



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

Title	Assessment of intermittently loaded woodchip and sand filters to treat dairy soiled water
Author(s)	Murnane, J. G.; Brennan, R. B.; Healy, Mark G.
Publication Date	2016-07-31
Publisher	IWA Publishing and Elsevier
Repository DOI	10.1016/j.watres.2016.07.067

1 *Published as: Murnane, J.G., Brennan, R.B., Healy, M.G., Fenton, O. 2016.*
2 *Assessment of intermittently loaded woodchip and sand filters to treat dairy*
3 *soiled water. Water Research 103: 408 – 415.*
4

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

7 J.G. Murnane^{1,2}, R.B. Brennan¹, M.G. Healy^{1*} O. Fenton³

8 ¹Civil Engineering, National University of Ireland, Galway, Co. Galway, Rep. of
9 Ireland.

10 ²Civil Engineering and Materials Science, University of Limerick, Co. Limerick, Rep.
11 of Ireland.

12 ³Teagasc, Johnstown Castle, Environment Research Centre, Co Wexford, Rep. of
13 Ireland

14
15 *Corresponding author. Tel: +353 91 495364; fax: +353 91 494507. E-mail address:
16 mark.healy@nuigalway.ie

17
18 **Abstract**

19 Land application of dairy soiled water (DSW) is expensive relative to its nutrient
20 replacement value. The use of aerobic filters is an effective alternative method of
21 treatment and potentially allows the final effluent to be reused on the farm.
22 Knowledge gaps exist concerning the optimal design and operation of filters for the
23 treatment of DSW. To address this, 18 laboratory-scale filters, with depths of either
24 0.6 m or 1 m, were intermittently loaded with DSW over periods of up to 220 days to
25 evaluate the impacts of depth (0.6 m versus 1 m), organic loading rates (OLRs) (50
26 versus 155 g COD m⁻²d⁻¹), and media type (woodchip versus sand) on organic,
27 nutrient and suspended solids (SS) removals. The study found that media depth was

28 important in contaminant removal in woodchip filters. Reductions of 78% chemical
29 oxygen demand (COD), 95% SS, 85% total nitrogen (TN), 82% ammonium-nitrogen
30 ($\text{NH}_4\text{-N}$), 50% total phosphorus (TP), and 54% dissolved reactive phosphorus (DRP)
31 were measured in 1 m deep woodchip filters, which was greater than the reductions in
32 0.6 m deep woodchip filters. Woodchip filters also performed optimally when loaded
33 at a high OLR ($155 \text{ g COD m}^{-2} \text{ d}^{-1}$), although the removal mechanism was primarily
34 physical (i.e. straining) as opposed to biological. When operated at the same OLR and
35 when of the same depth, the sand filters had better COD removals (96%) than
36 woodchip (74%), but there was no significant difference between them in the removal
37 of SS and $\text{NH}_4\text{-N}$. However, the likelihood of clogging makes sand filters less
38 desirable than woodchip filters. Using the optimal designs of both configurations, the
39 filter area required per cow for a woodchip filter is more than four times less than for
40 a sand filter. Therefore, this study found that woodchip filters are more economically
41 and environmentally effective in the treatment of DSW than sand filters, and optimal
42 performance may be achieved using woodchip filters with a depth of at least 1 m,
43 operated at an OLR of $155 \text{ g COD m}^{-2} \text{ d}^{-1}$.

44

45 *Keywords:* Passive filtration; woodchip; sand; dairy soiled water; organic loading rate.

46

47 **1. Introduction**

48 Dairy soiled water (DSW) (variously referred to as dairy effluent (Longhurst et al.,
49 2000; McFarland et al., 2003), dairy dirty water (Cannon et al., 2000; Moir et al.,
50 2005), or milk-house washwater (Joy et al., 2001)), is a variable strength dairy
51 effluent (typical range 1000 – 10000 mg 5-day biochemical oxygen demand (BOD_5)
52 L^{-1}) comprising milking parlour and holding area washings generated in large but

53 variable volumes (27 – 148 L cow⁻¹ d⁻¹), and is characterised by low dry matter (DM)
54 content (typically < 3 - 4%). Nutrient concentrations in DSW vary considerably,
55 typically between 70 to 500 mg total nitrogen (TN) L⁻¹ and 20 to >100 mg total
56 phosphorus (TP) L⁻¹ (Minogue et al., 2015). The volume and strength of DSW is
57 seasonal and depends on farm management practices, including the efficiency of
58 milking systems (Sweeten and Wolfe, 1994), size of herd, and amount of rainfall-
59 generated runoff from uncovered hard standings (Minogue et al., 2015). Dairy soiled
60 water is collected separately from dairy slurry and the main disposal route is directly
61 to land via landspreading or irrigation without any prior treatment. Because of its high
62 volume and often unpredictable composition, DSW is frequently perceived to be of
63 little or no agronomic benefit and is often applied repeatedly to land adjacent to the
64 milking parlour (Wang et al., 2004). Storage of DSW is required at locations where
65 landspreading is restricted due to adverse weather conditions, soil type, soil
66 conditions, ground slope, proximity to water sources, and volumetric spreading
67 limitations. In Ireland, for example, there is a legal requirement to provide a DSW
68 storage capacity of 10 - 15 days (S.I. No. 31 of 2014), which results in increased
69 infrastructure and associated costs for the dairy farmer. These costs, combined with
70 the low nutrient replacement value of the DSW, mean that treatment and reuse may be
71 a better option for the farmer.

72

73 The environmental impacts of repeated spreading of DSW on lands are well
74 documented (e.g. Fenton et al., 2011), and may result in oxygen depletion and
75 asphyxiation of aquatic life in surface waters, as well as a risk of nutrient leaching to
76 groundwater (Knudsen et al., 2006). Long-term DSW application to lands may also
77 result in soil accumulation of phosphorus (P) and heavy metals and increase

78 concentrations of microbial pathogens, odorants and oestrogens in the receiving
79 environment (Wang et al., 2004; Hao et al., 2008). Hence, there is a real need for cost-
80 effective, low energy, and low maintenance on-farm treatment processes that would
81 result in a reduced risk of pollution following application to land. Some multi-stage
82 biological treatment processes, such as combined sequencing batch reactors (SBRs)
83 and constructed wetlands (CWs) (Moir et al., 2005), and aerated settling tanks
84 followed by vertical flow CWs (Merlin and Gaillot, 2010), have been used with
85 varying degrees of success; however, much of the organic and nutrient reductions in
86 these studies have been reported to occur in the aeration rather than in the passive
87 processes. Passive treatment systems such as sand filters (Rodgers et al., 2005; Healy
88 et al., 2007) and woodchip filters (Ruane et al., 2011; McCarthy et al., 2015) have
89 also been investigated and have reported consistently high levels of organic, nutrient
90 and pathogenic removal. Woodchip, in particular, is a cheap, biodegradable material
91 which has potential use as a soil improver (Cogliastro et al., 2001; Miller and
92 Seastedt, 2009) and has previously shown to be effective in improving effluent quality
93 and ammonia emissions when used in out-wintering pads (Dumont et al., 2012).

