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PUBLISHED AS: RUANE, E.M., MURPHY, P.N.C., FRENCH, P., HEALY, M.G. 2014. COMPARISON OF A STRATIFIED AND A SINGLE-LAYER LABORATORY SAND FILTER TO TREAT DAIRY SOILED WATER FROM A FARM-SCALE WOODCHIP FILTER. WATER, AIR AND SOIL POLLUTION 225: 1915. DOI 10.1007/S11270-014-1915-Z

COMPARISON OF A STRATIFIED AND A SINGLE-LAYER LABORATORY SAND FILTER TO TREAT DAIRY SOILED WATER FROM A FARM-SCALE WOODCHIP FILTER

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ABSTRACT

Washing-down parlours and standing areas, following milking on dairy farms, produces dairy soiled water (DSW) that contains variable concentrations of nutrients. Aerobic woodchip filters can remove organic matter, nutrients and suspended solids (SS) in DSW, but the effluent exiting the filters may have to be further treated before it is suitable for re-use for washing yard areas. The performance of a single-layer sand filter (SF) and a

stratified SF, loaded at a 20 L m^{-2} d^{-1}, to polish effluent from a woodchip filter was investigated over 82 days. Average influent unfiltered chemical oxygen demand (COD_T), total nitrogen (TN), ammonium-N (NH_4-N), ortho-phosphorus (PO_4-P) and SS concentrations of 1991±296, 163±40, 42.3±16.9, 27.2±6.9 and 84±30 mg L^{-1} were recorded. The single-layer SF decreased the influent concentration of COD_T, TN, NH_4-N, PO_4-P and SS by 39, 36, 34, 58 and 52%, respectively. Influent concentrations of COD_T, TN_T, NH_4-N, PO_4-P and SS were decreased by 56, 57, 41, 74 and 62% in the stratified SF. The single-layer SF and the stratified SF were capable of reducing the influent concentration of total coliforms by 96 and 95%, respectively. Although a limited amount of biomass accumulated in the upper-most layers of both SFs, organic and particulate matter deposition within both filters affected rates of nitrification. Both types of SFs produced final water quality in excess of the standards for re-use in the washing of milking parlours.

Keywords: dairy soiled water; intermittent filtration; sand; treatment.

INTRODUCTION

Dairy soiled water (DSW) is water from concreted areas, hard stand areas, and holding areas for livestock that has become contaminated by livestock faeces or urine, chemical fertilisers and parlour washings (SI No. 610 of 2010), and contains high and variable levels of nutrients such as nitrogen (N) and phosphorus (P), as well as other constituents such as spilt milk and cleaning agents. It is legally defined in Ireland as having a five-day biochemical oxygen demand (BOD_5) of less than 2,500 mg L^{-1} and less than 1% dry matter (DM) content (S.I. No. 610 of 2010). Application of DSW to

54 the land has long been the most common method of disposal employed by farmers
55 (Martínez-Suller et al., 2010). However, when DSW is land applied at rates that
56 exceed the nutrient requirements of the pasture, it can create a number of problems,
57 including the threat of loss of P and N in runoff and, depending on the soil type,
58 subsurface leaching of P and N (Knudsen et al., 2006; Regan et al., 2010). Therefore,
59 treatment of DSW and re-use of filtered water may be considered as a management
60 option to divert DSW from land application.
61
62 Aerobic woodchip filters can decrease concentrations of organic matter, nutrients and
63 suspended solids (SS) in DSW (Ruane et al., 2011a), but their effluent may still
64 contain nitrate (NO_3) and coliforms (Ruane et al., 2011b), which means that it cannot
65 be re-used for wash-water for parlours (EC, 1980; IMQCS and Teagasc, 2004).
66 Therefore, before the use of aerobic woodchip filters can be recommended for on-
67 farm treatment of DSW, further treatment is needed. Intermittent sand filters (SFs)
68 may be used as a form of tertiary treatment for this wastewater. After passing through
69 a woodchip filter followed by further treatment in a SF, it is proposed that the effluent
70 could then be used to wash down a holding yard or milking parlour, depending on the
71 level of treatment achieved. This paper investigates the efficacy of two design
72 configurations of SFs – single layer and stratified.
73
74 Single-pass intermittent SFs (intermittently loaded SFs operated without recirculation
75 of a portion of the final effluent) have been employed as a tertiary treatment system to
76 polish several types of wastewaters (Leverenz et al., 2009; Healy et al., 2010). Their
77 ability to reduce the concentration of various water quality parameters, including N
78 and P, is well documented (Nakhla and Farooq, 2003; Healy et al., 2006). Stratified

