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Assessment of intermittently loaded woodchip and sand filters to treat dairy soiled water

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ABSTRACT

Land application of dairy soiled water (DSW) is expensive relative to its nutrient replacement value. The use of aerobic filters is an effective alternative method of treatment and potentially allows the final effluent to be reused on the farm. Knowledge gaps exist concerning the optimal design and operation of filters for the treatment of DSW. To address this, 18 laboratory-scale filters, with depths of either 0.6 m or 1 m, were intermittently loaded with DSW over periods of up to 220 days to evaluate the impacts of depth (0.6 m versus 1 m), organic loading rates (OLRs) (50 versus 155 g COD m^{-2} d^{-1}), and media type (woodchip versus sand) on organic, nutrient and suspended solids (SS) removals. The study found that media depth was important in contaminant removal in woodchip filters. Reductions of 78% chemical oxygen demand (COD), 95% SS, 85% total nitrogen (TN), 82% ammonium-nitrogen (NH_4–N), 50% total phosphorus (TP), and 54% dissolved reactive phosphorus (DRP) were measured in 1 m deep woodchip filters, which was greater than the reductions in 0.6 m deep woodchip filters. Woodchip filters also performed optimally when loaded at a high OLR (155 g COD m^{-2} d^{-1}), although the removal mechanism was primarily physical (i.e. straining) as opposed to biological. When operated at the same OLR and when of the same depth, the sand filters had better COD removals (96%) than woodchip (74%), but there was no significant difference between them in the removal of SS and NH_4–N. However, the likelihood of clogging makes sand filters less desirable than woodchip filters. Using the optimal designs of both configurations, the filter area required per cow for a woodchip filter is more than four times less than for a sand filter. Therefore, this study found that woodchip filters are more economically and environmentally effective in the treatment of DSW than sand filters, and optimal performance may be achieved using woodchip filters with a depth of at least 1 m, operated at an OLR of 155 g COD m^{-2} d^{-1}.

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1. Introduction

Dairy soiled water (DSW) (variously referred to as dairy effluent (Longhurst et al., 2000; McFarland et al., 2003), dairy dirty water (Cannon et al., 2000; Moir et al., 2005), or milk-house washwater (Joy et al., 2001)), is a variable strength dairy effluent (typical range 1000–10 000 mg 5-day biochemical oxygen demand (BOD_5) L^{-1}) comprising milking parlour and holding area washings generated in large but variable volumes (27–148 L cow^{-1} d^{-1}), and is characterised by low dry matter (DM) content (typically < 3–4%). Nutrient concentrations in DSW vary considerably, typically between 70 and 500 mg total nitrogen (TN) L^{-1} and 20 to >100 mg total phosphorus (TP) L^{-1} (Minogue et al., 2015). The volume and strength of DSW is seasonal and depends on farm management practices, including the efficiency of milking systems (Sweeten and Wolfe, 1994), size of herd, and amount of rainfall-generated runoff from uncovered hard standings (Minogue et al., 2015). Dairy soiled water is collected separately from dairy slurry and the main disposal route is directly to land via landspreading or irrigation without any prior treatment. Because of its high volume and often unpredictable composition, DSW is frequently perceived to be of little or no agronomic benefit and is often applied repeatedly to land adjacent to the milking parlour (Wang et al., 2004). Storage of DSW is required at locations where landspreading is restricted due to adverse weather conditions, soil type, soil conditions, ground slope, proximity to water sources, and volumetric spreading

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limitations. In Ireland, for example, there is a legal requirement to provide a DSW storage capacity of 10–15 days (S.I. No. 31 of 2014), which results in increased infrastructure and associated costs for the dairy farmer. These costs, combined with the low nutrient replacement value of the DSW, mean that treatment and reuse may be a better option for the farmer.

The environmental impacts of repeated spreading of DSW on lands are well documented (e.g. Fenton et al., 2011), and may result in oxygen depletion and asphyxiation of aquatic life in surface waters, as well as a risk of nutrient leaching to groundwater (Knudsen et al., 2006). Long-term DSW application to lands may also result in soil accumulation of phosphorus (P) and heavy metals and increase concentrations of microbial pathogens, odorants and oestrogens in the receiving environment (Wang et al., 2004; Hao et al., 2008). Hence, there is a real need for cost-effective, low energy, and low maintenance on-farm treatment processes that would result in a reduced risk of pollution following application to land. Some multi-stage biological treatment processes, such as combined sequencing batch reactors (SBRs) and constructed wetlands (CWs) (Moir et al., 2005), and aerated settling tanks followed by vertical flow CWs (Merlin and Gaillot, 2010), have been used with varying degrees of success; however, much of the organic and nutrient reductions in these studies have been reported to occur in the aeration rather than in the passive processes. Passive treatment systems such as sand filters (Rodgers et al., 2005; Healy et al., 2007) and woodchip filters (Ruane et al., 2011; McCarthy et al., 2015) have also been investigated and have reported consistently high levels of organic, nutrient and pathogenic removal. Woodchip, in particular, is a cheap, biodegradable material which has potential use as a soil improver (Cogliastro et al., 2001; Miller and Seastedt, 2009) and has previously shown to be effective in improving effluent quality and ammonia emissions when used in out-wintering pads (Dumont et al., 2012).

