters (Whatman GF-C, GE Healthcare, Little Chalfont, UK). Filtrated COD was taken from the filtrate. E. coli was determined following the standard analytical methods (American Public Health Association (APHA), 2005) using Colilert 18 test kits (IDEXX Laboratories Inc, Maine, US).

The obtained reduction efficiency (Rₑff) for mass or cell numbers load within the different treatment steps are calculated based on the average values that have been determined for a putative combined raw sewage (Craw) and the corresponding sampling place X Cₓ for each of the parameters (Eq. (2)).

\[ R_{\text{eff}} = C_x \times (1 - f_{\text{BW}}) / C_{\text{raw}} \] (2)

3. Results and discussion

The performance of the source separating sanitary system was assessed by evaluating its removal efficiency for organic matter, TSS, total P and indicator microorganisms. The subsequent effect is the result of a combination of biological and mechanical processes. Fig. 2 shows the average concentration of COD, BOD, TSS, P₅₀ and TC and E. coli for the combined sewage, raw greywater, GWTP effluent and infiltration trench effluent and the mass load reduction at each level. The mass load reduction line indicates the removal efficiency for each treatment step. For those parameters, an overall treatment efficiency of more than 90% was reached at the effluent of the fixed-film biofilter and more than 95% at the bottom of the constructed infiltration columns (Fig. 2). For coliform organism, the overall system reached a reduction of 4−5 log of which the major reduction was observed in the infiltration columns (Fig. 2).

3.1. Separation and collection of blackwater

The separation and collection of BW resulted into notable reductions for COD, BOD, TSS, N and P accounting for 64%, 61%, 75%, 85 and 88%, respectively (Fig. 2), which again is within a comparable range to the figures reported by other studies (Meinzinger and Oldenburg, 2009; Vinneras et al., 2006). The reduction of TCB and E.coli on the other hand was surprisingly low, only accounting for 0.5 log and 0.1 log for TCB and E.coli, respectively (Fig. 2). This is due to the high concentration of TCB and E. coli in the raw greywater of 6.2 ± 0.4 and 6.7 ± 0.3 log 100 ml⁻¹, respectively. Other GW studies reported comparable high concentrations on TCB ranging 7.2−8.8 log 100 ml⁻¹, but lower numbers for E.coli ranging from 3.2−6.0 log 100 ml⁻¹ (Ottoson and Stenström, 2003). However, TCB and E.coli encompass both fecal and none-fecal organism (Ottoson, 2003; Ottoson and Stenström, 2003). A recent study showed that the mean concentration of coprostanol, a biomarker formed by the intestinal microflora, was 3.1 log lower in GW than in combined household wastewater. The fecal load estimated with the biomarker coprostanol in GW is 0.04 g person⁻¹ d⁻¹ which is 2.1−3.2 log lower compared to 5.4 g and 65 g person⁻¹ d⁻¹ when using the indicator bacteria E.coli and Fecal enterococci (Ottoson and Stenström, 2003). Hence, the indicator parameters TCB and E.coli used by this study likely overestimates the concentration of fecal pathogens in GW by 3 log (Ottoson, 2003). A majority of the detected TCB and E.coli in our GW are therefore likely not fecal origin but rather originated from the kitchen where high concentration up to 7.4 log were also reported elsewhere (Naturvårdsverket, ...)
1995) or re-growth of particular Coliform species in sewer pipes (Manville et al., 2001). The latter likely occurred also in our GW sewer system which has a long hydraulic retention time of 36 h or more (Todt et al., 2015), with an average temperature of 15 °C.

These findings are supported by another study (Oliinyk et al., 2015) on our GW using quantitative PCR (qPCR) analysis for human specific Bacteriods and Enterococci in the BW and GW. The results showed that the number of gene copies was 3.7 and 1.5 log lower in GW than in BW for Bacteriods and Enterococci, respectively. Hence, in terms of fecal pathogens having human origin, a separation of BW likely results into 1–4 log reduction. More research is needed to assess the distribution of different pathogenic organism in the wastewater fractions and related health risks more in detail. In addition, a potential regrowth and decay of different, pathogenic and none-pathogenic microorganism across a sewer or treatment system has to be addressed more in detail.