94

95 In order to realise the full potential of woodchip filters, it is necessary to determine
96 the optimum media depths which will produce consistently high quality effluent when
97 subjected to variable strength influent DSW loading. Filters are usually designed and
98 operated with one hydraulic regime selected to deliver an optimum organic loading
99 rate (OLR). However, as the concentration of DSW varies seasonally (Rodgers et al.,
100 2005), woodchip filters may be subjected to OLRs far in excess of their design
101 capacity. Therefore, it is necessary to examine the performance of filters under these
102 extreme conditions. Limited information is available on the impact of woodchip filter

103 depths and OLRs on the quality of treated DSW effluent. Additionally, no information
104 is available on the comparative performances of woodchip and sand filters when
105 treating on-farm DSW.

106

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

116

117 **2. Materials and Methods**

118 Eighteen filters, with internal diameters of 0.1 m and depths of either 0.6 m (n=3
119 columns) or 1 m (n=15 columns), were constructed using uPVC. All filters were open
120 at the top and sealed at the base using uPVC end caps. The columns were placed on
121 timber support frames and located in a temperature-controlled room at 10.6 ± 0.7 °C
122 and relative humidity of 86.9 ± 4.5 % (replicating the average temperature and
123 humidity in Ireland). A 0.075 m layer of clean, crushed pea gravel, manually sieved to
124 a particle size of 10 – 14 mm, was placed at the base of each column to prevent
125 washout of the filter media. Each column was then filled with either woodchip (with a
126 particle size of 10 – 20 mm) or sand (effective size, $d_{10} = 0.2$, uniformity coefficient,
127 $UC = 1.4$) by placing the selected media in 0.050 m lightly tamped increments.

128 Influent DSW was pumped intermittently (four times per day, seven days per week)
129 onto the filters using peristaltic pumps controlled by electronic timers. Hydraulic
130 loading rates were adjusted using the manual flow control on the pumps and influent
131 was distributed evenly across the surface of the filter media using perforated uPVC
132 flow distribution plates (Fig. 1). Continuously operated submersible mixers were
133 placed in each DSW influent container (one container per column set) to prevent
134 stratification. Treated effluent samples from each filter were collected in an effluent
135 collection container and all influent DSW samples were taken simultaneously from
136 the influent containers.

137

138 To clean any organic material from the media, 70 L of potable water was pumped
139 onto each filter over a period of 5 days prior to their operation, before being
140 intermittently loaded with DSW for a period of 56 days. On day 15 of operation, each
141 filter was seeded with 500 mL of nitrifying activated sludge (mixed liquor suspended
142 solids, MLSS = 6,290 mg L⁻¹; sludge volume index, SVI = 143) collected from a local
143 wastewater treatment plant. The period from day 0 to 56 was taken as the start-up
144 period to reach steady state operation (defined by consistent chemical oxygen demand
145 (COD), N and P effluent concentrations) for all filters and therefore day 56 was taken
146 as the effective start day of the study (day 0).

147

148 This study compared three different operational setups to examine the impacts of (1)
149 filter depth (2) OLR and (3) type of media (woodchip/sand) on filter performance.
150 The filter configurations (Fig. 2) were (1) 0.6 and 1 m deep woodchip filters operating
151 for 105 days with an average OLR of 120 g COD m⁻² d⁻¹ (2) 1 m deep woodchip
152 filters operating for 105 days with average OLRs of 50 and 155 g COD m⁻² d⁻¹, and

153 (3) 1 m deep woodchip and sand filters operating for 220 days with an average OLR
154 of 35 g COD m⁻² d⁻¹. All configurations and treatments were constructed and operated
155 at n=3. The very high OLRs (120 and 155 g COD m⁻² d⁻¹) were selected to assess the
156 performance of filters under extreme loading events, which may arise if a filter is
157 designed and hydraulically loaded assuming a low influent organic concentration.

158

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

165

166 The woodchip used was a commercial tree species, Sitca spruce (*Picea sitchensis*).
167 Logs were debarked and then chipped using an industrial wood chipping machine
168 (Morbark post peeler) at an industrial facility in northwest Ireland. The woodchips
169 were sieved to a 10 – 20 mm grading prior to placing in the filter columns. The sand
170 used was sourced from a commercial quarry in Co. Galway, West of Ireland and was
171 graded to a d₁₀ of 0.2 mm and a UC of 1.4. The permeability of the saturated
172 woodchip and sand (Table 2) was measured using the constant head permeability test
173 in accordance with BS 1377-5 (BSI, 1990).

174

175 The ability of the woodchip and sand media to remove N (measured as ammonium-N
176 (NH₄-N)) and P (measured as dissolved reactive phosphorus (DRP)) from the DSW
177 was investigated in a batch experiment by placing varying masses of the washed,

178 graded media in flasks (n=3) and adding 40 mL of raw DSW to each sample. All
179 samples were shaken for 24 h at 250 excursions per minute (epm) on a reciprocating
180 shaker and on removal, were allowed to settle for 1 h, filtered through a 0.45 µm
181 filter, and tested colorimetrically using a nutrient analyser (Konelab 20, Thermo
182 Clinical Laboratories Systems, Finland). The data were then modelled using a
183 Langmuir isotherm to establish maximum adsorption capacities (Table 2).
184
185 Influent samples and effluent taken from each filter column were tested for pH using a
186 pH probe (WTW, Germany) and for suspended solids (SS) using vacuum filtration on
187 a well-mixed subsample through Whatman GF/C (pore size 1.2 µm) filter paper. Sub-
188 samples were filtered through 0.45 µm filters and analysed colorimetrically for DRP,
189 NH₄-N, total oxidised nitrogen (TON) and nitrite-N (NO₂-N) using a nutrient
190 analyser (Konelab 20, Thermo Clinical Labsystems, Finland). Nitrate-N was
191 calculated by subtracting NO₂-N from TON. Unfiltered samples were tested for TP
192 and filtered (0.45 µm) samples for total dissolved phosphorus (TDP) using acid
193 persulphate digestion. Particulate phosphorus (PP) was calculated by subtracting TDP
194 from TP. Unfiltered samples were tested for TN using a BioTector Analyser
195 (BioTector Analytical Systems Ltd., Cork, Ireland) and for COD (dichromate
196 method). Influent DSW was tested for DM content by drying at 105 °C for 24 h. All
197 water quality parameters were tested in accordance with standard methods (APHA,
198 2005).

199

200 **2.1 Statistical analysis**

201 The data were analysed using independent sample t-tests in SPSS (IBM SPSS
202 Statistics 20 Core System) with column depth, OLRs and filter media as grouping

203 variables. The data were checked for normality and, where necessary, were log
204 transformed to satisfy the normal distributional assumptions required. Where
205 normality was not achieved, the non-parametric Mann Whitney U test was used.
206 Probability values of $p > 0.05$ were deemed not to be significant.