SFs, containing sand media organised into layers of varying particle size, are also commonly used to treat wastewater (Nichols et al., 1997; Rodgers et al., 2005), due to the perception that the performance may be enhanced. Both types of SFs are also noted for their ability to reduce the concentration of pathogenic bacteria and micro-organisms (Bahgat et al., 1999; Stevik et al., 2004). However, operational problems still exist. These are primarily associated with clogging within the matrix of the sand, due to accumulation of particulate matter and micro-organisms on surfaces as biofilms, and the finite ability of the SF to remove P (Campos et al., 2002; Rodgers et al., 2005). As the presence of biofilm ultimately determines the longevity of a SF and is related to organic loading rate (OLR) and the size of the filter media (Rodgers et al., 2004), as well as hydraulic loading rate (HLR), filter dosing frequency and time of operation (Leverenz et al., 2009), it is important to monitor biofilm development in tandem with treatment efficacy when operating SFs. The deconstruction of SFs at the end of an experimental trial period allows a unique opportunity to assess the extent of biofilm development.

No study has compared the efficiency of stratified and single-layer SFs in treating DSW under the same organic and HLRs. In addition, although the build-up of biofilm on specific types of SFs has been well documented (sand - Rodgers et al., 2004; crushed glass and soil - Healy et al., 2011), no study has evaluated the difference between both filters vis-à-vis biofilm development. The build-up of biofilm, combined with an evaluation of their relative performance, may be used to determine the practicability of one design over another.

Physical and chemical mechanisms responsible for clogging, such as the field-saturated hydraulic conductivity (K_{fs}) and the organic matter content of the media, may be used to quantify the extent of biofilm build-up within a filter. In an intermittent SF, loaded with synthetic wastewater resembling DSW and operated for a period of 806 days, Rodgers et al. (2004) quantified the extent of biofilm build-up by dismantling the filter and measuring the K_{fs} in 0.02 m-depth increments from the surface of the filter. The reduction of K_{fs} appeared to extend deep into the sand filter and only returned to a K_{fs} of virgin sand (packed to the same density as in the sand filter) at a depth of 0.165 m below the filter surface. Rodgers et al. (2004) also used loss on ignition (LOI; BSI, 1990) to give an indication of biomass distribution within the SF and found similar trends to the K_{fs} measurements.

As the use of a SF is proposed as a final step in the overall treatment of DSW, the aim of this study was to compare the efficacy of two common types of SFs, stratified and single-layer, for their ability to polish effluent from farm-scale woodchip filters. The purpose of investigating two alternative SF designs was to propose a final tertiary treatment step as part of an overall on-farm system for the treatment of DSW, which would incorporate woodchip filters and – potentially – SFs.

MATERIALS AND METHODS

Two types of SFs were compared at laboratory-scale: (1) stratified SFs and (2) single-layer SFs (Figure 1). There were three replicates of each filter type. Filters were 0.3 m in diameter; stratified SFs were 1 m deep and single-layer SFs were 0.9 m deep. The stratified SF consisted of a 0.25 m-deep layer of distribution stone (6 - 10 mm diameter), underlain

by a 0.2 m-deep layer of coarse sand (effective size, D_{10}, 0.5 - 1.0 mm), a 0.075 m-deep layer of distribution stone, a 0.1 m-deep layer of medium sand (D_{10}, 0.4 - 0.8 mm), a 0.075 m-deep layer of distribution stone and a 0.2 m-deep layer of fine sand (D_{10}, 0.2 – 0.63 mm). The bottom layer of sand was underlain by a 0.1 m-deep layer of distribution stone. The sand depth and classification specifications were based on a study by Gross and Mitchell (1985), who obtained good removal efficiency with this design. The stratified SF consisted of a 0.1 m-deep distribution layer (6 -10 mm in size) underlain by a 0.7 m-deep layer of fine sand (D_{10}, 0.2 – 0.63 mm) and a 0.1 m-deep layer of washed stone (6-10 mm in size). The design specifications used for the single-layer SF are based on recommendations made in previous studies and by the USEPA, which state that a single-pass, single-layer SF should have a depth of 0.61 – 0.91 m and a D_{10} of 0.33 mm (USEPA, 1980; Ball and Denn, 1997; Loomis and Dow, 1999). Double-leaf CorriPipe™ (a locally sourced, commercially available pipe; JFC Ltd., Tuam, Co. Galway), 0.3 m in diameter (after Rodgers et al., 2005), was used to contain the sand. A steel mesh was attached to the base of the filters to hold the sand columns in place, and a plastic container was placed under the base of each SF to collect the treated effluent.

