NIWA
Taihoro Nukurangi

Waituna catchment - evaluation of nutrient mitigation options

Nitrogen and Phosphorus filters

Prepared for DairyNZ
Living Water - Department of Conservation - Fonterra Partnership
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Executive summary

Nutrient losses from agricultural lands is a consequence of intensification, which may require importation of feed, and use of fertilisers. A proportion of the nutrient in feed is deposited to land directly in animal wastes and as irrigated farm dairy effluent application. Although the bulk of nutrients in animal wastes and fertilisers are used to promote crop or fodder growth, the potential exists for excess nutrients, faecal bacteria and sediment to be lost from agricultural lands. Local communities and the farming industry is concerned with the potential for adverse effects on surface and groundwater. Several technologies have demonstrated potential for either retaining excess nutrients on farm, or for converting contaminants into less potentially harmful materials.

Tile drains are used extensively on poorly drained soil types to maximise the agricultural potential of farmland. Tile drains also effectively create “conduits” that accelerate delivery of excess nutrients from paddock to streams. This nutrient is also likely to be present in dissolved form, able to exert an immediate biological effect. Their widespread use and their ability to directly transfer nutrients (largely in bioavailable form) from land to water makes tile drains prime candidates for nutrient mitigation.

NIWA was commissioned by DairyNZ and the Department of Conservation-Fonterra “Living Water” initiative to design and install two nutrient mitigation filters – an N-filter and a P-filter, to assess their potential to remove nutrients from tile drainage discharge. The filter devices are installed on farms in the Waituna Lagoon catchment, Southland Region, where nutrient removal is being measured on typical or representative farming systems. The Waituna Lagoon catchment was selected for several reasons:

- other water quality improvement work is under way by regional council, DairyNZ and the Living Water initiative
- Waituna Lagoon has great cultural and community value, and is manifesting symptoms of environmental degradation (including water quality problems related to increasing nitrogen and phosphorus concentrations)
- tile drainage is widely used across the catchment, and is representative of other areas where tile drainage is used
- significant P loss occurs from soils in the catchment with low anion storage capacity
- soils in the catchment are typical of New Zealand soils with regard to nitrate-N loss.

The P filter bed is a 1 m^3 above-ground polythene crate containing Aqual P, a modified zeolite. The zeolite is a carrier for aluminium (the modifier), which is the P binding agent. Aqual P does not cause acidification and therefore does not require buffering. A metered flow from the tile drainage system of a cultivated paddock (approximately 1 ha in extent) was delivered to the filter bed using a pump. The inflow rate, total drainage volume and the outflow rate from the filter bed was measured continuously. Automatic samplers were used to collect water from the inflow to, and outflow from, the filter bed. Mercuric chloride preserved samples were analysed at the NIWA Hamilton Water Quality Laboratory routinely to determine dissolved reactive phosphate (DRP) and total phosphorus (TP) concentrations.

P removal efficacy was assessed by comparing the inflow and outflow concentrations of DRP and TP. The drainage flow ceased for some time when soil moisture was low. The small inflow to the filter bed was difficult to measure continuously because the inlet filter to the pump was prone to partial blockage by fine suspended sediment transported in the tile drainage. Consequently, phosphorus removal efficacy was estimated from the concentration data only, rather than from the flux or load data. This approach was justified because the inflow closely matched the outflow.

Phosphorus removal efficacy is summarised in Table i.

Table i: Summary statistics for inflow and outflow dissolved and total phosphorus concentrations and removal efficacies.