In order to realise the full potential of woodchip filters, it is necessary to determine the optimum media depths which will produce consistently high quality effluent when subjected to variable strength influent DSW loading. Filters are usually designed and operated with one hydraulic regime selected to deliver an optimum organic loading rate (OLR). However, as the concentration of DSW varies seasonally (Rodgers et al., 2005), woodchip filters may be subjected to OLRs far in excess of their design capacity. Therefore, it is necessary to examine the performance of filters under these extreme conditions. Limited information is available on the impact of woodchip filter depths and OLRs on the quality of treated DSW effluent. Additionally, no information is available on the comparative performances of woodchip and sand filters when treating on-farm DSW.

As there are still knowledge gaps concerning the optimal design and operation of woodchip filters for the treatment of DSW, including the appropriate OLR and filter depth for optimal performance, the objectives of this study were to examine the impacts of filter depth and OLR on their performance when loaded with DSW and to compare them to sand filters operated under the same experimental conditions. An overarching objective of the study was to contribute to an improved understanding of the factors which should be considered in the design, construction and management of passive woodchip filters to treat on-farm DSW. Once such factors are resolved, pilot-scale filters may be effectively operated on the farm.

2. Materials and methods

Eighteen filters, with internal diameters of 0.1 m and depths of either 0.6 m ($n = 3$ columns) or 1 m ($n = 15$ columns), were constructed using uPVC. All filters were open at the top and sealed at the base using uPVC end caps. The columns were placed on timber support frames and located in a temperature-controlled room at 10.6 ± 0.7 °C and relative humidity of $86.9 \pm 4.5\%$ (replicating the average temperature and humidity in Ireland). A 0.075 m layer of clean, crushed pea gravel, manually sieved to a particle size of 10–14 mm, was placed at the base of each column to prevent washout of the filter media. Each column was then filled with either woodchip (with a particle size of 10–20 mm) or sand (effective size, $d_{10} = 0.2$, uniformity coefficient, $UC = 1.4$) by placing the selected media in 0.050 m lightly tamped increments. Influent DSW was pumped intermittently (four times per day, seven days per week) onto the filters using peristaltic pumps controlled by electronic timers. Hydraulic loading rates were adjusted using the manual flow control on the pumps and influent was distributed evenly across the surface of the filter media using perforated uPVC flow distribution plates (Fig. 1). Continuously operated submersible mixers were placed in each DSW influent container (one container per column set) to prevent stratification. Treated effluent samples from each filter were collected in an effluent collection container and all influent DSW samples were taken simultaneously from the influent containers.

To clean any organic material from the media, 70 L of potable water was pumped onto each filter over a period of 5 days prior to their operation, before being intermittently loaded with DSW for a period of 56 days. On day 15 of operation, each filter was seeded with 500 mL of nitrifying activated sludge (mixed liquor suspended solids, $MLSS = 6290$ mg L^{-1}; sludge volume index, $SVI = 143$) collected from a local wastewater treatment plant. The period from day 0–56 was taken as the start-up period to reach steady state operation (defined by consistent chemical oxygen demand (COD), N and P effluent concentrations) for all filters and therefore day 56 was taken as the effective start day of the study (day 0).

This study compared three different operational setups to examine the impacts of (1) filter depth (2) OLR and (3) type of media (woodchip/sand) on filter performance. The filter configurations (Fig. 2) were (1) 0.6 and 1 m deep woodchip filters operating for 105 days with an average OLR of 120 g COD m^{-2} d^{-1} (2) 1 m deep woodchip filters operating for 105 days with average OLRs of 50 and 155 g COD m^{-2} d^{-1}, and (3) 1 m deep woodchip and sand filters operating for 220 days with an average OLR of 35 g COD m^{-2} d^{-1}. All configurations and treatments were constructed and operated at $n = 3$. The very high OLRs (120 and 155 g COD m^{-2} d^{-1}) were selected to assess the performance of filters under extreme loading events, which may arise if a filter is designed and hydraulically loaded assuming a low influent organic concentration.

Dairy soiled water was collected weekly for the duration of the experiments in 25 L capacity containers from a dedicated DSW collection tank at a 150 cow dairy farm in south west Ireland (51°37′35.8″N 8°46′06.6″W). A submersible pump was used to fill the containers, which were then transferred directly to a temperature-controlled room in the laboratory. The average physical and chemical characteristics of the influent DSW are shown in Table 1.

The woodchip used was a commercial tree species, Sitca spruce (*Picea sitchensis*). Logs were debarked and then chipped using an industrial wood chipping machine (Morbark post peeler) at an industrial facility in northwest Ireland. The woodchips were sieved to a 10–20 mm grading prior to placing in the filter columns. The sand used was sourced from a commercial quarry in Co. Galway, West of Ireland and was graded to a d_{10} of 0.2 mm and a UC of 1.4. The permeability of the saturated woodchip and sand (Table 2) was measured using the constant head permeability test in accordance with BS 1377-5 (BSI, 1990).