Each single-layer SF was instrumented with an access tube (type ATL1, Delta-T Devices Ltd., Cambridge, UK) to allow volumetric water content to be measured at various depths. A capacitance probe (type PR1/6d-02, Delta-T Devices Ltd., Cambridge, UK) was inserted into the access tube and readings taken using a voltmeter (type HR2 Delta-T Devices Ltd., Cambridge, UK). Readings were taken at 0.3, 0.4 and 0.6 m depth. These readings were then converted into volumetric water content ($m^3\ m^{-3}$) using the manufacturer's calibration curve.

153 Effluent from a farm-scale woodchip filter treating fresh DSW (Ruane et al., 2011b)
154 was collected every three days, stored in a tank, and loaded onto the SFs at a HLR of
155 20 L m^{-2} d^{-1}. The HLR was based on the average effluent COD concentration from the
156 farm-scale woodchip filters (Ruane et al., 2011b) and was chosen so as the OLR on
157 the SFs would not exceed approximately 25 g COD m^{-2} d^{-1} – the OLR above which
158 sand filters cease to be effective (Rodgers et al., 2005). Effluent from the woodchip
159 filters was pumped onto the surface of all six SFs every two hours using a peristaltic
160 pump (Masterflex L/S 16, Illinois, USA) delivering 118 mls per dose via a spiral
161 distribution manifold, positioned on the surface of each SF. The total study duration
162 was 82 days.
163
164 A 50-ml sample was collected from the influent for the SFs (discharge effluent from
165 the woodchip filter) and from the effluent at the base of each SF for analysis twice
166 weekly. After collection of the 50-ml sample the buckets beneath each sand filter
167 were emptied and cleaned. Therefore, the 50-ml sample is a subsample of the effluent
168 collected during a three or four-day period. Samples were frozen immediately and
169 tested within a period of 14 days. A closed reflux method was used to test for
170 unfiltered COD (COD_T) and filtered COD (COD_F). Total N and filtered TN (TN_F)
171 were measured using the persulfate method. Suspended solids were measured by
172 filtering a 10-ml sample through a filter paper (1.4 μm) and drying the solids captured
173 and the filter paper for 24 hours at 103 – 105 °C. Filtered samples were measured for
174 ammonium-N (NH_4-N), nitrite-N (NO_2-N), total oxidized N (TON) and ortho-
175 phosphorus (PO_4-P) using a Konelab 20 nutrient analyser (Fisher Scientific, Walthan,
176 Massachusetts). Nitrate-N was calculated by subtracting NO_2-N from TON. Dissolved
177 organic N (DON) was calculated by subtracting NO_2-N, NO_3-N and NH_4-N from

178 TN_F. Particulate N (PN) was calculated by subtracting TN_F from TN. Inorganic N
179 comprised NH_4-N, NO_2-N and NO_3-N. Total organic N (TON) was calculated by
180 subtracting inorganic N from TN. Samples of influent and effluent from all six SFs,
181 taken on days 62 and 75, were also analysed for total coliform (TC) content. All tests
182 were carried out in accordance with the standard methods (APHA-AWWA-WEF,
183 1995).
184
185 Removal of nutrients and other water quality parameters was calculated as the influent
186 concentration minus the effluent concentration, expressed as a percent of the influent
187 concentration. Descriptive statistics were used to characterise influent and effluent
188 concentrations and removal rates. Percent removal data were analysed using ANOVA
189 (SAS Institute, 2004) in a one-factorial design to test the effect of filter type on
190 performance.
191