207

208 **3. Results and Discussion**

209

210 **3.1 Impact of media depth**

211 Treated effluent concentrations from the 1 m deep woodchip filters were consistently
212 lower than those from the 0.6 m deep filters for all measured parameters at an OLR of
213 $120 \text{ g COD m}^{-2} \text{ d}^{-1}$ (Fig. 3). However, the concentrations for COD in the final effluent
214 ($1469 \pm 587 \text{ mg L}^{-1}$ for the 0.6 m filter and $587 \pm 113 \text{ mg L}^{-1}$ for the 1 m filter) were still
215 far in excess of the limit value for discharge to urban waters (125 mg L^{-1} ; SI No 254
216 of 2001). The 0.6 m deep filters reduced COD, SS, TP and DRP by 46%, 54%, 7%
217 and 5%, respectively (based on average influent and effluent concentrations), but did
218 not reduce TN and $\text{NH}_4\text{-N}$ concentrations to below those of the influent. Reductions
219 of 78% COD, 95% SS, 85% TN, 82% $\text{NH}_4\text{-N}$, 50% TP and 54% DRP were measured
220 for the 1 m deep filters and were consistent with those of Ruane et al. (2011), who
221 measured reductions of 66% COD and 57% TN for 1 m deep woodchip filter pads
222 operating at an average OLR of $173 \pm 43 \text{ g COD m}^{-2} \text{ d}^{-1}$ for a 1 year period. These
223 findings indicate that filter depth is an important consideration in the design of
224 woodchip filters, as the 0.6 m deep filters did not provide sufficient detention time to
225 reduce COD and SS by more than approximately 50% at an average OLR of 120 g
226 $\text{COD m}^{-2} \text{ d}^{-1}$. These removals were increased by a factor of approximately 1.7 when
227 the filter depth was increased to 1 m with consequent increase in detention time.

228

229 Ammonium-N was not nitrified in any of the woodchip filters and this was most
230 likely as a result of the high average C:N ratio (30) of the influent DSW, which was
231 far above the optimum C:N ratio of 3 - 6 for nitrification (Henze et al., 2001; Eding et
232 al., 2006). This, combined with a high OLR ($120 \text{ g COD m}^{-2} \text{ d}^{-1}$), likely resulted in the
233 formation of a dense, non-porous heterotrophic biofilm structure, reducing the
234 available sites for the slow growing nitrifiers (Okabe et al., 1996; Wijeyekoon et al.,
235 2004; Nogueira et al., 2002). A nitrogen mass balance between influent and effluent
236 carried out on the 0.6 m deep filters showed that the mass of organic nitrogen (Norg)
237 was reduced by 23% while the mass of $\text{NH}_4\text{-N}$ increased by 8%, with no overall TN
238 removal. For the 1 m deep filters, the mass of Norg was reduced by 37% with a
239 corresponding reduction in $\text{NH}_4\text{-N}$ of 82% and an overall decrease in TN of 85%,
240 with $\text{NH}_4\text{-N}$ as the dominant fraction in the final effluent. Therefore, while significant
241 TN and $\text{NH}_4\text{-N}$ removals were achieved in the 1 m deep filters (85% and 82%,
242 respectively), the removal processes were by physical filtration of SS and associated
243 N (Fig. 4(A)) rather than biological transformations. Much lower SS removals were
244 measured in the 0.6 m deep filters (Fig. 3). The average pH of the treated effluent was
245 7.41 ± 0.26 , indicating that alkalinity was not an inhibiting factor for nitrification.
246 Ruane et al. (2011) reported an average concentration of $22.5 \text{ mg NO}_3\text{-N L}^{-1}$ in treated
247 effluent from 1 m deep woodchip filter pads loaded with DSW, which had an average
248 influent concentration of $12.9 \text{ mg NO}_3\text{-N L}^{-1}$ and C:N ratio of 16. In the current study,
249 there was no $\text{NO}_3\text{-N}$ in the influent and this may have influenced the biofilm
250 formation and consequent opportunity for development of $\text{NH}_4\text{-N}$ oxidizers (Okabe et
251 al., 1996).

252

253 **3.2 Impact of organic loading rates**

254 There were no significant differences in the final effluent concentrations of $\text{NH}_4\text{-N}$
255 (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
256 filters operated at OLRs of 50 and 155 $\text{g COD m}^{-2} \text{d}^{-1}$; however, the average effluent
257 DRP concentration (3.8 ± 1.5 mg L^{-1}) from the 50 $\text{g COD m}^{-2} \text{d}^{-1}$ filters was
258 significantly lower ($p < 0.001$) than from the 155 $\text{g COD m}^{-2} \text{d}^{-1}$ filters (10.2 ± 2.9 mg L^{-1}).
259 As the woodchip had no ability to adsorb P (Table 2), physical removal was the
260 main mechanism for P removal. Based on the influent and effluent loading rates, 2.5
261 mg PP d^{-1} (318 $\text{mg PP m}^{-3} \text{d}^{-1}$) was retained in the 155 $\text{g COD m}^{-2} \text{d}^{-1}$ filters, whereas
262 0.4 mg PP d^{-1} (51 $\text{mg PP m}^{-3} \text{d}^{-1}$) was retained in the 50 $\text{g COD m}^{-2} \text{d}^{-1}$ filters.

263

264 Removals (based on the average influent and effluent load and expressed in mg d^{-1}) in
265 the range of 71% to 97% were measured for COD, SS, TN and $\text{NH}_4\text{-N}$, and 54% to
266 74% for TP and DRP, were measured in both sets of filters. Final effluent
267 concentrations of SS, $\text{NH}_4\text{-N}$ and DRP ranged from 23 to 37 mg L^{-1} , 4.1 to 4.6 mg L^{-1} ,
268 and 3.8 to 10.4 mg L^{-1} , respectively. However, the final effluent COD concentrations
269 from both filters (766 ± 221 mg L^{-1} for the 50 $\text{g COD m}^{-2} \text{d}^{-1}$ filters and 604 ± 112 mg L^{-1}
270 for the 155 $\text{g COD m}^{-2} \text{d}^{-1}$ filters) were well above the limit values for discharge to
271 urban waters in Ireland (S.I. No 254 of 2001). Effluent mass loads for COD, SS, $\text{NH}_4\text{-N}$
272 and DRP (Fig. 5) remained consistent over the duration of the study period,
273 highlighting the capacity of the filters to effectively and consistently treat variable
274 strength and variably loaded influent DSW.

275

276 Negligible $\text{NO}_3\text{-N}$ concentrations were measured in the effluent, underlining the
277 reliance on physical filtration for $\text{NH}_4\text{-N}$ removal as illustrated by the close
278 correlations between SS and $\text{NH}_4\text{-N}$ mass removals for both loading rates (Fig 4(B)).

279

280 **3.3 Impact of filter media**

281 There were no significant differences between the treated effluent from 1 m deep
282 woodchip and 1 m deep sand media (average OLR = $35 \text{ g COD m}^{-2} \text{ d}^{-1}$) for SS (23 ± 13
283 and $16 \pm 20 \text{ mg L}^{-1}$) and $\text{NH}_4\text{-N}$ (2.9 ± 3.4 and $0.8 \pm 0.5 \text{ mg L}^{-1}$); however, the sand
284 outperformed the woodchip in COD removal (a final effluent of $146 \pm 52 \text{ mg L}^{-1}$ versus
285 $873 \pm 242 \text{ mg L}^{-1}$) and DRP removal up to day 150 (a final effluent of $0.1 \pm 0.1 \text{ mg DRP}$
286 L^{-1} versus $4.9 \pm 2.7 \text{ mg DRP L}^{-1}$). The enhanced COD removals in the sand filters were
287 reflective of their higher hydraulic retention time when compared to the woodchip
288 filters (the hydraulic conductivity of the sand was >40 times lower than that of the
289 woodchip (Table 2)). The enhanced DRP removals in the sand filters were as a result
290 of their higher P adsorption capacity ($136 \text{ g DRP kg}^{-1}$) compared with the woodchip,
291 which had no affinity for P, and DRP reductions in the woodchip filters were
292 associated with SS removals (Fig. 4(C)). After 150 days of operation, DRP
293 breakthrough occurred quite quickly in the sand filters and at a slower rate in the
294 woodchip filters (Fig. 3). From day 200 to the end of the study, neither the sand nor
295 the woodchip filters removed any DRP from the influent DSW (Fig. 3). The average
296 mass of P retained up to day 150 was 1.61 ± 1.30 and $3.89 \pm 0.76 \text{ mg TP d}^{-1}$, 0.61 ± 0.31
297 and $0.96 \pm 0.32 \text{ mg PP d}^{-1}$ and 1.33 ± 0.84 and $2.58 \pm 0.60 \text{ mg DRP d}^{-1}$ for woodchip and
298 sand filters, respectively, indicating that the sand was more effective at removing PP
299 and also had a greater affinity for adsorption of DRP (Table 2). The mass removal

300 rates also indicate that sand had more consistent P removal than woodchip up to day
301 150.