The ability of the woodchip and sand media to remove N

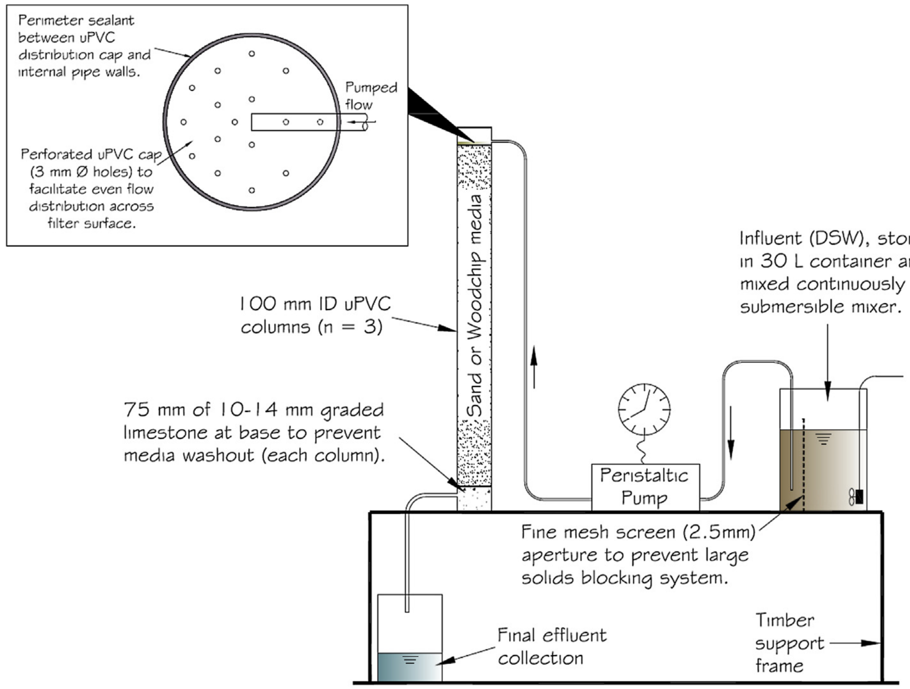

Fig. 1. Schematic diagram of typical laboratory filter setup. (Not to scale).

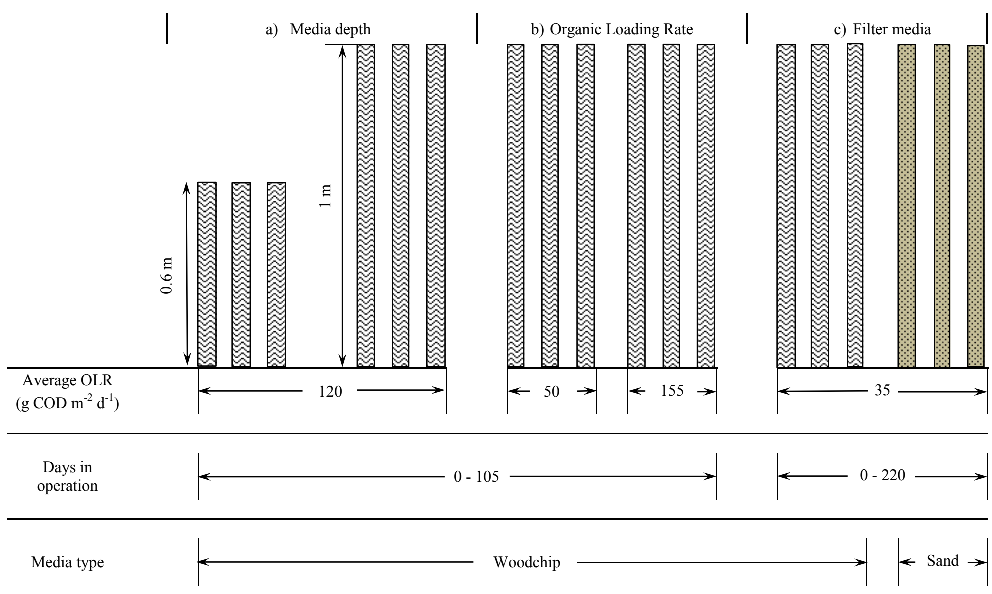

Fig. 2. Combinations of a) media depth, b) organic loading rates and c) filter media used in this study. The woodchip used was 10–20 mm Sitka spruce (picea sitchensis). The sand used had a d_{10} = 0.2 mm and a uniformity coefficient (UC) = 1.4.

Table 1
Physical and chemical properties of the influent DSW used in this study.

Parameter	Average ± standard deviation
COD (mg L^{-1})	2798 ± 1503
SS (mg L^{-1})	874 ± 614
TN (mg L^{-1})	81.5 ± 34.1
NH_4–N (mg L^{-1})	63.9 ± 32.3
TP (mg L^{-1})	29.8 ± 14.4
DRP (mg L^{-1})	24.3 ± 16.0
pH	7.22 ± 0.71
Dry matter (%)	0.2 ± 0.1

(measured as ammonium-N (NH_4–N)) and P (measured as dissolved reactive phosphorus (DRP)) from the DSW was investigated in a batch experiment by placing varying masses of the washed, graded media in flasks (n = 3) and adding 40 mL of raw DSW to each sample. All samples were shaken for 24 h at 250 excursions per minute (epm) on a reciprocating shaker and on removal, were allowed to settle for 1 h, filtered through a 0.45 μm filter, and tested colorimetrically using a nutrient analyser (Konelab 20, Thermo Clinical Laboratories Systems, Finland). The data were then modelled using a Langmuir isotherm to establish maximum adsorption capacities (Table 2).

Table 2
Properties of the filter media used in this study.