192 *1.1. Phosphorus adsorption isotherms*

193 A P adsorption isotherm test was carried out on the three sands used in the SFs. A
194 sample of each sand (n=3) was first washed in distilled water diluted with 10%
195 hydrochloric acid to eliminate interference from the native P content of the sand.
196 Solutions containing four known concentrations of PO_4-P (21.5, 46.1, 61.4 and 92.1
197 mg PO_4-P L^{-1}) were used. Approximately 15 g of each type of sand - fine, medium
198 and coarse - was added to separate containers and mixed with 115 ml of each solution
199 concentration. Each mixture was then shaken for 24 hours using an end-over-end
200 mixer. The solids were separated from the mixture using a centrifuge and the
201 supernatant water was tested for PO_4-P. The data obtained was then modelled using a
202 Langmuir adsorption isotherm (McBride, 2000):

$$\frac{x}{m} = q_{MAX}\left(\frac{k_A Ce}{1 + k_A Ce}\right) \quad [1]$$

where x/m is the quantity of P adsorbed per gram of sand, Ce is the equilibrium P concentration in the pore water (g m^{-3}) at the end of the test, k_A (m^3 g^{-1}) is a measure of the affinity of the P for the sand, and q_{MAX} (g P g^{-1} sand) is the maximum amount of P that can be adsorbed onto the sand.

1.2. Assessing biomass build-up

After 82 days of operation, two columns from each set of SFs were destructively sampled so that the build up of biomass within each filter could be quantified using LOI (BS 1377-3:1990; BSI, 1990). For determination of the mass LOI, replicated sand samples (n=3) were taken from four layers below the surface distribution gravel: 0 - 0.03 m, 0.03 - 0.06 m, 0.06 - 0.09 m and 0.09 - 0.12 m. Samples were dried at 50 °C until a constant weight was achieved, then ground down until they passed through a 425 μm sieve. The prepared samples were placed in a cool muffle furnace and then heated to 450 °C for over 3 hours. The LOI from the dismantled SFs were then compared with virgin sand samples.

Physical changes in the single-layer SFs were also investigated by measuring the K_{fs} (m s^{-1}) (constant-head method; BSI, 1990) of samples taken at the same depth increments as the LOI samples. To measure K_{fs}, an open-ended circular container, 0.05 m in diameter, was used to extract a sand core of height, *l* (30 mm in height). Three replicate samples were taken from two single-layer SFs. Water was supplied to

the container and an overflow pipe maintained a constant head in the container, z (20 mm). Once constant flow rates were maintained, the flow rate, Q, was measured by calculating the time taken for a known volume of water to be collected. Taking the base of the open-ended cell as datum, the hydraulic gradient was defined as:

$$\frac{dH}{dZ} = 1 + \frac{z}{l} \quad [2]$$

The K_{fs} was then calculated using Darcy's Law (Craig, 1997):

$$\frac{Q}{A} = K_{fs} \frac{dH}{dZ} \quad [3]$$

where A is the cross-sectional area of the open-ended cell.

RESULTS AND DISCUSSION

2.1 Organic carbon (COD) and SS removal

The influent COD concentration in this study was higher than most other studies using SFs to treat wastewater that had undergone at least primary treatment (Table 1). Influent concentrations of COD_T were, on average, 1991±296 mg L^{-1}. The single-layer SF decreased the influent concentration by 39% to 1204±270 mg L^{-1} (Table 2). The performance of the stratified SF was significantly better ($P<0.001$), achieving an average decrease of 56%, resulting in an effluent concentration of 871±121 mg L^{-1}. A study by Rodgers et al. (2005) found that, after 230 days, a stratified SF treating synthetic DSW at the same HLR achieved a removal rate of 96%, decreasing an influent concentration of 1340±285 mg L^{-1} to 60±125 mg L^{-1}. However, the better performance achieved by Rodgers et al. (2005) may be a result of the enhanced straining effects of the medium due to the higher deposition of organic materials,

sediment and bacteria on the SF surface over a longer operational period (230 days versus 82 days in the present study).

An average influent COD_F concentration of 1073±221 mg L^{-1} was measured and removal rates of 38 and 55% were achieved by the single-layer and the stratified SF, respectively. The removal rates achieved by the stratified SF were significantly better ($P<0.001$) for both COD_T and COD_F. This would indicate that the stratified SFs were better at decreasing the soluble fraction of the influent as well as the fraction associated with influent SS. Therefore, both physical filtration and the oxidation of organic compounds may have contributed to the decrease in concentrations of COD_T and COD_F.