302

303 During the first 85 days of operation, nitrification occurred in the sand filters and the
304 $\text{NO}_3\text{-N}$ concentration rose from $0.1\pm 0.1 \text{ mg L}^{-1}$ in the influent to $43\pm 18 \text{ mg L}^{-1}$ in the
305 effluent. However, the effluent $\text{NO}_3\text{-N}$ subsequently reduced considerably, and
306 attained an average concentration of $7.2\pm 1.6 \text{ mg L}^{-1}$ by the end of the study (Fig. 3).

307 The reasons for the suppressed levels of $\text{NO}_3\text{-N}$ were possibly due to the preferential
308 formation of heterotrophic-dominated biofilm layers limiting dissolved oxygen (DO)
309 to the nitrifiers (Nogueira et al., 2002) as a consequence of the high influent C:N
310 ratios in the influent wastewater (average of 38). Negligible $\text{NO}_3\text{-N}$ concentrations
311 were measured in the treated effluent from the woodchip filters and were always
312 below $0.21\pm 0.19 \text{ mg L}^{-1}$. This indicates that even at the low OLRs used in this study,
313 which are at the upper limit at which nitrification normally occurs in sand filters
314 treating a similar type of wastewater (around $30 \text{ g COD m}^{-2} \text{ d}^{-1}$; Rodgers et al., 2005),
315 woodchip filters are unable to nitrify DSW.

316

317 **3.4 Assessment of optimum filter media, configuration and operation**

318 When assessing the suitability of the filters to treat on-farm DSW, key operating
319 criteria must be taken into account, together with the main objective of reducing
320 organic and nutrient concentrations to levels which would not adversely impact the
321 environment if landspread. These operating criteria include items such as cost and
322 availability of the media, robustness and longevity of performance (i.e. how well can
323 media deal with daily and seasonal variations in flow and strength and for how long),
324 biodegradability, and disposal of spent media.

325

326 The results of this study show that woodchip filters should have a minimum depth of
327 1 m to achieve required removals and can reduce the measured water quality
328 parameters at OLRs up to at least $155 \text{ g COD m}^{-2} \text{ d}^{-1}$. However, based on the N mass
329 balances and effluent concentrations of $\text{NO}_3\text{-N}$ measured in this study, the removal
330 mechanisms in woodchip filters are primarily physical (straining) and not biological
331 (nitrification did not occur). The suppression of biological activity may have been a
332 function of the OLRs employed in this study, where the lowest OLR studied (35 g
333 $\text{COD m}^{-2} \text{ d}^{-1}$) was still at the upper limit at which nitrification normally occurs in
334 filters (Rodgers et al., 2005).

335

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

346

347 Surface clogging of the filter media is an operational issue that must be considered for
348 on-farm use and while neither the sand nor the woodchip media in this study
349 experienced surface clogging, Healy et al. (2007) reported clogging of sand filters

350 after 42 days at an OLR of $43 \text{ g COD m}^{-2} \text{ d}^{-1}$. In contrast, we are not aware of any
351 reported issues with surface clogging of woodchip media, and it has been estimated
352 that a woodchip filter may be operational for 2 – 3 years before surface ponding
353 occurs (Ruane et al., 2011).

354

355 The decision to use woodchip or sand filter media is ultimately taken by synthesizing
356 environmental benefits versus capital and operating costs. Operating costs are similar
357 for both woodchip and sand filters (the modes of operation are identical for both),
358 while capital costs are differentiated only by the cost of the media (filter setup for
359 woodchip and sand are similar), which may also not differ significantly and will be
360 location specific. Cost comparisons therefore can be made by comparing the required
361 footprint of woodchip and sand media, both at a depth of 1 m – the minimum
362 acceptable filter depth identified in this study. Based on the optimal OLRs identified
363 in this study (an OLR of $155 \text{ g COD m}^{-2} \text{ d}^{-1}$ for woodchip filters, which treated the
364 wastewater through physical processes, if not necessarily biological processes, and an
365 OLR of $35 \text{ g COD m}^{-2} \text{ d}^{-1}$ for sand filters, which only temporarily caused the
366 occurrence of nitrification, but clearly was at the upper OLR limit at which such
367 filters may be operated), a filter surface area of $0.48 \text{ m}^2 \text{ cow}^{-1}$ for woodchip versus 2.1
368 $\text{m}^2 \text{ cow}^{-1}$ for sand would be required (Table 3). The larger area required for the sand
369 filter combined with their lack of robustness to deal with shock loads (Healy et al.,
370 2007) and the potential for surface clogging (Rodgers et al., 2005), indicate that
371 woodchip filters are a better on-farm treatment option.

372

373 The optimal filter configuration identified in the current study produced a final
374 effluent that was in excess of permissible discharge standards. For the water to be

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

389

390 **4. Conclusions**

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

399

400

401 **References**

- 402 Augustenborg, C.A., Carton, O.T., Schulte, R.P.O., Suffet, I.H. 2008a. Response of
403 silage yield to land application of out-wintering pad effluent in Ireland.
404 *Agricultural Water Management* 95: 367-374.
405
- 406 Augustenborg, C.A., Carton, O.T., Schulte, R.P.O., Suffet, I.H. 2008b. Silage dry-
407 matter yield and nitrogen response following land application of spent timber
408 residue from out-wintering pads to Irish grassland. *Communications in Soil
409 Science and Plant Analysis* 39: 1122 – 1137.
410
- 411 APHA, 2005. Standard methods for the examination of water and wastewater,
412 American Public Health Association, American Water Works Association, Water
413 Pollution Control Federation, Washington, DC.
414
- 415 British Standards Institution. 1990. British standard methods of test for soils for civil
416 engineering purposes. Part 5: Compressibility, permeability and durability tests.
417 BS 1377-5:1990. BSI, London.
418
- 419 Cannon, A.D., Gray, K.R., Biddlestone, A.J., Thayanithy, K. 2000. SE—structures
420 and environment: pilot-scale development of a bioreactor for the treatment of
421 dairy dirty water. *Journal of Agricultural Engineering Research* 77: 327-334.
422
- 423 Carney, K., Rodgers, M., Lawlor, P.G., Zhan, X. 2011. A sustainable technology for
424 the treatment of piggery wastewaters. In: *Proceedings of the Global Conference
425 on Global Warming*, 11 – 14 July, Lisbon, Portugal.
- 426 Cogliastro, A., Domon, G., Daigle, S., 2001. Effects of wastewater sludge and
427 woodchip combinations on soil properties and growth of planted hardwood trees
428 and willows on a restored site. *Ecological Engineering* 16: 471-485.
- 429 Dumont, P.A., Chadwick, D.R., Misselbrook, T.H., Robinson, J.S., Smith, K.A.,
430 Sagoo, E., Camp, V., Murray, R., French, P., Hill, R.A., Scott, A., 2012. Effluent
431 quality and ammonia emissions from out-wintering pads in England, Wales and
432 Ireland. *Agriculture, Ecosystems & Environment* 160: 82-90.
- 433 Eding, E.H., Kamstra, A., Verreth, J.A.J., Huisman, E.A., Klapwijk, A., 2006. Design
434 and operation of nitrifying trickling filters in recirculating aquaculture: a review.
435 *Aquacultural Engineering* 34: 234-260.
- 436 Hao, X., Godlinski, F., Chang, C., 2008. Distribution of phosphorus forms in soil
437 following long-term continuous and discontinuous cattle manure applications.
438 *Soil Science Society of America Journal* 72: 90-97.
- 439 Joy, D., Weil, C., Crolla, A., Bonte-Gelok, S., 2001. New technologies for on-site
440 domestic and agricultural wastewater treatment. *Canadian Journal of Civil
441 Engineering* 28: 115-123.
- 442 Fenton, O., Serrenho, A., Healy, M.G. 2011. Evaluation of amendments to control
443 phosphorus losses in runoff from dairy-soiled water. *Water, Air, & Soil Pollution*
444 222: 185-194.