Statistic	Dissolved reactive phosphate (DRP)			Total phosphorus (TP)		
	Inlet	Outlet	Efficacy (%)	Inlet	Outlet	Efficacy (%)
Median concentration (µg/L)	88	0.5	99.4	195	42	78.5
95th percentile concentration (µg/L)	338	6.91	98	883	112	87.3

The N filter bed is a 10 m × 10 m × 1 m deep membrane-lined pit, filled with wood chip. An inlet sump receives field tile drainage from a 9.4 ha grazing paddock. The inlet flow was measured at five minute intervals, together with water temperature, electrical conductivity and turbidity. A manifold distributes the drainage across the width of the filter bed. The outflow is directed to an outlet sump, where temperature and water level was measured, as well as the outlet flow. An automatic sampler collected samples of the inflow and outflow under baseflow conditions, and in response to rainfall events. Mercuric chloride preserved samples were analysed at the NIWA Hamilton Water Quality Laboratory to determine ammoniacal-N, nitrate-N and total N concentrations. Electrical conductivity was also determined in selected samples.

Several methods were used to estimate the instantaneous mass load (flux) of the three forms of nitrogen entering and leaving the filter bed, and the resulting nutrient removal efficacies compared. Performance data derived from a 15-month assessment period is summarised in Table ii. Concentration data reflect grab samples collected over this period and the median and 95th percentile load values are derived from regression models.

Table ii: Summary statistics for inflow and outflow nitrate-N concentrations and loads, and nitrate-N load removal efficacy.

Statistic	Ammoniacal-N			Nitrate-N			Total N		
	Inlet	Outlet	Efficacy (%)	Inlet	Outlet	Efficacy (%)	Inlet	Outlet	Efficacy (%)
Median concentration (µg/L)	36	77.5	-114	2190	421.5	80.7	2495	1105	55.7
Median load (g/d)	0.59	1.838	-210.5	47.1	2.90	93.8	55.6	28.5	48.7
95th percentile load (g/d)	1.36	3.45	-154.2	460.1	438.6	4.7	497.5	400.1	19.6

Similar nutrient removal efficacy was determined from the instantaneous loads or flux obtained as the product of grab sample concentrations and flow at the time of sampling. Instantaneous median and 95th percentile removal efficacies are summarised in Table iii:

Table iii: Nitrogen removal efficacy for three forms of nitrogen.

Nitrogen species	Nutrient removal efficacy	
	Median concentration	Ninety-fifth percentile concentration
Ammoniacal-N	-182	-492
Nitrate-N	79.6	99.7
Total-N	54.7	77.3

The anaerobic conditions in the filter bed required for conversion of nitrate-N to nitrogen gas also favoured the formation of ammoniacal-N, causing the filter to be a net source of ammoniacal N throughout the assessment period. As Table ii indicates, however, ammoniacal-N concentrations were low in comparison to inflow nitrate concentrations. Deployment of nitrogen filter beds (nitrate-N scrubbers) should have regard for the potential for these beds to form ammoniacal-N, which is toxic to aquatic organisms at relatively high concentrations and under certain pH conditions.

The consistent performance of the filters for nitrate removal indicates that this simple, low maintenance technology has potential to provide water quality benefits when used in the right circumstances. Additional information requirements include:

- The ultimate life expectancy of the filter bed materials.
- The level of inspection and management of these systems required to maintain nutrient removal performance.
- Long-term operating and management costs.
- More accurate assessment of the potential for ammonia toxicity in receiving waters.

Integration of these filters into existing farming systems requires special consideration because of the complex interaction between farm management, local soil and drainage conditions, the likely hydraulic load, the nutrient load in the drainage water and the existence of other mitigation tools. Accordingly, although it is not possible to provide detailed design and build specifications, several guiding principles should be considered at local catchment scale, as well as at farm scale:

- How many devices would be required to make a measurable difference to the load of material entering surface waters?
- What volume of water will the catchment/tile drain network deliver to the treatment filter?
- How will seasonal/event-scale hydraulic variability be addressed in the design?
- What will it cost to build and maintain the number of filters required?

In this report, treatment efficacy (nutrient-removal performance) of these systems is described in as much detail and accuracy as available data allows. It identifies that hydraulic loading rate is a key determinant of nitrate-N removal. Several options are identified to manage hydraulic loads and optimise performance. The performance data will also allow the number of treatment units required to deliver water quality improvements to be identified.