Influent SS concentrations were, on average, 84±30 mg L^{-1} (Table 2). The single-layer SF achieved an average decrease of 52%, giving an effluent concentration of 41±8 mg L^{-1}. Effluent concentrations of 32±6 mg L^{-1} were achieved by the stratified SF; a decrease of 62% on the influent concentration. Removal rates for the stratified SF were significantly better than the single-layer SF ($P<0.001$). Straining is the main mechanism of removing SS in SFs with interception, impaction and adhesion also contributing to the overall reduction of solids in the effluent (Prochaska and Zouboulis, 2003).

2.2 Nitrogen removal and conversion

Influent concentrations of TN ranged from 124 mg L^{-1} to 250 mg L^{-1} with a mean of 163 mg L^{-1}. The single-layer SF decreased the influent by, on average, 36% to 104±18 mg L^{-1} and the stratified SF by 57% to 70±21 mg L^{-1} ($P<0.001$) (Table 2). The

influent TN_F concentration of 113±25 mg L^{-1} was decreased by an average of 38% for the single-layer SF to 61±21 mg L^{-1} and by 41% for the stratified SF to 59±21 mg L^{-1} ($P>0.05$) (Table 2). Influent PN was, on average, 57±45 mg L^{-1}. The stratified SF outperformed the single-layer SF, decreasing the influent concentration by 80% compared to 25% for the single-layer SF (Table 2). Given its direct association with the SS concentration, it is most likely that the PN was reduced primarily by filtration and straining.

At 89±42 mg L^{-1}, TON (dissolved and particulate) accounted for, on average, 54% of the influent TN concentration. The single-layer SF decreased the influent concentration by an average of 39%, producing an effluent with a TON concentration of 54±15 mg L^{-1}. The stratified SF produced an effluent concentration of 22±19 mg L^{-1}; an average decrease of 76% (Table 2). The principal mechanisms of removing and transforming influent concentrations of TON in these aerobic SFs are likely to be filtration and mineralisation. As with SS, physical straining of particulate organic N as the influent water interacts with the porous media, is likely the main mechanism of removing SS in SFs.

Dissolved organic N in the influent was, on average, 39±25 mg L^{-1} over the duration of the study. The single-layer SF achieved a decrease of 65% to produce an effluent concentration of 13±16 mg L^{-1}. The stratified SF produced an effluent concentration of 15±18 mg L^{-1}, which represented an overall decrease of 61%. Mineralisation of DON was likely the main transformation mechanism for decreasing the influent concentration of DON. Oxygen is transported into the filter with the intermittently applied influent water and by diffusion in porous and unsaturated media (Schwager

301 and Boller, 1997). As DON decreases were likely due to mineralisation, it might have
302 been expected that NH_4–N concentrations might increase in the effluent. However, the
303 influent NH_4-N concentration decreased from 42±17 mg L^{-1} to 24±7 mg L^{-1} in the
304 single-layer SF and to 21±4 mg L^{-1} in the stratified SF, representing a decrease of 34
305 and 41%, respectively ($P>0.05$) (Table 2). Influent inorganic N concentration was, on
306 average, 74±22 mg L^{-1} over the duration of the study. Of this, NH_4-N accounted for
307 the largest fraction at 57%. As the SFs were aerobic, the principal mechanism for the
308 decrease in NH_4-N, despite the presumed production of additional NH_4-N via
309 mineralisation, was nitrification. Sorption of NH_4^+ was possible but, as the clay
310 content of the filter sands was negligible, this was very unlikely.
311
312 As NH_4-N decreases were likely due to nitrification, it might have been expected that
313 NO_2-N and NO_3-N concentrations might increase in the effluent. However, the single-
314 layer SF decreased NO_2-N and NO_3-N by 33 and 27% and the stratified SF by 50 and
315 4%, respectively ($P>0.05$). During the first 40 days, there was an increase in NO_3-N
316 and a decrease in NH_4-N that would indicate that nitrification was occurring. After 40
317 days, effluent concentrations of NH_4-N and NO_3-N reach an equilibrium suggesting
318 that biofilm may have established itself and NO_3-N may have been taken up into
319 biomass in biofilms growing on the filter medium (Bouwer et al., 2000). A more
320 likely scenario would be due to the retention of PN and TON within both sets of
321 filters: mineralisation of organic N would have affected NH_4-N removals, which, in
322 turn, may have affected nitrification rates. The high retention of PN and TON in the
323 stratified SF (80% and 76%, respectively) would appear to be related to the NO_3-N
324 removal rate of 4%. This was much lower than the single-layer SF (27%), which had
325 comparatively low removals of PN and TON (25 and 39%, respectively). This,