Media type	Grading	Hydraulic conductivity of saturated media (mm s^{-1})	Maximum adsorption capacity (g kg^{-1})	
			P	N
Woodchip	10–20 mm	1.25	–	3
Sand	d_{10} = 0.2 mm; UC = 1.4	0.03	136	–

Influent samples and effluent taken from each filter column were tested for pH using a pH probe (WTW, Germany) and for suspended solids (SS) using vacuum filtration on a well-mixed subsample through Whatman GF/C (pore size 1.2 μm) filter paper. Sub-samples were filtered through 0.45 μm filters and analysed colorimetrically for DRP, NH_4–N, total oxidised nitrogen (TON) and nitrite-N (NO_2–N) using a nutrient analyser (Konelab 20, Thermo Clinical Labsystems, Finland). Nitrate-N was calculated by subtracting NO_2–N from TON. Unfiltered samples were tested for TP and filtered (0.45 μm) samples for total dissolved phosphorus (TDP) using acid persulphate digestion. Particulate phosphorus (PP) was calculated by subtracting TDP from TP. Unfiltered samples were tested for TN using a BioTector Analyser (BioTector Analytical Systems Ltd., Cork, Ireland) and for COD (dichromate method). Influent DSW was tested for DM content by drying at 105 °C for 24 h. All water quality parameters were tested in accordance with standard methods (APHA, 2005).

2.1. Statistical analysis

The data were analysed using independent sample t-tests in SPSS (IBM SPSS Statistics 20 Core System) with column depth, OLRs and filter media as grouping variables. The data were checked for normality and, where necessary, were log transformed to satisfy the normal distributional assumptions required. Where normality was not achieved, the non-parametric Mann Whitney *U* test was used. Probability values of $p > 0.05$ were deemed not to be significant.

3. Results and discussion

3.1. Impact of media depth

Treated effluent concentrations from the 1 m deep woodchip filters were consistently lower than those from the 0.6 m deep filters for all measured parameters at an OLR of 120 g COD m^{-2} d^{-1} (Fig. 3). However, the concentrations for COD in the final effluent (1469 ± 587 mg L^{-1} for the 0.6 m filter and 587 ± 113 mg L^{-1} for the 1 m filter) were still far in excess of the limit value for discharge to urban waters (125 mg L^{-1}; SI No 254 of 2001). The 0.6 m deep filters reduced COD, SS, TP and DRP by 46%, 54%, 7% and 5%, respectively (based on average influent and effluent concentrations), but did not reduce TN and NH_4–N concentrations to below those of the influent. Reductions of 78% COD, 95% SS, 85% TN, 82% NH_4–N, 50% TP and 54% DRP were measured for the 1 m deep filters and were consistent with those of Ruane et al. (2011), who measured reductions of 66% COD and 57% TN for 1 m deep woodchip filter pads operating at an average OLR of 173 ± 43 g COD m^{-2} d^{-1} for a 1 year period. These findings indicate that filter depth is an important consideration in the design of woodchip filters, as the 0.6 m deep filters did not provide sufficient detention time to reduce COD and SS by more than approximately 50% at an average OLR of 120 g COD m^{-2} d^{-1}. These removals were increased by a factor of approximately 1.7 when the filter depth was increased to 1 m with consequent increase in detention time.

Ammonium-N was not nitrified in any of the woodchip filters and this was most likely as a result of the high average C:N ratio (30) of the influent DSW, which was far above the optimum C:N ratio of 3–6 for nitrification (Henze et al., 2001; Eding et al., 2006). This, combined with a high OLR (120 g COD m^{-2} d^{-1}), likely resulted in the formation of a dense, non-porous heterotrophic biofilm structure, reducing the available sites for the slow growing nitrifiers (Okabe et al., 1996; Wijeyekoon et al., 2004; Nogueira et al., 2002). A nitrogen mass balance between influent and effluent carried out on the 0.6 m deep filters showed that the mass of organic nitrogen (Norg) was reduced by 23% while the mass of NH_4–N increased by 8%, with no overall TN removal. For the 1 m deep filters, the mass of Norg was reduced by 37% with a corresponding reduction in NH_4–N of 82% and an overall decrease in TN of 85%, with NH_4–N as the dominant fraction in the final effluent. Therefore, while significant TN and NH_4–N removals were achieved in the 1 m deep filters (85% and 82%, respectively), the removal processes were by physical filtration of SS and associated N (Fig. 4(A)) rather than biological transformations. Much lower SS removals were measured in the 0.6 m deep filters (Fig. 3). The average pH of the treated effluent was 7.41 ± 0.26, indicating that alkalinity was not an inhibiting factor for nitrification. Ruane et al. (2011) reported an average concentration of 22.5 mg NO_3–N L^{-1} in treated effluent from 1 m deep woodchip filter pads loaded with DSW, which had an average influent concentration of 12.9 mg NO_3–N L^{-1} and C:N ratio of 16. In the current study, there was no NO_3–N in the influent and this may have influenced the biofilm formation and consequent opportunity for development of NH_4–N oxidizers (Okabe et al., 1996).