Earlier work done by NIWA provides budgetary estimates of cost for each of these units. Combining this information while considering local factors will allow farmers and industry groups to design systems that will enable them to reduce farm and catchment nutrient losses.

1 Introduction

The input of excess nutrients (specifically nitrogen and phosphorus, N and P) to surface waters has the potential to encourage proliferation of excessive nuisance plant growth, result in ammonium and/or nitrate toxicity and impair water quality and ecosystem health. To avoid these and other adverse outcomes, the National Policy Statement for Freshwater Management (NPS-FM) requires water quality is to be managed to achieve certain values (MfE 2014). When describing the health and mauri of water (one of the compulsory national values), a healthy ecosystem has characteristics such as:

- ability to supporting a healthy ecosystem appropriate to that freshwater body type (river, lake, wetland, or aquifer)
- ability to maintain ecological processes, which in turn support a range and diversity of indigenous flora and fauna, and where resilience to change exists, and where
- adverse effects on flora and fauna of contaminants, changes in freshwater chemistry, excessive nutrients, algal blooms, high sediment levels, high temperatures, low oxygen, invasive species, and changes in flow regime are minimised through management actions.

To improve water quality outcomes, inputs of nutrient from various land uses may need to be managed. Various on-farm tools are available to reduce the N and P in water leaving paddocks and entering streams, including constructed and natural wetlands, riparian buffers, or addition of reactive materials (McKergow et al. 2008).

While the science behind many attenuation tools is reasonably well understood, their performance at the field-scale is variable and poorly quantified. Farmers require certainty regarding the efficacy and cost-effectiveness of different mitigation options, before they will adopt new strategies for reducing contaminant losses. Industry bodies (e.g. Dairy NZ, Fonterra), regulatory agencies (Department of Conservation, Regional Councils etc), and researchers, consultants and farm advisers all need to have confidence in the mitigation measures being deployed. Field trials are required under conditions relevant to New Zealand pastoral farming to verify performance, refine design, demonstrate applicability and provide realistic information regarding construction and maintenance costs.

On-going research in the Waituna catchment aims to provide cost-effective and practical solutions for farmers to enable them to reduce their environmental footprint and contribute towards the long-term management of the Waituna Lagoon. A scientific workshop led by DairyNZ and attended by 14 scientists from NIWA, AgResearch, DoC, Environment Southland and DairyNZ in October 2013 identified denitrification and phosphorus sorption filters as having significant potential to reduce nutrient loading to Waituna Lagoon, alongside other on-farm nutrient management tools. Tile drains are an important feature of Southland’s agricultural landscape, providing drainage essential for pasture production. Subsurface drainage reduces surface runoff, peak outflow rates and Improved drainage accelerates the transport of nutrients off-farm, particularly nitrate-nitrogen. This form of nitrogen is readily mobilised through the soil profile with drainage water. Tile drainage effectively shortens nutrient discharge pathways (reducing denitrification capacity), thereby increasing the delivery of nitrogen to surface waters (Maalim and Melesse 2013; Christianson et al. 2016; Villeneuve 2017). Arenas Amado et al (2017) demonstrated that tiles delivered up to 80% of the stream N load while providing only 15–43% of the streamflow. The use of treatment systems to intercept and treat tile drain discharges would have wide scale applicability if it could be demonstrated that this was a cost-effective and practical mitigation tool.

Nutrient attenuation or removal can be enhanced by the addition of reactive materials to flowpaths, such as tile drains. Materials are added to target one nutrient attenuation process, typically the addition of carbon for N removal by denitrification, or addition of reactive materials to facilitate P removal by adsorption.

Adsorption is the physical or chemical binding of molecules to the surface of solids (soil, sand, clay, pumice, limestone, shells, and modified materials such as aluminised clays). A wide range of materials are available, but any material selected should have a moderate to high affinity for P, be relatively abundant, be readily available at low cost, be non-toxic, be suitable for reuse with no risk to soil or water quality in either the short or long term, and ideally be a renewable and natural material (Ballantine and Tanner 2010). Melter slag, fly ash and alum have been through basic 'proof of concept' testing, but field scale performance assessments are required.