- 445 Healy, M.G., Rodgers, M. and Mulqueen, J. 2007. Performance of a stratified sand
446 filter in removal of chemical oxygen demand, total suspended solids and
447 ammonia nitrogen from high-strength wastewaters. *Journal of Environmental*
448 *Management* 83: 409-415.
- 449 Healy, M., Rodgers, M., Walsh, G. 2011. Different depth sand filters for laboratory
450 treatment of synthetic wastewater with concentrations close to measured septic
451 tank effluent. *Journal of Environmental Science and Health, Part A:*
452 *Toxic/Hazardous Substances and Environmental Engineering* 46: 80 – 85.
- 453 Henze, M., Harremoës, P., la Cour Jansen, J., Arvin, E. 2001. *Wastewater treatment:*
454 *biological and chemical processes.* Springer Science & Business Media.
- 455 Knudsen, M.T., Kristensen, I.S., Berntsen, J., Petersen, B.M., Kristensen, E.S. 2006.
456 Estimated N leaching losses for organic and conventional farming in Denmark.
457 *The Journal of Agricultural Science* 144: 135-149.
- 458 Lekang, O.I. and Kleppe, H. 2000. Efficiency of nitrification in trickling filters using
459 different filter media. *Aquacultural Engineering*, 21: 181-199.
- 460 Longhurst, R.D., Roberts, A.H.C., O'Connor, M.B. 2000. Farm dairy effluent: a
461 review of published data on chemical and physical characteristics in New
462 Zealand. *New Zealand Journal of Agricultural Research* 43: 7-14.
- 463 McCarthy, G., Lawlor, P.G., Carney, K.N., Zhan, X., Gutierrez, M., Gardiner, G.E.
464 2015. An investigation into the removal of Salmonella and enteric indicator
465 bacteria from the separated liquid fraction of raw or anaerobically digested pig
466 manure using novel on-farm woodchip biofilters. *Science of the Total*
467 *Environment* 514: 140-146.
- 468 McFarland, A.M.S., Hauck, L.M., Kruzic, A.P. 2003. Phosphorus reductions in runoff
469 and soils from land-applied dairy effluent using chemical amendments: An
470 observation. *The Texas Journal of Agriculture and Natural Resource* 16: 47-59.
- 471 Merlin, G., Gaillot, A. 2010. Treatment of dairy farm effluents using a settling tank
472 and reed beds: performance analysis of a farm-scale system. *Transactions of the*
473 *ASABE* 53: 1681-1688.
- 474 Miller, E.M., Seastedt, T.R. 2009. Impacts of woodchip amendments and soil nutrient
475 availability on understory vegetation establishment following thinning of a
476 ponderosa pine forest. *Forest Ecology and Management*, 258: 263-272.
- 477 Minogue, D., French, P., Bolger, T. and Murphy, P.N.C. 2015. Characterisation of
478 dairy soiled water in a survey of 60 Irish dairy farms. *Irish Journal of*
479 *Agricultural and Food Research* 54: 1-16.
- 480 Moir, S.E., Svoboda, I., Sym, G., Clark, J., McGechan, M.B., Castle, K. 2005. An
481 experimental plant for testing methods of treating dilute farm effluents and dirty
482 water. *Biosystems Engineering* 90: 349-355.
- 483 Nogueira, R., Melo, L.F., Purkhold, U., Wuertz, S. and Wagner, M., 2002. Nitrifying
484 and heterotrophic population dynamics in biofilm reactors: effects of hydraulic
485 retention time and the presence of organic carbon. *Water Research* 36: 469-481.

- 486 Okabe, S., Oozawa, Y., Hirata, K. and Watanabe, Y. 1996. Relationship between
487 population dynamics of nitrifiers in biofilms and reactor performance at various
488 C: N ratios. *Water Research* 30: 1563-1572.
- 489 Rodgers, M., Healy, M.G., Mulqueen, J. 2005. Organic carbon removal and
490 nitrification of high strength wastewaters using stratified sand filters. *Water*
491 *Research* 39: 3279-3286.
- 492 Ruane, E.M., Murphy, P.N., Healy, M.G., French, P., Rodgers, M. 2011. On-farm
493 treatment of dairy soiled water using aerobic woodchip filters. *Water Research*
494 45: 6668-6676.
- 495 SI No. 31 of 2014. European Union (Good agricultural practice for protection of
496 waters) Regulations 2014.
497 <https://www.agriculture.gov.ie/media/migration/ruralenvironment/environment/nitrates/SI31of2014290114.pdf>
498
- 499 SI No 254 of 2001. Urban waste water treatment regulations, 2001.
500 <http://www.irishstatutebook.ie/eli/2001/si/254/made/en/print>
- 501 Sweeten, J.M., Wolfe, M.L. 1994. Manure and wastewater management systems for
502 open lot dairy operations. *Transactions of the ASAE* 37: 1145-1154.
- 503 Wang, H., Magesan, G.N., Bolan, N.S. 2004. An overview of the environmental
504 effects of land application of farm effluents. *New Zealand Journal of*
505 *Agricultural Research* 47: 389-403.
- 506 Wijeyekoon, S., Mino, T., Satoh, H., Matsuo, T. 2004. Effects of substrate loading
507 rate on biofilm structure. *Water Research* 38: 2479-2488.

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

Parameter	Average \pm standard deviation
COD (mg L ⁻¹)	2798 \pm 1503
SS (mg L ⁻¹)	874 \pm 614
TN (mg L ⁻¹)	81.5 \pm 34.1
NH ₄ -N (mg L ⁻¹)	63.9 \pm 32.3
TP (mg L ⁻¹)	29.8 \pm 14.4
DRP (mg L ⁻¹)	24.3 \pm 16.0
pH	7.22 \pm 0.71
Dry matter (%)	0.2 \pm 0.1

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	-

Table 3. Comparative filter areas (per cow) of a full scale filter for average 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 ¹ (L d ⁻¹ cow ⁻¹)	COD load ² (g COD d ⁻¹)	Filter area per cow (m ²)	
		Woodchip ³	Sand ⁴
27	73.7	0.48	2.1

¹Minogue et al., 2015;

²Assuming an annual average COD concentration of 2,750 mg L⁻¹;

³Using an OLR of 155 g COD m⁻² d⁻¹;

⁴Using an OLR of 35 g COD m⁻² d⁻¹.