3.2. Impact of organic loading rates

There were no significant differences in the final effluent concentrations of NH_4–N (4.1 ± 4.1; 4.6 ± 4.2 mg L^{-1}) and SS (23 ± 16; 37 ± 22 mg L^{-1}) from the 1 m deep woodchip filters operated at OLRs of 50 and 155 g COD m^{-2} d^{-1}; however, the average effluent DRP concentration (3.8 ± 1.5 mg L^{-1}) from the 50 g COD m^{-2} d^{-1} filters was significantly lower ($p < 0.001$) than from the 155 g COD m^{-2} d^{-1} filters (10.2 ± 2.9 mg L^{-1}). As the woodchip had no ability to adsorb P (Table 2), physical removal was the main mechanism for P removal. Based on the influent and effluent loading rates, 2.5 mg PP d^{-1} (318 mg PP m^{-3} d^{-1}) was retained in the 155 g COD m^{-2} d^{-1} filters, whereas 0.4 mg PP d^{-1} (51 mg PP m^{-3} d^{-1}) was retained in the 50 g COD m^{-2} d^{-1} filters.

Removals (based on the average influent and effluent load and expressed in mg d^{-1}) in the range of 71%–97% were measured for COD, SS, TN and NH_4–N, and 54%–74% for TP and DRP, were measured in both sets of filters. Final effluent concentrations of SS, NH_4–N and DRP ranged from 23 to 37 mg L^{-1}, 4.1–4.6 mg L^{-1}, and 3.8–10.4 mg L^{-1}, respectively. However, the final effluent COD concentrations from both filters (766 ± 221 mg L^{-1} for the 50 g COD m^{-2} d^{-1} filters and 604 ± 112 mg L^{-1} for the 155 g COD m^{-2} d^{-1} filters) were well above the limit values for

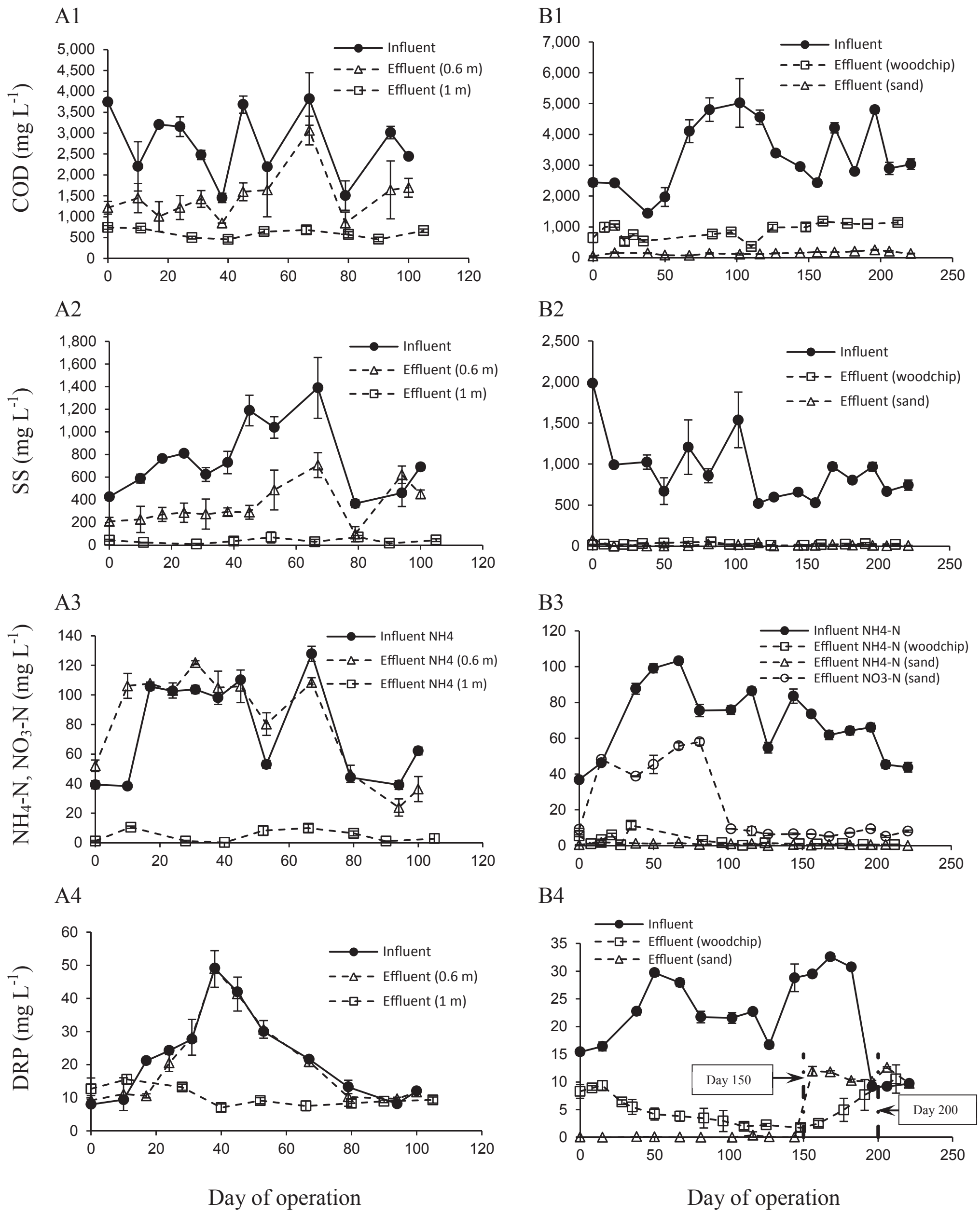

Fig. 3. Impact of media depth (A1 – A4) and media type (B1 – B4) on COD, SS, NH_4–N and DRP removals. An average organic loading rate of 120 g COD m^{-2} d^{-1} was applied to woodchip media (10–20 mm Sitka spruce) when comparing the impact of media depth (A1 – A4). An average organic loading rate of 35 g COD m^{-2} d^{-1} was applied to woodchip (10–20 mm Sitka spruce) and sand (d_{10} = 0.2 mm, UC = 1.4) media, both 1 m deep when comparing the impact of media type (B1 – B4). Error bars indicate standard deviations.

