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DENITRIFICATION BIOREACTOR WORK IN WAITUNA LAGOON CATCHMENT, SOUTHLAND

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Abstract

Subsurface drainage of agricultural land provides multiple "conduits" that potentially facilitate transport of contaminants from field to stream. In addition to increasing the efficiency of transport, these "conduits" may also allow contaminants to bypass natural attenuation zones and processes. Adoption of tile or mole drainage has contributed to the trend of increasing nitrate-nitrogen loads observed in many rural streams. Despite creating the potential to degrade downstream water quality, relatively few long-term, field-scale trials have been undertaken to quantify the hydraulic and contaminant load from these drainage systems.

We quantified the mass load of nitrogenous material derived from a representative subsurface drain, and the reduction in nitrate-nitrogen provided by a woodchip filter during a 15-month trial. The tile drain delivers seasonally-varying concentrations and mass of contaminants, driven primarily by rainfall. The annual median concentrations of nitrate-nitrogen entering the filter (i.e. discharged from the tile drain), and discharged from the filter were 2190 µg/L and 421.5 µg/L, respectively. The median daily load was reduced from 47.1 g/d to 2.9 g/d (a reduction of 93.8 percent), and the median nitrate-nitrogen yield from the field was reduced from 1.83 kg/ha/yr to 0.11 kg/ha/yr.

Efficacy of the woodchip treatment filter was strongly dependent on hydraulic retention time, and to a lesser extent, temperature. These dependencies were evident as seasonally varying treatment efficacy. These factors should be considered before modifying the design of a woodchip filter to enhance efficacy. For example, if seasonally large hydraulic loading determines treatment efficacy during autumn and winter, it may be possible to improve performance by buffering the flow (e.g. by incorporating a buffer chamber into the filter design), or by temporarily retaining some of the drainage water in the landscape, within the drains themselves. Another approach could include a bypass system to limit inflow to the filter once a defined flow threshold was achieved. Several of these and other approaches may be combined to accommodate site-specific considerations (e.g. soil, gradient and farming system factors). It is also conceivable that seasonally varying treatment efficacy may be consistent with the seasonality of the receiving environment and with water quality improvement goals.

Introduction

Although drainage has substantially increased the land area available for intensive agriculture, this practice has substantially altered the hydrology of landscapes relative to pre-human conditions (Arenas Amado et al. 2017). These changes include transformation of aquatic habitat types from wetland mosaics to linear systems, with reduced surface storage and increased conveyance and drainage (Blann et al. 2009).

Subsurface drainage provides a conduit that facilitates transport of contaminants from field to stream, bypassing natural attenuation processes (Houlbrooke et al. 2008; Blann et al. 2009; Frankenberger et al. 2017; Villeneuve 2017). Tile drainage is typically implemented in shallow gradient, poorly drained, land adjacent to low-order streams, but may also be implemented on steeper slopes to reduce the hydraulic load on lower-lying shallow gradient land. Adoption of tile or mole drainage (both examples of subterranean drainage) has contributed to the increasing trend in nitrate-nitrogen loads observed in many rural streams. In the Mississippi River basin, nitrogen export has increased two to seven-fold in the last century, with direct losses from surface agricultural drainage ranging from 1-50 kg/ha/year, and higher loss rates from sub-surface drainage (2-100 kg/ha/year) (Blann et al. 2009). Subsurface drainage is also associated with increased phosphorus losses from agricultural lands, particularly under baseflow conditions (Jordan et al. 2012).

The problem of excessive nutrient loss from agricultural landscapes has been recognised and various attenuation filters, walls etc have been developed, particularly for nitrogen (Christianson 2011; Addy et al. 2016). These treatment facilities allow interaction between contaminant and electron donors (organic carbon) under anaerobic conditions where microbial population converts nitrate-N (ultimately) to nitrogen gas, which is lost to the atmosphere. These filters have also found application in reducing the N load from urban areas, roads, on-site wastewater treatment systems, and in conjunction with other nutrient mitigation strategies (e.g., Woli et al. 2010; Christianson et al. 2016; Arenas Amado et al. 2017; Villeneuve 2017).