Denitrification is the conversion of simple organic carbon and an electron acceptor (such as nitrate), to energy, carbon dioxide and gaseous oxides (nitric oxide (NO) and nitrous oxide (N_2O)) or nitrogen gas (N_2) (Christianson 2011). A diverse range of microorganisms (bacteria, proteobacteria, archaea and fungi) are capable of denitrification. Optimal denitrification conditions for these specialist microbes include:

1. A slow release carbon source.
2. Nitrate source.
3. Anoxic (low oxygen) conditions.

Passive filter systems have been extensively trialled at laboratory- and mesocosm-scales around the world. Recently, larger-scale trials have been initiated in the US for treatment of diffuse agricultural run-off and drainage from cropped lands, and preliminary implementation guidelines have been developed (Christianson et al. 2012a; Christianson et al. 2012b). Although performance is promising, it is expected to be highly dependent on the seasonality and variability of drainage flows. To date these systems have not been applied to treat agricultural tile drain runoff in New Zealand.

Denitrification walls and small-scale woodchip filters have been evaluated under New Zealand conditions. Denitrification walls (trench filled with sawdust and soil mix) are best constructed where the full extent and flow direction of nitrate-polluted groundwater (including shallow sub-surface drainage) can be determined, such as sites used for intensive land application of wastewater, cattle feedlots, and old fertiliser dumps (e.g., Schipper and Vojvodic-Vukovic 1998).

Small-scale woodchip filters have been evaluated in the Waikato (Sukias et al. 2005; Sukias et al. 2006). Three medium (1.2% of catchment area) and one small (0.6% of catchment area) pilot-scale woodchip filters receiving tile drainflow on a dairy farm in the Waikato were monitored. Annual mass loads of nitrate-N were reduced by 55-79% for over a two-year period, representing average annual removal rates in the range of ~0.09-0.3 g $N/m^3/d$. Increases in levels of ammonium-N and, in the first year of operation, organic-N, reduced the efficacy of total N removal (16-49%).

Higher denitrification rates (in the range of 2-10 g $N/m^3/d$) have been recorded in other field-scale trials under continuous flow where nitrate concentrations are non-limiting (Schipper et al. 2010).

NIWA was commissioned by DairyNZ and the Living Water - Department of Conservation - Fonterra Partnership to design, install and operate a woodchip-filled nitrate-N filter, and a smaller phosphorus filter. The latter was filled with a modified zeolite medium – the modification involves inclusion of aluminium as the primary phosphorus binding agent. Once the two filters were installed and operational, NIWA was to manage/operate a monitoring programme that would provide the data and information required to estimate the N and P removal efficacy of these filters. The selection of the filter deployment sites, and design and construction of these filters was described previously (Tanner et al. 2013; McKergow et al. 2015; McKergow et al. 2016). This report summarises the outcomes of the monitoring conducted over the previous 10- and 16-month periods for the P and N filters respectively.

2 Materials and methods

The location of sites that could be used as candidates for installation of nutrient mitigation actions such as constructed wetlands was summarised by Tanner et al. (2013). One of the sites identified (Site C, on the Pirie property) was identified as suitable for construction of the N filter. The characteristics that made it attractive for installation of the N filter included:

- adequate grade, which made a gravity-fed system possible
- a clearly defined catchment area
- soils and lithology that made the farming system susceptible to N loss through the root zone, and
- existence of tile drainage that could potentially be intercepted and directed to a filter bed.

The P filter site was selected following assessment of the soil anion storage capacity. These soils tend to be dominated by peat, which largely determines the susceptibility to P loss. The gradient over the candidate site did not permit a gravity fed system, and it was decided that a pumped system would be installed on the Foveaux Investment property, in the Currans Creek catchment. The location of the two filters is indicated in Figure 2-1.

Figure 2-1: Location of the N and P filters, Waituna Lagoon catchment.