combined with the high OLR applied to the filters (approximately 40 g COD_T m^{-2} d^{-1}), may have suppressed nitrification activity. US EPA guidelines (US EPA, 1980) recommend a maximum COD_T loading rate of approximately 10 g COD_T m^{-2} d^{-1}. Operating within this OLR, other studies achieved almost complete nitrification (Nichols et al., 1997; Rodgers et al., 2005).

2.3 Phosphorus retention

The influent concentration of PO_4-P was, on average, 27.3 ± 6.9 mg L^{-1} (Table 2). The single-layer SF decreased the influent concentration by 58% to an average effluent concentration of 11.4±7.3 mg L^{-1}. The stratified SF achieved an average decrease of 74%, producing an average effluent concentration of 7.1±3.2 mg L^{-1} ($P<0.05$). For the single-layer SF, which consisted solely of medium sand (D_{10}, 0.4 – 0.8 mm), using the Langmuir isotherm, the theoretical maximum mass of P adsorbed per mass of sand was calculated to be 379 mg P kg^{-1} sand. The maximum mass of P adsorbed per mass of sand was calculated as 759.2 mg P kg^{-1} for the fine sand (D_{10}, 0.2 – 0.63 mm) and 1452.3 mg P kg^{-1} for the coarse sand (D_{10}, 0.5 – 1 mm). The high adsorbancy of the coarse sand may have been related to its mineralogy (not tested). These results are consistent with other studies showing a strong link between P sorption capacity of a filter medium and P removal. Healy et al. (2010) used three media (crushed glass – 0.5 to 1.1 mm in size; sand – D_{10}, 0.15 mm; and a shallow podzolized soil sieved to less than 5 mm) in 0.65 m-deep filters to treat low-strength domestic wastewater. The respective P adsorption capacities of the filter media (measured using a Langmuir adsorption isotherm) were: 10.3, 85 and 1043 mg P kg^{-1} and decreases of PO_4-P of 2.4, 4.3 and 100% were achieved in the glass, sand and soil filters.

2.4 Microbial analysis

Influent TC values of $8.5x10^6 \pm 7.1x10^5$ CFU 100 mls^{-1} were measured on days 62 and 75 of operation. Both the single-layer SFs and the stratified SFs recorded similar very high rates of removal of 96 and 95 %, reducing the concentration of TC to $3.2 \times 10^5 \pm 1.6 \times 10^5$ CFU 100 mls^{-1} and $4.2x10^5 \pm 13.2x10^5$ CFU 100 mls^{-1}, respectively. Physical filtration and adsorption, or adhesion, are believed to be the principal mechanisms for removing pathogenic bacteria from wastewater in a SF (Stevik et al., 2004). However, the water supply for cleaning operations in the milking parlour must be of potable standard (IMQCS and Teagasc, 2004) and for a treated wastewater to reach a standard of potable water, no coliforms may be present (EC, 1980). Therefore, some form of disinfection would be required to bring the treated effluent from the SFs used in this study to the standard of potable water. A UV treatment may be suitable for this purpose.

2.5 Pore-water profile and biomass build-up

Analysis of the volumetric water content of the single-layer SFs to a depth of 0.6 m indicated there was no significant change with depth or with time from the start of operation (Figure 3). Rodgers et al. (2005) examined a stratified SF loaded for 342 days with synthetic DSW at a HLR of 20 L m^{-2} d^{-1} and a SS loading rate of between 5.2 and 12 g SS m^{-2} d^{-1} and found that the volumetric water content increased to a maximum value of approximately 40% at the filter surface. The present study was only operational for 82 days at a SS loading rate of approximately 1.7 g m^{-2} d^{-1}. Therefore, the volumetric water contents indicated that significant amounts of biofilm, which could be detected by a capacitance probe, did not build up in the filter media.