3.2. Onsite treatment of greywater in a fixed-film biofilter

The concentration of raw greywater was $137 \pm 38 \text{ mg O}_2/\text{L}$ for BOD, $267 \pm 71$ for COD $\text{ mg O}_2 L^{-1}$, $14 \pm 3 \text{ mg L}^{-1}$ for $N_{tot}$ and $1.2 \pm 0.3 \text{ mg L}^{-1}$ for $P_{tot}$. No notable difference to our earlier sampling period (Todt et al., 2015) could be identified for these parameters, indicating that the GW composition remains constant over time. As evaluated in our previous study (Todt et al., 2015), load and composition of our GW is comparable to other studies in Europe, except for $P$, which is slightly lower, likely due to the absence of dishwashing machines at the dormitories. Detergents for dishwashing became the major source of $P$ in GW after the introduction of phosphate free laundry agents.

Referring to raw GW, the GWTP reached a removal efficiency of 80%, 88%, 86%, 49% and 55% for BOD, COD, TSS, $N_{tot}$ and $P_{tot}$ respectively (data not shown). Together with a separation and collection of BW a removal efficiency of 93%; 95%; 96%; and 94% was obtained for BOD, total COD, TSS, and $P$, respectively. These figures are the average over the whole sample period, encompass-
ing also periods with overloading, under loading and power breaks (Table 2). In periods with nominal load, the removal efficiency of the GWTP reached more than 90% for BOD, COD and TSS and close to 60% for N and P. However, a reduced P-removal efficiency has to be expected on locations using phosphate in dishwashing agents, possibly in a range of 70–80% considering the range of P concentration reported by other GW studies (Palmquist and Haneus, 2005; Meinzinger and Oldenburg, 2009). In Norway, use of P containing detergents is prohibited and the raw greywater contain therefore low P, in our case less than 1.2 mg/L. In other areas where P containing detergents are in use the P concentration in the effluent may still be above the permissible limit. An additional polishing step for P-removal may therefore be needed on particular places. In such cases, reactive filter materials are increasingly used as post-filtration treatment to ensure removal of residual P. The filter materials for instance enriched lightweight aggregate LWA, Filtralite P® and Shell-sand (Adam et al., 2007), biochar and Filtralite (Eshetu et al., 2015) and Polonite® (Gustafsson et al., 2008) can be mentioned as efficient polishing filter. The power consumption of the system was determined to 0.34 kWh m⁻³ hydraulic load (data not shown). By taking into account an average GW production of 1081 dm⁻³ per person (Todt et al., 2015) this corresponds to 13 kWh y⁻¹ capita⁻¹, which is almost one order of magnitude lower than 93–217 kWh y⁻¹ capita⁻¹ that has been reported for onsite treatment of combined sewage (Straub, 2008).

The impact of overloading on the removal efficiency of the system was evaluated by comparing periods with 100% load to periods with 150% load (Fig. 3). Overloading with 150% of the nominal loading did not show significant difference on the removal of TSS and total P tot (p > 0.05) while, a significant lower (p < 0.001) removal efficiency was observed for organic matter. The high filter surface loading rate of 423 mm d⁻¹ resulted into a lower contact time with the biofilm, which again could have reduced the degradation of organic substrates as reflected by the lower removal efficiency for BOD as well as filtered COD. Regardless the reduced organic matter degradation, the filter still achieved an average removal efficiency of 70% for both BOD and COD during the 150% loading periods, which proves the high stability of fixed-film biofilter systems.

### 3.3. Post treatment

Unsaturated soil infiltration systems commonly used for the disposal of treated greywater as post treatment step. In this study, the columns represent discharge points in an infiltration trench with a single-hole in the perforated disposal pipe that is placed on the top of the infiltration trench in the actual disposal system. Short-circuiting is a common phenomenon in unsaturated infiltrations but the configuration of the filter media, the development of biofilm and the presence of geotextile layers allow a uniform distribution with time. This is demonstrated from the results of bacterial removal, which was increased from period 1 to period 2 and 3.