Although the efficacy of nitrate-N filters has been published for several laboratory and field trials, these results tend to be observational, and based on short-term assessments. The work described here quantifies the discharge from a tile drain system underlying a cultivated field in Southland, New Zealand over a 15-month period, quantifying the nitrogen yield from the tile drainage system, and the removal efficacy of a passive, woodchip filter.

The Waituna Estuary catchment, Southland, New Zealand, was chosen for the trial. The numbers of dairy cattle have increased dramatically over the few decades, raising concerns about water quality generally, and specifically regarding impacts on the iconic Waituna Estuary, which has strong significance to Maori and other stakeholders. This has led to the development of several strategies and action plans (e.g., Lagoon Technical Group et al. 2013; Waituna Partners' Group 2015), and associated strands of investigation (e.g., Diffuse Sources and NIWA 2012; Tanner et al. 2013). The site chosen for the trial of a woodchip filter was determined using several criteria including: adequate grade (allowing for a gravity-fed system), a reasonably defined catchment source area, soils and lithology that made the farming system susceptible to N loss through the root zone, and existence of tile drainage that could be intercepted and directed to a filter bed (McKergow et al. 2015). The location of the N-filter and Waituna Lagoon in Southland, New Zealand is indicated in Figure 1.

Figure 1: Location of the N filter and Waituna Lagoon, Southland.

Design and construction

The basic design of the woodchip filter is shown in Figure 2. Features include inlet- and outlet-flow measurement equipment, a perforated pipe across the inlet side of the woodchip filter, which ensured flow was distributed across the filter bed, and the 10 m × 10 m × 1 m bed. For the period of the trial, the water level in the bed was maintained at approximately 700 mm depth.

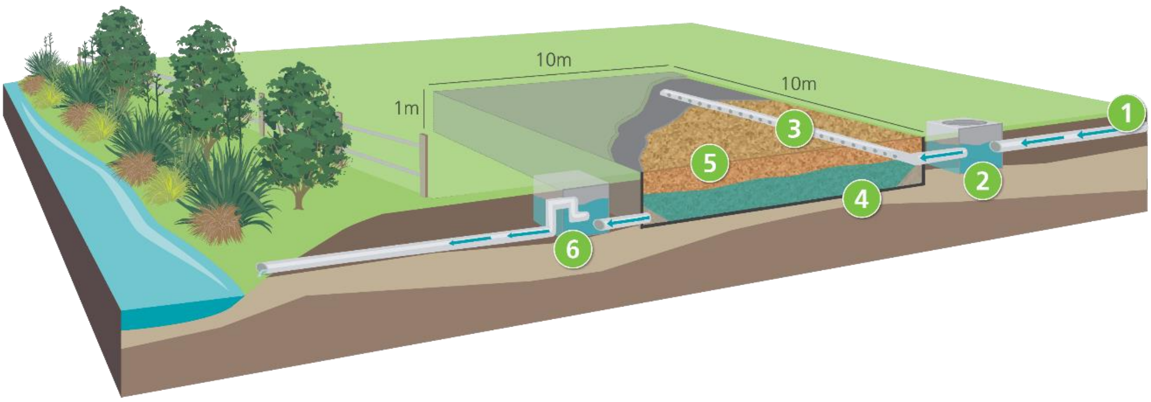

Figure 2: Schematic of the N-filter. 1) = inflow from two tile drains; 2) inlet flow measurement structure; 3) = inlet flow distribution manifold; 5) = woodchip filter bed; 6) = outlet flow measurement structure.

Measurement and monitoring

Flow was measured continuously in the inlet and outflow measurement structures. Other continuous measurements included water temperature, turbidity, electrical conductivity and rainfall. Flow-proportional grab samples were collected in the inflow and outflow structures using Isco automatic samplers. Samples were preserved using mercuric chloride. These samples were sent to the NIWA Hamilton water quality laboratory for analysis. All samples were analysed for ammoniacal-N, nitrate- and nitrite-N, and total nitrogen. A smaller selection of samples were analysed for turbidity, electrical conductivity, dissolved- and total organic carbon, and dissolved- and total phosphorus.