discharge to urban waters in Ireland (S.I. No 254 SI No. 31 of, 2001). Effluent mass loads for COD, SS, NH_4–N and DRP (Fig. 5) remained consistent over the duration of the study period, highlighting the capacity of the filters to effectively and consistently treat variable strength and variably loaded influent DSW.

Negligible NO_3–N concentrations were measured in the effluent, underlining the reliance on physical filtration for NH_4–N removal as illustrated by the close correlations between SS and NH_4–N mass removals for both loading rates (Fig 4(B)).

3.3. Impact of filter media

There were no significant differences between the treated effluent from 1 m deep woodchip and 1 m deep sand media (average OLR = 35 g COD m^{-2} d^{-1}) for SS (23 ± 13 and 16 ± 20 mg L^{-1}) and NH_4–N (2.9 ± 3.4 and 0.8 ± 0.5 mg L^{-1}); however, the sand outperformed the woodchip in COD removal (a final effluent of 146 ± 52 mg L^{-1} versus 873 ± 242 mg L^{-1}) and DRP removal up to day 150 (a final effluent of 0.1 ± 0.1 mg DRP L^{-1} versus 4.9 ± 2.7 mg DRP L^{-1}). The enhanced COD removals in the sand filters were reflective of their higher hydraulic retention time when compared to the woodchip filters (the hydraulic conductivity of the sand was >40 times lower than that of the woodchip (Table 2)). The enhanced DRP removals in the sand filters were as a result of their higher P adsorption capacity (136 g DRP kg^{-1}) compared with the woodchip, which had no affinity for P, and DRP reductions in the woodchip filters were associated with SS removals (Fig. 4(C)). After 150 days of operation, DRP breakthrough