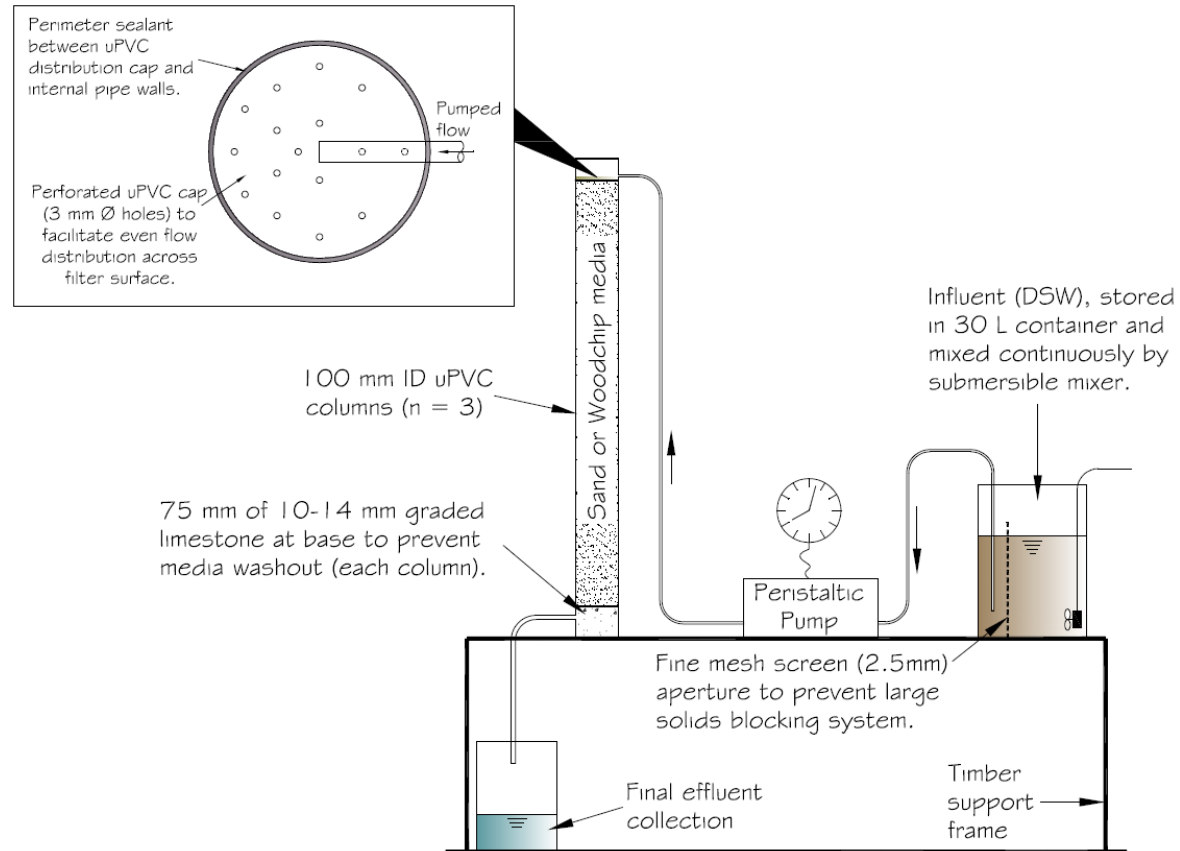


Fig. 1: Schematic diagram of typical laboratory filter setup. (Not to scale)

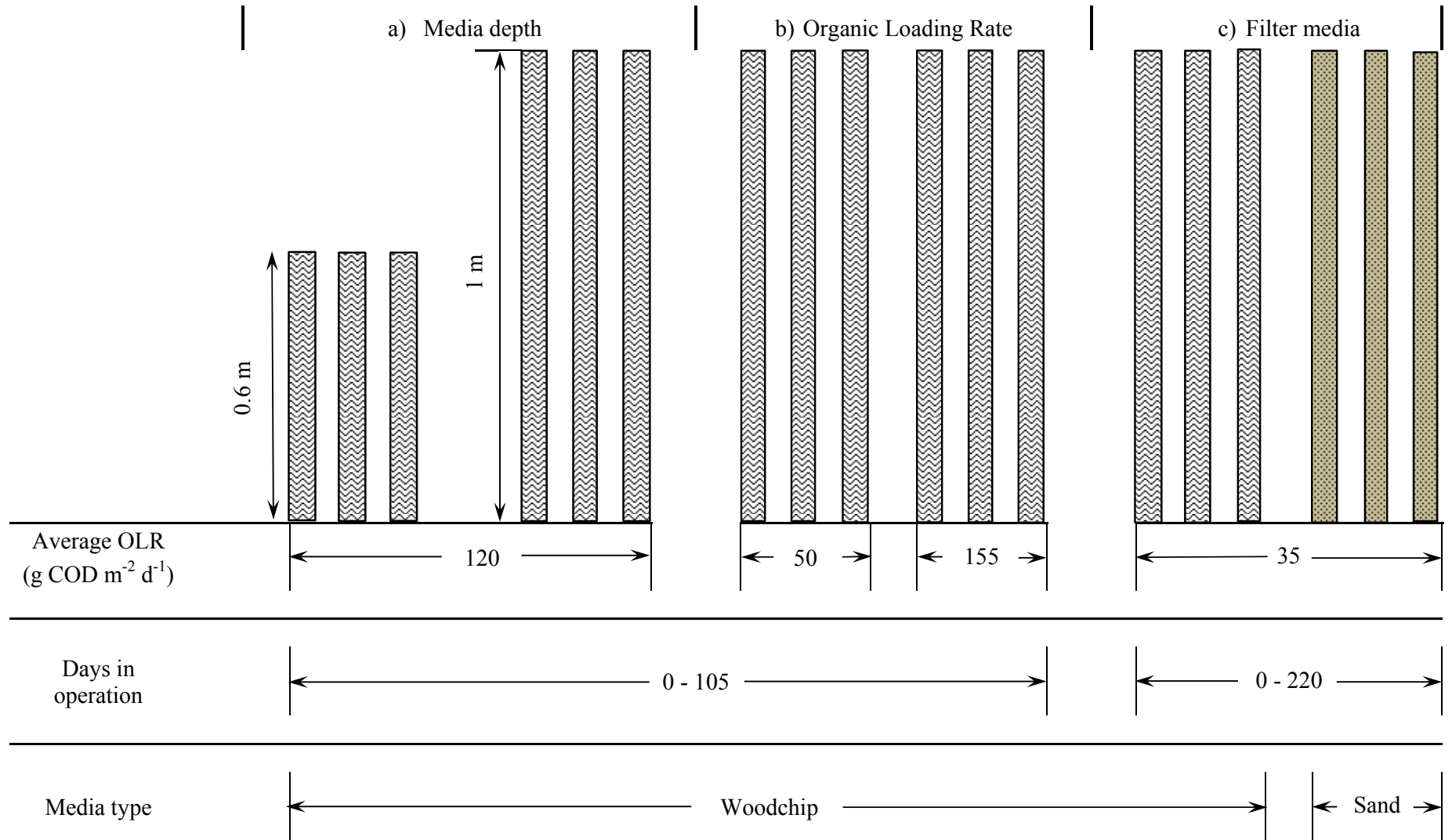


Fig. 2 Combinations of a) media depth, b) organic loading rates and c) filter media used in this study. The woodchip used was 10 - 20 mm Sitka spruce (*picea sitchensis*). The sand used had a $d_{10} = 0.2$ mm and a uniformity coefficient (UC) = 1.4.