Nitrate removal performance was assessed by comparing inflow and outflow loads of nitrate-N. Load calculation was calculated as the product of flow (measured at the time of sample collection) and concentration, expressed as mass per unit of time (e.g. g/day).

Relatively few grab samples were collected. To estimate loads at times when concentrations were not measured, several modelling techniques were used, including “rating table” procedures, based on the relationship between concentration and flow, and more advanced procedures included in the LOADEST modelling suite (Runkel et al. 2004).

Removal efficacy (reduction in mass relative to influent load, per unit time) is expressed in Equation 1:

$$\text{Removal efficacy (\%)} = \left(\frac{\text{influent mass-effluent mass}}{\text{influent mass}}\right) \times 100 \qquad \text{Eq. 1}$$

Exploratory data analysis, and generation of figures and summary statistics was undertaken using Systat v13.[1] Systat was also used to calculate removal efficiencies and nutrient fluxes as required.

Results

Rainfall depths recorded at the site are compared with long-term precipitation records derived from the Invercargill Airport Automatic weather station in Figure 3. The rainfall generally follows the long-term seasonal average trend, but there were several months when rainfall was less than half the average, and one month when it greatly exceeded the long-term average.

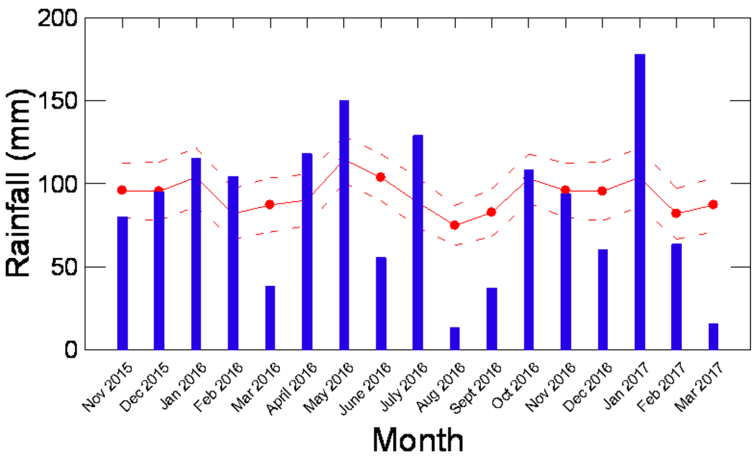

Figure 3: Comparison of on-site rainfall (blue bar) with long-term monthly average rainfall (red dot). The long-term average rainfall is from the Invercargill Automatic Weather Station site and represents the period January 1990 – December 2017. The upper and lower broken lines indicate the lower and upper 95th percent confidence intervals of the long term monthly average value.

Total monthly rainfall exceeded 100 mm on nine of 15 months of the assessment period. Discharge from the tile drain appears to be a better indicator of soil moisture and capacity that does rainfall, and it appears likely that immediacy and extent of discharge response to rainfall is related to soil moisture conditions.

[1] https://systatsoftware.com/

Figure 4 provides a seasonal summary of inflow and outflow data. In all months, inflows and outflows are similar, and the small difference is related to measurement accuracy. Flows reflect measured rainfall and ambient temperatures closely.