375 Analysis of the LOI showed that most organic matter resided in the top 0.03 m in the
376 single-layer SF (Figure 4). The LOI in this layer, at 0.52 ± 0.02%, was more than five
377 times the LOI of the virgin coarse sand (0.09 ± 0.045%). For the stratified SF, LOI
378 was also greatest in the top layer at 0.38 ± 0.14% (Figure 4). Loss on ignition in the
379 0.03 - 0.06 m layer for the stratified SF was 0.19 ± 0.04%, compared with 0.3 ± 0.07
380 % in the single-layer filter. This could suggest that organic matter build-up extends
381 slightly deeper in the single-layer SF. Rodgers et al. (2004) measured maximum LOI
382 values of approximately 2.3% in the uppermost layer (to a depth of 0.02 m below the
383 filter surface) of a stratified SF loaded at OLRs ranging from 6.5 to 76 g COD_T m^{-2} d^{-1}
384 over an 806-day period.
385
386 Measurements of K_{fs}, carried out on the single-layer SFs (Figure 5) showed decreases
387 in K_{fs} in upper layers, indicating that some organic matter build-up occurred in the
388 upper layer of the filters. In the 0-0.03 m layer, K_{fs} was, on average, $1.34x10^{-4}$
389 $\pm 4.23x10^{-5}$ m s^{-1} and increased to $2.09x10^{-4} \pm 7.05x10^{-5}$ m s^{-1} in the 0.03 - 0.06 m
390 layers, and to $2.3x10^{-4} \pm 7.01x10^{-5}$ m s^{-1} in the 0.06-0.09 m layers. Analysis of the
391 single-layer SFs, on which both LOI and K_{fs} were measured, suggests a correlation
392 between measurements for K_{fs} and the build-up of organic matter to a depth of 0.06 m
393 in the single-layer SFs. The decreasing LOI with depth implies a decrease in the
394 build-up of organic matter on the filter medium with depth. The organic matter build-
395 up may be due to both entrapment of SS from the influent and also the growth of a
396 biofilm on the filter medium. The growth of biofilm in soil and sand filters receiving
397 DSW and septic tank effluent has been elucidated by Rodgers et al. (2004) and Nie et
398 al. (2011), amongst others. The growth of such a biofilm would be consistent with

399 uptake by microorganisms of NO_3 released through mineralisation and nitrification in
400 these aerobic filters.
401
402 **CONCLUSION**
403
404 Both types of SFs were capable of decreasing the concentrations of influent water
405 quality parameters. Both filters, however, produced an effluent with an NO_3-N
406 concentration greater than the MAC and total coliform limits for re-use in the washing
407 of milking parlours. Analysis of the distribution and build-up of organic matter within
408 the top layers of both SFs indicated that some organic matter build-up had occurred.
409 However, the build-up was only pronounced in the very top layers of both filters.
410 Over time, this build-up might be expected to increase. This might increase filter
411 effectiveness but may also lead, eventually, to problems of filter clogging.
412
413 On the basis of this study, stratified SFs are more effective in organic carbon, nutrient
414 and SS removal. The reason for this improved performance is likely the more tortuous
415 flow path and greater retention time that is likely in a SF compared to a single-layer
416 SF. However, issues such as difficulty in construction and the sourcing of sand of an
417 appropriate size hinder their implementation at farm-scale. Both filters ultimately do
418 not provide water of a quality suitable for discharge or for the cleaning of milking
419 parlours, and the final effluent may only be used for cleaning of the holding yard or
420 for landspreading.
421
422 **ACKNOWLEDGEMENTS**
423

The authors are grateful to Teagasc for the award of a Walsh Fellowship to the first author and for financial support provided by the Research Stimulus Fund (Department of Agriculture, Fisheries and Food). The authors appreciate the technical help of J. Kennelly and D. Minogue (Teagasc).

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565 **LIST OF FIGURES**

566

567 **Figure 1.** Schematic design of a stratified sand filter (a) and a single-layer sand filter
568 (b).
569

570 **Figure 2.** Comparison of the concentration of NH_4-N and NO_3-N in the effluent from
571 the single-layer SF (top) and stratified SF (bottom).

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573 **Figure 3.** Volumetric water content measurements in a single-layer SF. Each point is
574 an average of measurements taken on three different occasions (after 34, 47 and 82
575 days of operation)

576

577 **Figure 4** Mass loss on ignition in the upper layer of medium sand of a laboratory-
578 scale single-layer sand filter (top) and stratified sand filter (bottom)

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580 **Figure 5.** Field saturated hydraulic conductivity measurements for the single-layer
581 sand filters.

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