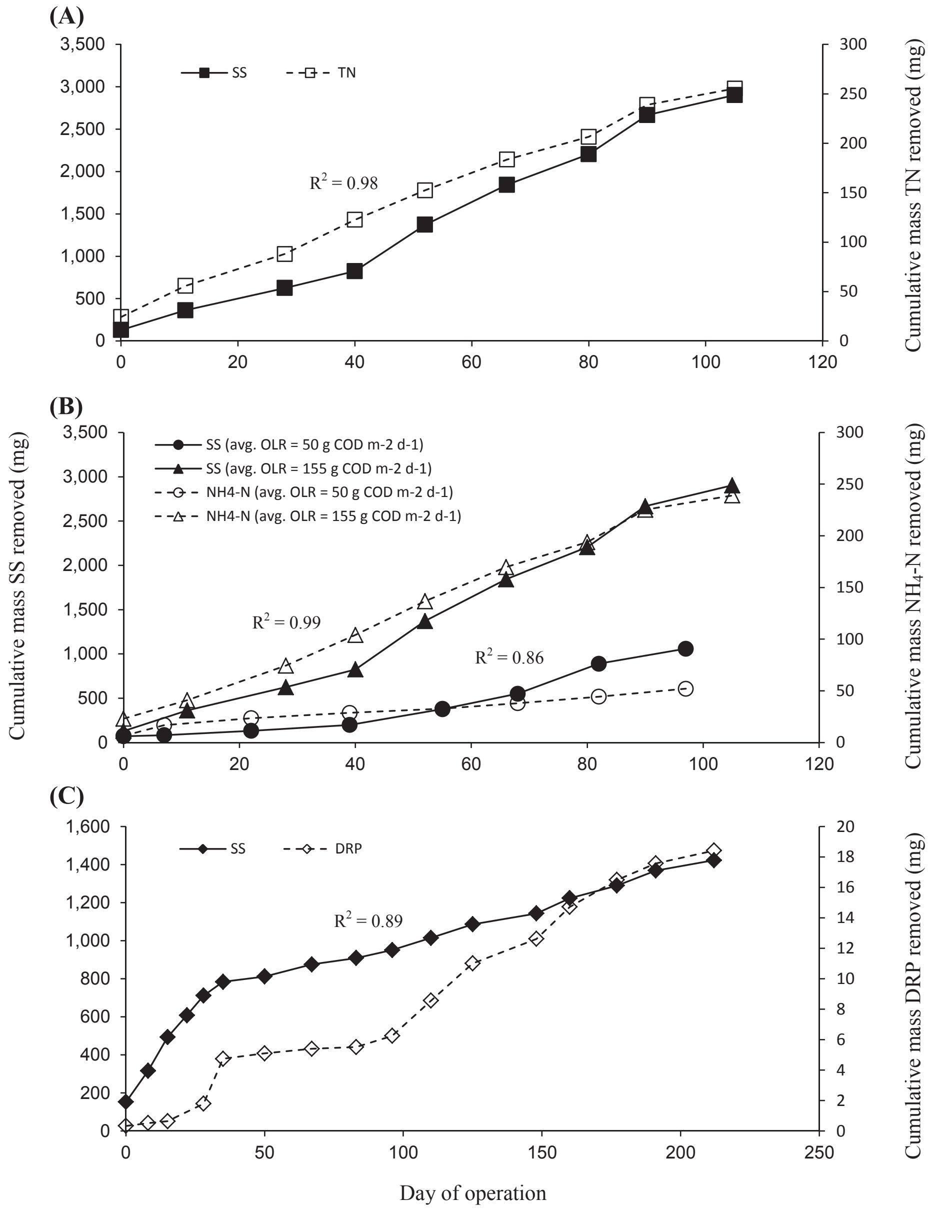

Fig. 4. Correlations between cumulative mass removals of suspended solids (SS) for1 m deep × 0.1 m Ø woodchip filters (n = 3, each set) and (A) TN loaded at 120 g COD m^{-2} d^{-1} (B) NH_4–N loaded at 50 and 155 g COD m^{-2} d^{-1} respectively and (C) DRP loaded at 35 g COD m^{-2} d^{-1}. Correlation coefficients, (R^2) indicated.

occurred quite quickly in the sand filters and at a slower rate in the woodchip filters (Fig. 3). From day 200 to the end of the study, neither the sand nor the woodchip filters removed any DRP from the influent DSW (Fig. 3). The average mass of P retained up to day 150 was 1.61 ± 1.30 and 3.89 ± 0.76 mg TP d^{-1}, 0.61 ± 0.31 and 0.96 ± 0.32 mg PP d^{-1} and 1.33 ± 0.84 and 2.58 ± 0.60 mg DRP d^{-1} for woodchip and sand filters, respectively, indicating that the sand was more effective at removing PP and also had a greater affinity for adsorption of DRP (Table 2). The mass removal rates also indicate that sand had more consistent P removal than woodchip up to day 150.

During the first 85 days of operation, nitrification occurred in the sand filters and the NO_3–N concentration rose from 0.1 ± 0.1 mg L^{-1} in the influent to 43 ± 18 mg L^{-1} in the effluent. However, the effluent NO_3–N subsequently reduced considerably, and attained an average concentration of 7.2 ± 1.6 mg L^{-1} by the end of the study (Fig. 3). The reasons for the suppressed levels of NO_3–N were possibly due to the preferential formation of heterotrophic-dominated biofilm layers limiting dissolved oxygen (DO) to the nitrifiers (Nogueira et al., 2002) as a consequence of the high influent C:N ratios in the influent wastewater (average of 38). Negligible NO_3–N concentrations were measured in the treated effluent from the woodchip filters and were always below 0.21 ± 0.19 mg L^{-1}. This indicates that even at the low OLRs used in this study, which are at the upper limit at which nitrification normally occurs in sand filters treating a similar type of wastewater (around 30 g COD m^{-2} d^{-1}; Rodgers et al., 2005), woodchip filters are unable to nitrify DSW.

3.4. Assessment of optimum filter media, configuration and operation

When assessing the suitability of the filters to treat on-farm DSW, key operating criteria must be taken into account, together

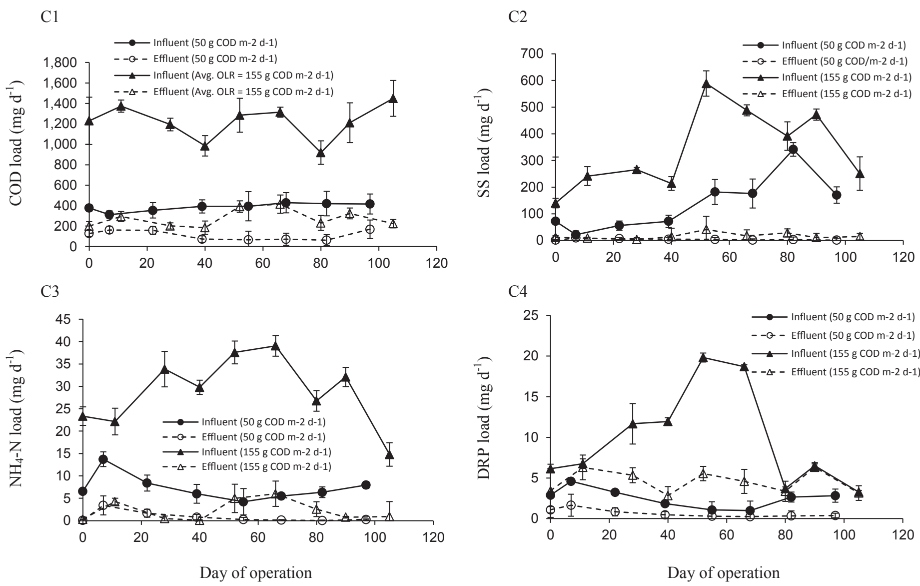

Fig. 5. Impact of organic loading rates on COD, SS, NH_4–N and DRP mass removals. The filter material used was 10–20 mm Sitka spruce woodchip, 1 m deep. Error bars indicate standard deviations.

with the main objective of reducing organic and nutrient concentrations to levels which would not adversely impact the environment if landspread. These operating criteria include items such as cost and availability of the media, robustness and longevity of performance (i.e. how well can media deal with daily and seasonal variations in flow and strength and for how long), biodegradability, and disposal of spent media.