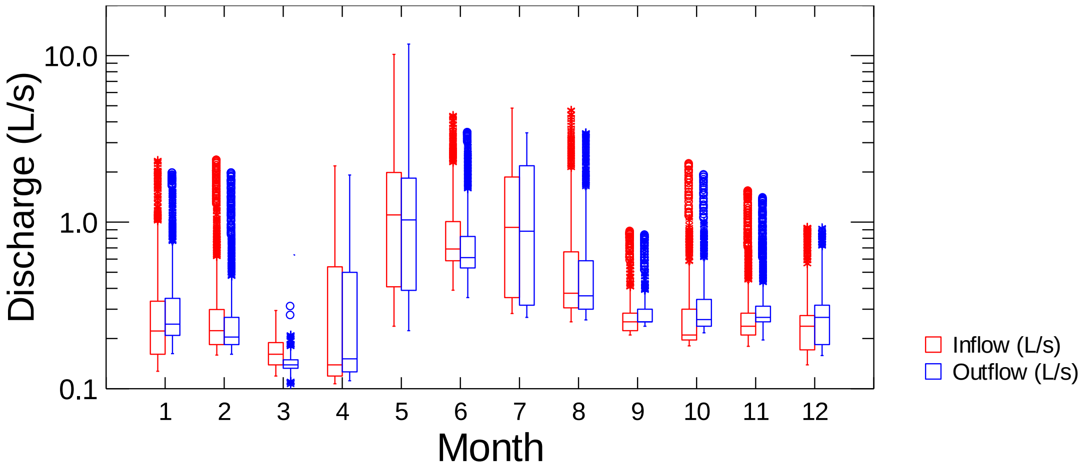

Figure 4: Seasonal summary of inflow and outflow. Incorporates data for the 15-month assessment period.

Figure 5 demonstrates that inflow nitrate-N concentrations were typically 2000 mg/m^3 during baseflow although they were higher during runoff events. Inflow concentrations were consistently higher in the inflow than the outflow. The outflow concentrations varied widely (more than two orders of magnitude), and varied seasonally (as well as under influence of other factors).

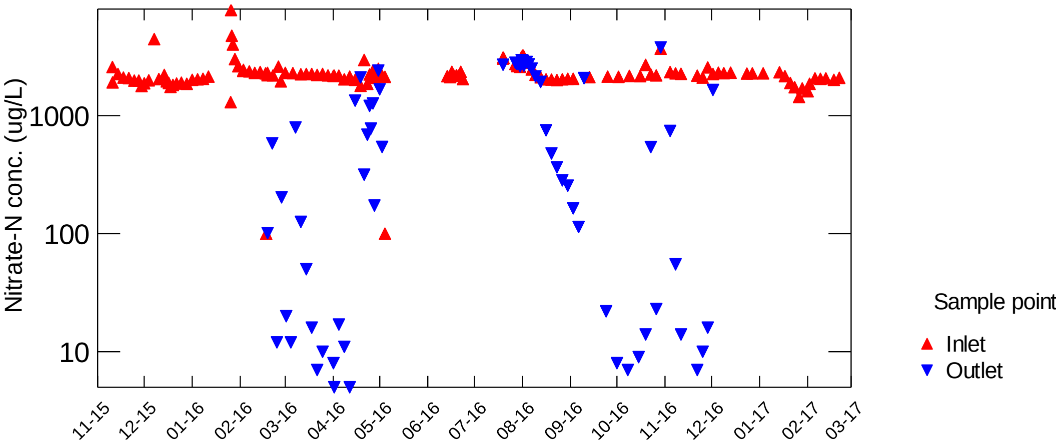

Figure 5: Time series of nitrate-N grab sample concentrations. Fewer samples were collected for the outlet following two failures of automatic samplers.

Figure 6 shows a time series of hourly inflow and outflow nitrate-N load estimates derived from modelling. The models provided good estimates of the inflow load, but the outflow load predictions were less consistent. Figure 6 indicates that the model over-predicts nitrate-N outflow loads in the late summer/autumn period (i.e., removal is likely to be under-predicted), whereas both models appear to represent peak nitrate-N loads adequately.