The results from the effluent polishing experiment further indicate a significant reduction of 85–90% for BOD, TSS and total P across the filter columns. As a result, a TSS of <2 mg/L, and total P <0.1 mg PL⁻¹, BOD <2 mg O₂L⁻¹ was achieved. Determination of total coliform bacteria and E. coli from the columns effluent in three periods showed significant reduction. Average TCB and E. coli log reduction during in the first period was 2.4 and 2.5 respectively. The reduction increased by more than 1 log after three weeks of operation. The average TCB and E. coli log reduction in the last two periods were 3.4 and 3.8, respectively (Fig. 4). The increase in log reduction of E. coli and TCB in the second and third period could be due to development of biofilm and an improved water distribution in the columns. This has been shown to increase the pathogen removal efficiency in filter systems (Heistad et al., 2009). Therefore, the polishing filtration step raised the total coliform and E. coli removal efficiency of the system up to 4.8 and 4.7 log reduction, respectively. This is in agreement with a previous study with biochar and Filtralite polishing filters (Eshetu et al., 2015). The issue of micropollutants is one of the most important concern to be considered for any on-site wastewater treatment. A separate study on heavy metal analysis in the system have shown a significant reduction of their concentration in the effluent (data not shown here). Moreover, separate blackwater collection allows concentration of organic micropollutants, particularly pharmaceuticals and hormones, in a significantly lower volume in the blackwater (Butkovskiy et al., 2017; De Graaff et al., 2011). Since most organic micropollutants are biodegradable, the significant reduction in the BOD of greywater may suggest a lower level of micropollutants in the effluent. However, due to the presence of some persistent
organic substances in greywater (Hernandez et al., 2007), low BOD may not necessarily indicate low concentration of micropollutants. Hence, further thorough investigation of the pharmaceutical and personal care products in the raw greywater and effluents are needed.

The biofilter system is efficient to remove most of the organic and particulate matter and waxy substances. The effluent that flows into the infiltration column, therefore, does not contain suspended solids that may cause clogging. No sign of clogging was observed throughout the experiment period. However, with time and development of biofilm on the surface of the geotextile layer an impediment could occur. The experiment is running further to assess the lifetime of the filtration column and associated occurrence of clogging through time.

The quality of effluent from the GWTP complies with the Norwegian discharge limit for discharge of treated household wastewater to sensitive recipients (Table 3). However, in terms of pathogen removal, only a 1–2 log reduction was observed. Although the present Norwegian regulations do not define discharge limits for indicator organisms (Table 3), the risks of faecal contamination from blackwater should not be overlooked (Stenström, 2013). Without post polishing, the effluent of the GWTP do not fulfill the requirements for present reuse standards (Table 3) and may be rather critical for sensitive recipient that are close to drinking water sources. For sensitive recipients as well as reuse applications, a multiple barrier approach including a post polishing in an infiltration trench is needed in order to minimise the related health risks (Table 3).

4. Conclusion

- It was observed that separation of BW from the rest of household wastewater streams resulted into a significant reductions for COD, BOD, TSS, N and P accounting for 64%, 61%, 75%, 85 and 88%, respectively. Separate treatment of GW in a biofilter reduced the concentration of organic matter and nutrients to discharge limit levels. Together with a separation and collection of BW a removal efficiency of 93%, 95%, 96%, and 94% was achieved for BOD, total COD, TSS, and P, respectively.
- Overloading the system up to 150% of the nominal loading did not affect the removal efficiency of the system for TSS and total Ptot (p > 0.05), However, lower removal efficiency was observed for organic matter.
- In terms of removal of indicator organisms, further treatment is a necessity. Infiltration trench or filtration columns as effluent polishing can significantly reduce the microbial concentration in the effluent. Overall system reached up to 5-log reduction of coliform bacteria, of which the major reduction was observed in the infiltration columns.
- For reuse applications or in drinking water areas a separate collection of blackwater in combination with a multiple barrier approach for the treatment of greywater including soil infiltration as final polishing is recommended in order to minimise the related health risks.

Acknowledgements

Ecomotive AS, Runde, Norway financially supported this study through its research and development program. The authors gratefully acknowledge Jostein Greve Gård (Ecomotive) for his unrestrained technical support, and Magnus Riakvam for his involvement in partial data collection.

References