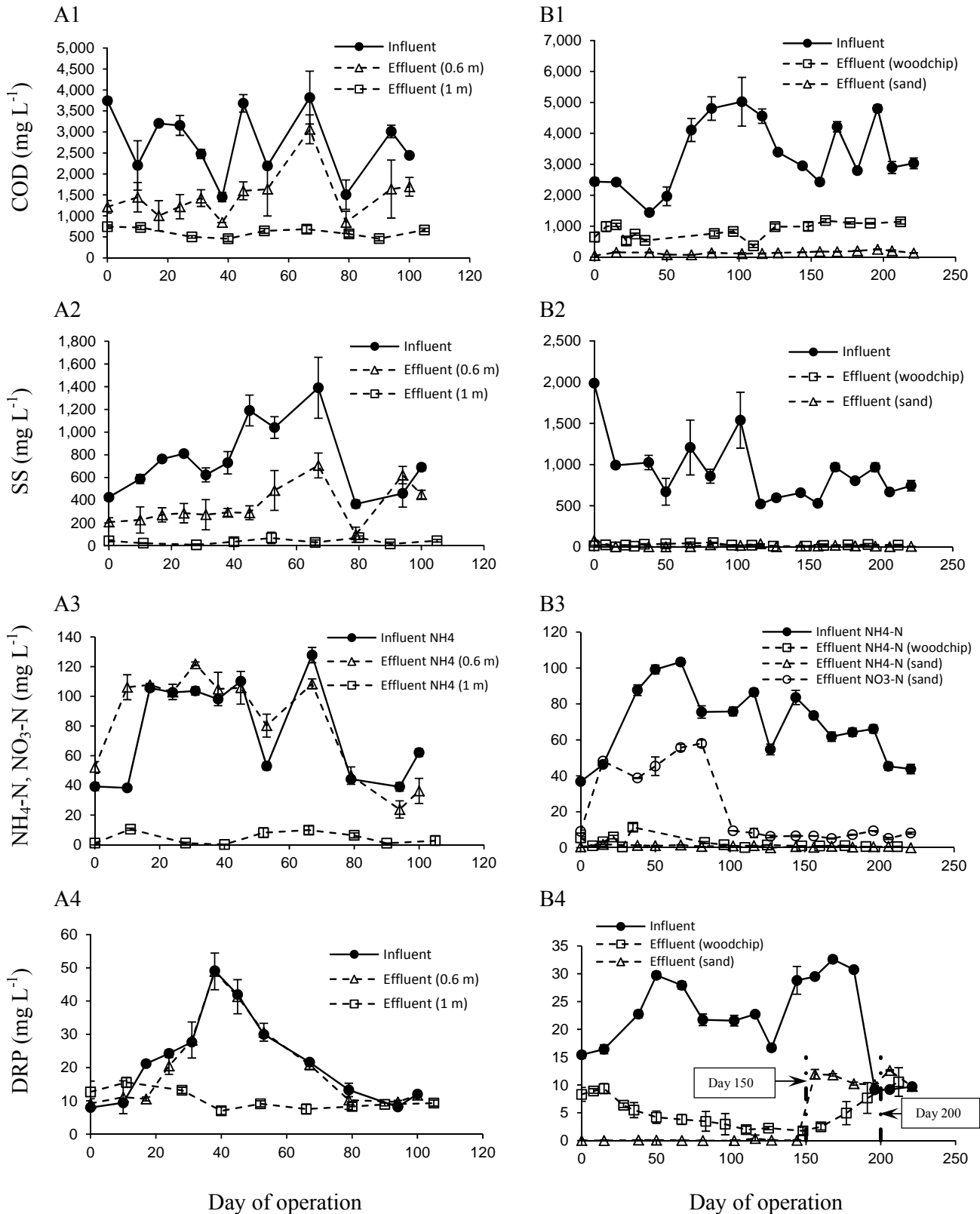


Fig. 3 Impact of media depth (A1 – A4) and media type (B1 – B4) on COD, SS, NH₄-N and DRP removals. An average organic loading rate of 120 g COD m⁻² d⁻¹ was applied to woodchip media (10 – 20 mm Sitka spruce) when comparing the impact of media depth (A1 – A4). An average organic loading rate of 35 g COD m⁻² d⁻¹ was applied to woodchip (10 – 20 mm Sitka spruce) and sand (d₁₀ = 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.