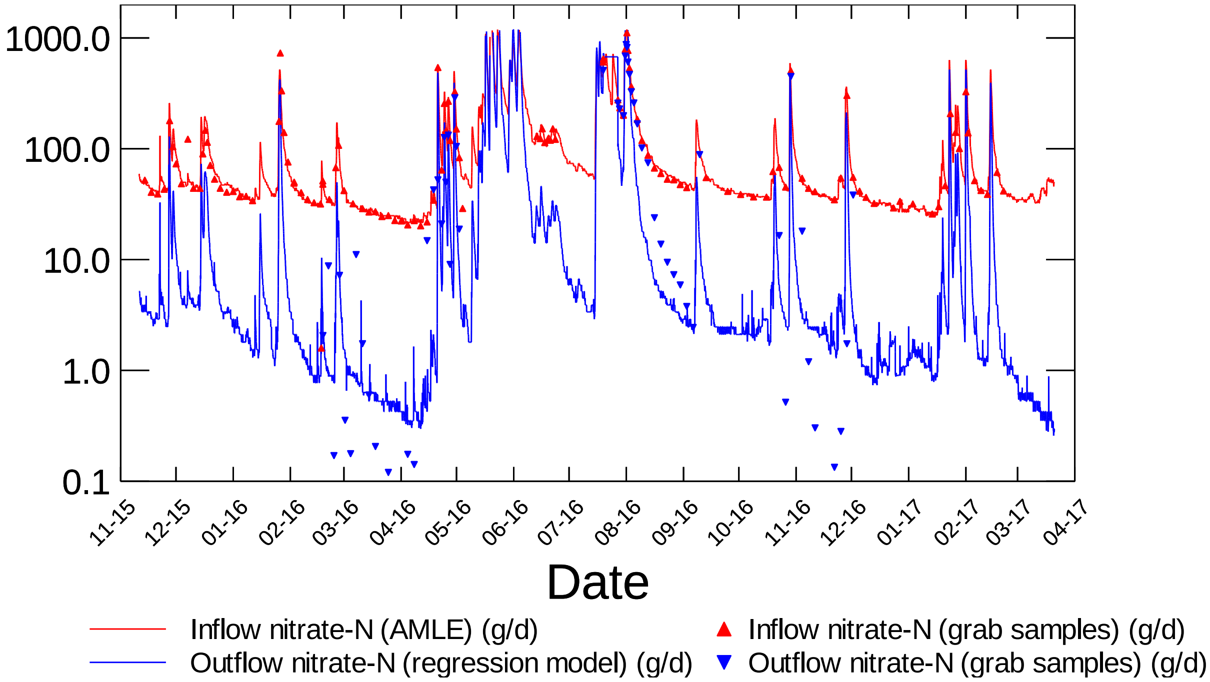

Figure 6: Time series of measured and predicted nitrate-N loads. "AMLE" refers to the specific regression model used (Runkel et al. 2004).

Nitrate-N removal efficacy is summarised in Figure 7. Except for periods of high inflow (i.e., over periods of short detention time), removal efficacy is generally high. Figure 7 indicates that median removal efficacy is approximately 94%, and efficacy exceeds approximately 90% removal up to the 70th percentile (viz., for 70% of time). The relationship between removal efficacy and retention time is apparent in Figure 7.