The results of this study show that woodchip filters should have a minimum depth of 1 m to achieve required removals and can reduce the measured water quality parameters at OLRs up to at least 155 g COD m^{-2} d^{-1}. However, based on the N mass balances and effluent concentrations of NO_3–N measured in this study, the removal mechanisms in woodchip filters are primarily physical (straining) and not biological (nitrification did not occur). The suppression of biological activity may have been a function of the OLRs employed in this study, where the lowest OLR studied (35 g COD m^{-2} d^{-1}) was still at the upper limit at which nitrification normally occurs in filters (Rodgers et al., 2005).

Biological N transformations are a sustainable long-term process to reduce effluent N when compared to removal by physical straining alone. While nitrification was not observed to occur in the woodchip filters in the current study, other studies (e.g. Carney et al., 2011) have reported its occurrence for piggery wastewaters at OLRs in the range 14–128 g COD m^{-2} d^{-1}. Nitrification of DSW in sand filters has been reported in many studies (e.g. Rodgers et al., 2005; Healy et al., 2011) at OLRs in the range 20–40 g COD m^{-2} d^{-1}. Given that the composition of raw DSW normally contains very low, if any, NO_2 or NO_3 concentrations (Minogue et al., 2015), long start-up times are likely to be required to establish an active population of NH_4 oxidizers in any filter medium (Okabe et al., 1996; Lekang and Kleppe, 2000).

Surface clogging of the filter media is an operational issue that must be considered for on-farm use and while neither the sand nor the woodchip media in this study experienced surface clogging, Healy et al. (2007) reported clogging of sand filters after 42 days at an OLR of 43 g COD m^{-2} d^{-1}. In contrast, we are not aware of any reported issues with surface clogging of woodchip media, and it has been estimated that a woodchip filter may be operational for 2–3 years before surface ponding occurs (Ruane et al., 2011).

The decision to use woodchip or sand filter media is ultimately taken by synthesizing environmental benefits versus capital and operating costs. Operating costs are similar for both woodchip and sand filters (the modes of operation are identical for both), while capital costs are differentiated only by the cost of the media (filter setup for woodchip and sand are similar), which may also not differ significantly and will be location specific. Cost comparisons therefore can be made by comparing the required footprint of woodchip and sand media, both at a depth of 1 m – the minimum acceptable filter depth identified in this study. Based on the optimal OLRs identified in this study (an OLR of 155 g COD m^{-2} d^{-1} for woodchip filters, which treated the wastewater through physical processes, if not necessarily biological processes, and an OLR of 35 g COD m^{-2} d^{-1} for sand filters, which only temporarily caused the occurrence of nitrification, but clearly was at the upper OLR limit at which such filters may be operated), a filter surface area of 0.48 m^2 cow^{-1} for woodchip versus 2.1 m^2 cow^{-1} for sand would be required (Table 3). The larger area required for the sand filter combined with their lack of robustness to deal with shock loads (Healy et al., 2007) and the potential for surface clogging (Rodgers et al., 2005), indicate that woodchip filters are a better on-farm treatment option.

The optimal filter configuration identified in the current study produced a final effluent that was in excess of permissible discharge standards. For the water to be discharged to surface waters, some form of primary and tertiary treatment may be required. Primary treatment may consist of a simple sedimentation

Table 3
Comparative filter areas (per cow) of a full scale filter for optimal organic loading rates investigated in this study of 155 g COD m^{-2} d^{-1} for woodchip and 35 g COD m^{-2} d^{-1} for sand.

Q[a]	COD load[b]	Filter area per cow (m^b)	
(L d^{-1} cow^{-1})	(g COD d^{-1})	Woodchip[c]	Sand[d]
27	73.7	0.48	2.1

[a] Minogue et al., 2015.
[b] Assuming an annual average COD concentration of 2750 mg L^{-1}.
[c] Using an OLR of 155 g COD m^{-2} d^{-1}.
[d] Using an OLR of 35 g COD m^{-2} d^{-1}.

tank upstream of the woodchip filters to reduce SS in the influent DSW, and tertiary treatment might comprise the addition of downstream polishing filters using, for example, zeolite for enhanced N removal and flue gas desulphurization (FGD) gypsum for enhanced P removal. However, this would be costly for the farmer and, moreover, would mean that a discharge license may be required. Additionally, the technical and economic feasibility of using such tertiary media to act as polishing filters for DSW treatment would need to be established. Based on the results of the current study, a 1 m deep woodchip filter, with an OLR of 155 g COD m^{-2} d^{-1}, may retain up 600 mg SS d^{-1} (Fig. 5) and may reduce over 90% of the SS. Therefore, the liquid portion of the wastewater may be used in irrigation, which requires no discharge license or transport costs, and is safer (Augustenborg et al., 2008a); and, once exhausted, the spent timber residue may be incorporated into the soil (Augustenborg et al., 2008b).

4. Conclusions

On the basis of this study, woodchip filters are more effective in the treatment of DSW than sand filters. In this study, optimal performance in terms of mass of contaminants removed per day was achieved using a 1 m deep woodchip filter operated at an OLR of 155 g COD m^{-2} d^{-1}. Filtration was the dominant mechanism for N removal in the woodchip filters. The final effluent was above the concentrations at which it may be legally discharged to receiving waters. Therefore, management option employed to re-use the final effluent may be to use the liquid portion of the effluent in irrigation and, in time, to incorporate the spent timber residue into the soil.

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