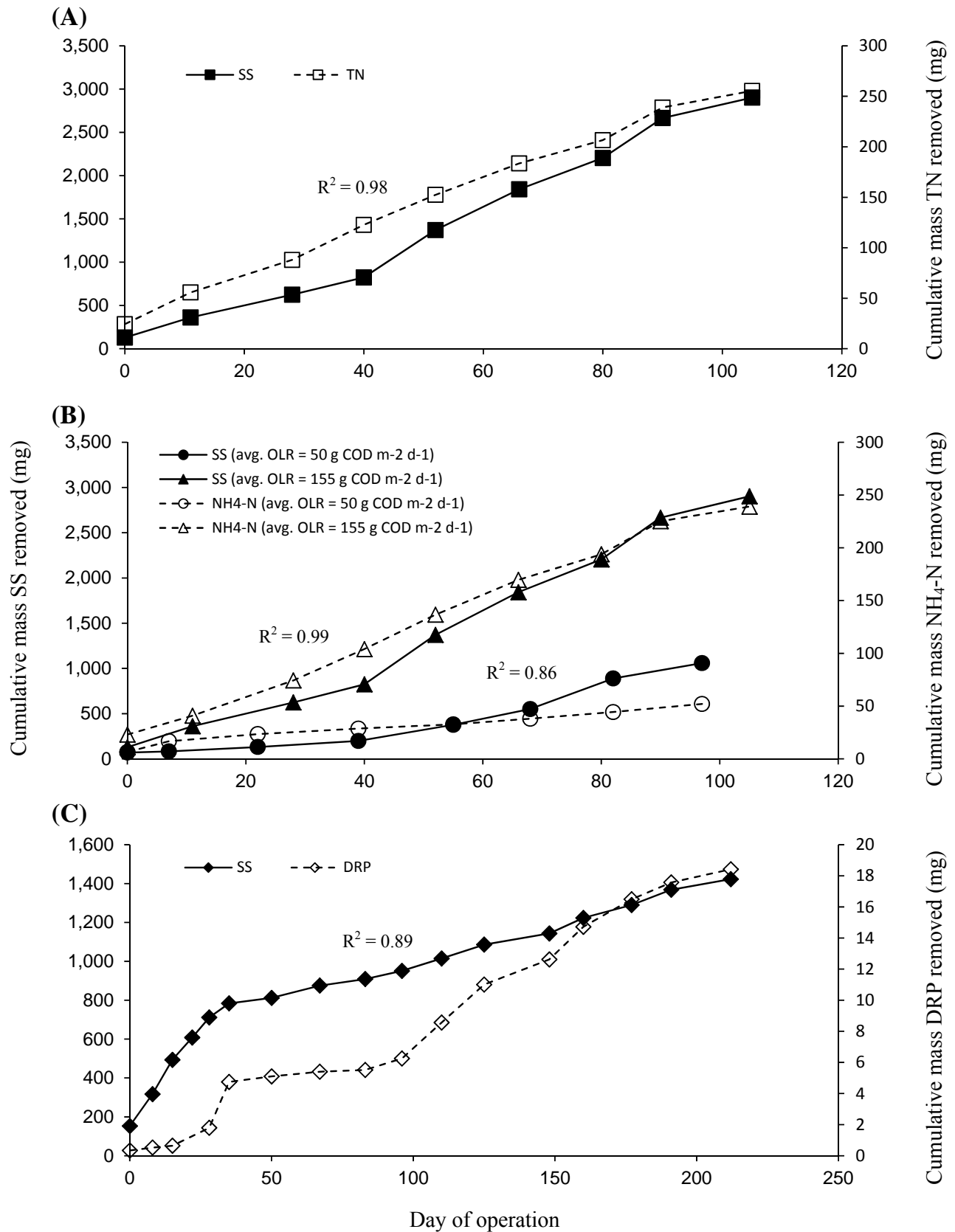


Fig. 4 Correlations between cumulative mass removals of suspended solids (SS) for 1 m deep \times 0.1 m \varnothing woodchip filters ($n=3$, each set) and **(A)** TN loaded at $120 \text{ g COD m}^{-2} \text{ d}^{-1}$ **(B)** $\text{NH}_4\text{-N}$ loaded at 50 and $155 \text{ g COD m}^{-2} \text{ d}^{-1}$ respectively and **(C)** DRP loaded at $35 \text{ g COD m}^{-2} \text{ d}^{-1}$. Correlation coefficients, (R^2) indicated.

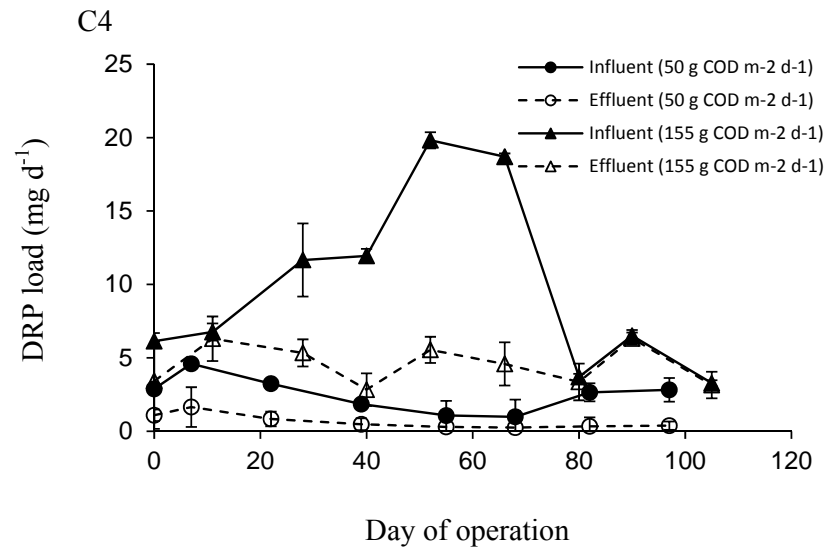
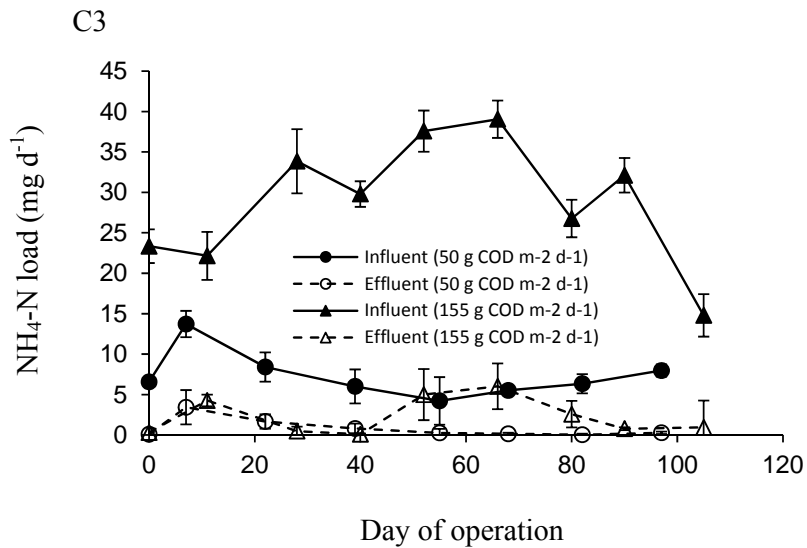
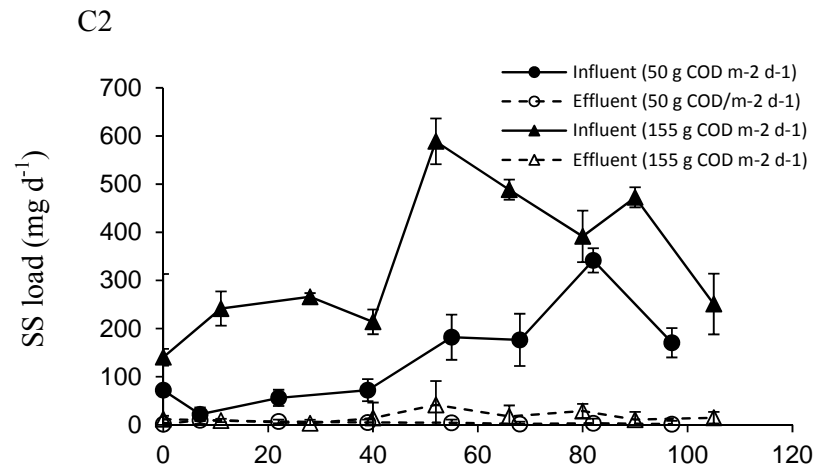
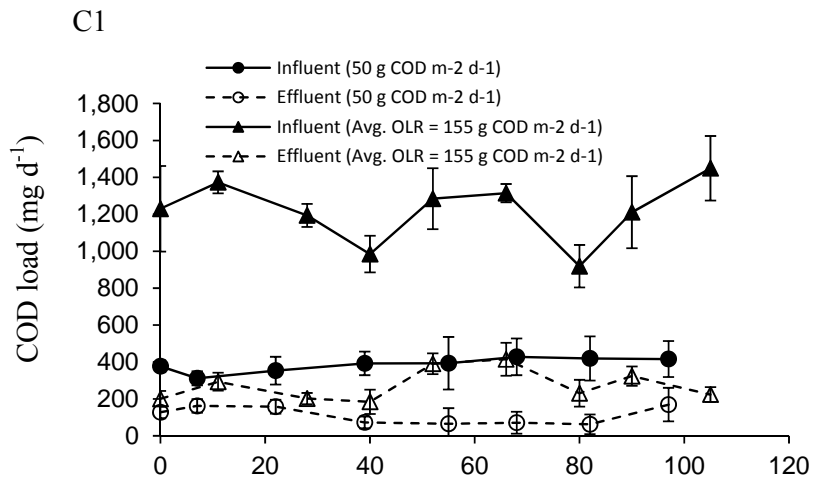


Fig. 5 Impact of organic loading rates on COD, SS, NH₄-N and DRP mass removals. The filter material used was 10 – 20 mm Sitka spruce woodchip, 1 m deep. Error bars indicate standard deviations.