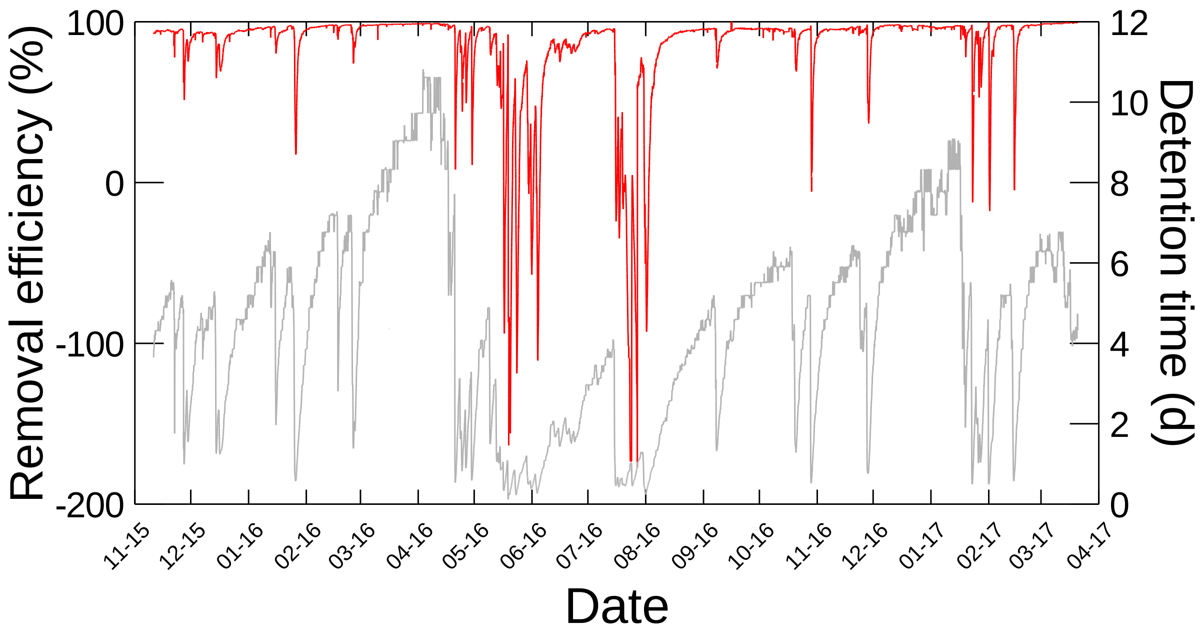

Figure 7: Time series of removal efficacy - nitrate-N loads. The red line indicates removal efficacy and the gray line indicates detention time within the filter bed.

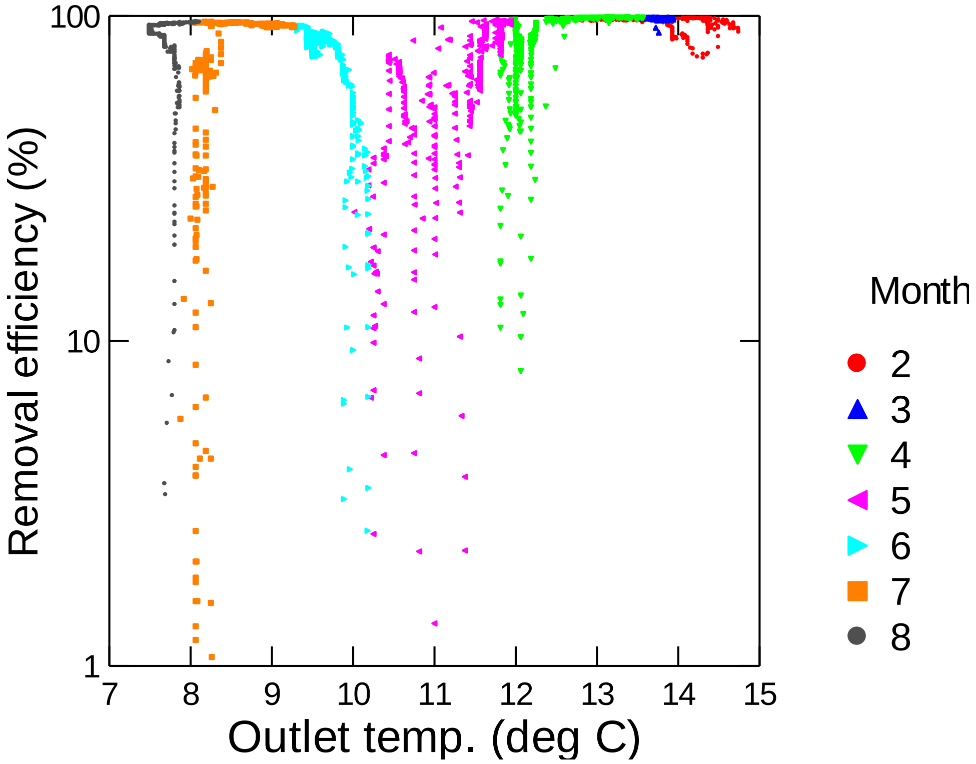

Figure 8: Relationship between removal efficacy and temperature for the N filter in selected months. Data for 2016 calendar year only.

In Figure 8, removal efficacy and outflow are plotted against temperature for selected months (representing the greatest annual temperature range). Figure 7 and Figure 8 suggest that removal efficacy is a function of flow (i.e. residence time), and temperature. In general, removal efficacy increases with temperature, and decreases as flow increases (i.e., as detention time decreases). In any month, nitrate removal is greater under low flow conditions (when residence times are longest).

The annual median concentrations of nitrate-nitrogen entering the filter (i.e. discharged from the tile drain), and discharged from the filter were 2190 µg/L and 421.5 µg/L, respectively. The median daily load was reduced from 47.1 g/d to 2.9 g/d (a reduction of 93.8 percent), and the median nitrate-nitrogen yield from the field was reduced from 1.83 kg/ha/yr to 0.11 kg/ha/yr.

Discussion and conclusions

Nitrate-N removal efficacy is compared with selected values from the literature in Table 1. Although the removal rates are low, they are within the ranges reported in several field trial studies, and may therefore be regarded as "typical".

Table 1: Comparison of selected reported nitrate-N removal rates with those observed in Waituna Lagoon catchment (this study)

Indicative nitrate-N removal rates ($g/m^3/d$)	Factor dominating treatment efficacy or performance	Reference
5 - 10	Nitrate-N load	Schipper et al. (2010)
16 – 6.4	Filter medium age	Robertson (2010)
6.4	Nitrate-N non-limiting	Woli et al. (2010)
23 – 44 1.2 - 11	Degradable carbon (medium age), ambient temperature	David et al. (2016)
7.6	Not identified	Warnecke et al. (2011)
0.38 – 1.06	Ambient temperature, hydraulic load	Christianson et al. (2013)
0 – 72	Ambient temperature	Hassanpour et al. (2017)
0.7 – 22	Not identified	Halaburka et al. ((2017))
0.7 ± 1.6	Hydraulic load, ambient temperature	This study (arithmetic average ± standard deviation)

Efficacy of the woodchip treatment filter was strongly dependent on hydraulic retention time, and to a lesser extent, temperature. These dependencies were evident as seasonally varying treatment efficacy. These factors should be considered when designing a woodchip filter, or before modifying an existing filter to enhance efficacy. For example, if seasonally large hydraulic loading determines treatment efficacy during autumn and winter, it may be possible to improve performance by buffering the inflow (e.g. by incorporating a buffer chamber into the filter design), or by temporarily retaining some of the drainage water in the landscape, within the drains themselves. This is done as part of a controlled drainage water and nutrient management strategy (e.g., Christianson et al. 2016). Another approach could include a bypass system that limits the inflow to the filter once a defined flow threshold is achieved. This functionality could be incorporated in a buffer chamber system, or by having two adjustable weirs on the inflow chamber. Adjusting the relative heights of these weirs would allow seasonal adjustment of the inflow. Several of these and other approaches may be combined to accommodate site-specific considerations (e.g. soil, gradient and farming system factors).

When designing nutrient management or mitigation tools of this nature, it is essential to have a "whole catchment" view of nutrient management, specifically the target nutrient load for the receiving environment. In New Zealand the National Policy Statement for Freshwater Management (NPS-FM) provides target attribute values intended to meet identified water quality objectives (MfE 2014). For example, numeric attribute state thresholds are defined for nitrate-N in terms of annual median and annual 95th percentile concentrations. These concentrations relate to nitrate-N toxicity and protection of various species. Protecting other ecosystem values may require more stringent concentration reductions. Most regulatory agencies in New Zealand are currently determining the nutrient concentrations required in their catchments to achieve appropriate water quality management objectives. These nutrient concentrations will determine the overall nutrient reduction required in the catchment, which in turn will determine the reduction in nutrient load required in each subcatchment, and ultimately, each farm. Once those targets are known, it is possible to estimate the number and location of mitigation tools that will be required to deliver the environmental outcomes desired.

This approach allows the specific resource management to be matched with the type, number and location of mitigation tools. The latter will include woodchip filters, as well as phosphorus filters, riparian planting and setbacks, reforestation and constructed wetlands.

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