McKergow et al. (2015, 2016) identified key factors to consider when selecting sites for placement of N and or P filters. These reports also summarised the design, materials used and methods used to construct the N and P filters, and the reader is referred to them for further details.

2.1 Water sample collection and analysis

Assessing the performance of the N and P filters was one of the primary objectives of this project, and is the focus of this report. Automatic, high frequency measurements (5-minute intervals) of a range of variables were made and used to determine the performance of the filters. Appendix A has a schematic of the P filter (Figure A-1) and N filter (Figure A-2), and along with a list of measured variables. These schematics indicate that automatic samplers were deployed at each site, on the inflow to and the outflow from each treatment device. Grab water samples were collected on a fixed time interval (baseflow conditions) or flow volume basis (during rainfall events). These samples were stored in the automatic sampler in mercuric chloride-preserved bottles, to minimise biological transformations of the nutrients. Preservation was required because samples were submitted to the NIWA Hamilton water quality laboratory for analysis only when each carousel of 24 bottles was filled. The methods of analysis and method detection limits are summarised in Table B-1 (Appendix B).

2.2 Data processing and analysis

Data collected on site (summarised in Appendix A) were transmitted from the NEON loggers deployed at each site to a server in located at NIWA Christchurch. Data could be retrieved from this server at any time, and were used to ensure that systems were operating correctly and to determine the number of sample bottles that were filled. These data were retrieved in as-recorded state (five-minute intervals), and manipulated within Microsoft Excel as required. Operations included:

- collating the grab sample data with the flow measurements at the correct time (using the VLOOKUP function)
- generating additional date and time fields
- transforming flow values into non-metric units for input to modelling packages
- calculating average daily flow values (pivot tables)
- selecting data using native filter tools of Excel (data filter and pivot tables)
- calculating instantaneous loads (flux) for the various species of N and P
- estimating continuous nutrient flux using
  - simple linear regression modelling
  - more advanced regression modelling utilising a bootstrapping technique.[1]

Exploratory data analysis, and generation of figures and summary statistics was undertaken using Systat v13.[2] Systat was also used to calculate removal efficiencies and nutrient fluxes as required.

The LOADEST[3] modelling system was used to estimate nutrient flux in the inflows and outflows from the N filter. The United States Geological Services describes the modelling system as:

> *"LOAD ESTimator (LOADEST) is a FORTRAN program for estimating constituent loads in streams and rivers. Given a time series of streamflow, additional data variables, and constituent concentration, LOADEST assists the user in developing a regression model for the estimation of constituent load (calibration). Explanatory variables within the regression model include various functions of streamflow, decimal time, and additional user-specified data variables. The formulated regression model then is used to estimate loads over a user-specified time interval (estimation). Mean load estimates, standard errors, and 95 percent confidence intervals are developed on a monthly and(or) seasonal basis.*

[1] Developed by Dr Kit Rutherford, Emeritus Scientist, NIWA Hamilton
[2] https://systatsoftware.com/
[3] https://water.usgs.gov/software/loadest/

The calibration and estimation procedures within LOADEST are based on three statistical estimation methods. The first two methods, Adjusted Maximum Likelihood Estimation (AMLE) and Maximum Likelihood Estimation (MLE), are appropriate when the calibration model errors (residuals) are normally distributed. Of the two, AMLE is the method of choice when the calibration data set (time series of streamflow, additional data variables, and concentration) contains censored data. The third method, Least Absolute Deviation (LAD), is an alternative to maximum likelihood estimation when the residuals are not normally distributed. LOADEST output includes diagnostic tests and warnings to assist the user in determining the appropriate estimation method and in interpreting the estimated loads."

Several techniques were applied to the inflow and outflow measurements to estimate the fluxes of nutrients. This was necessary to identify a suitable model able to predict the nutrient flux in the outflow from the N filter, given the biological removal of nitrate-N and conversion to other forms (principally the gases N_2 and N_2O). The workflow associated with data collation, processing and analysis is summarised in Figure 2-2.

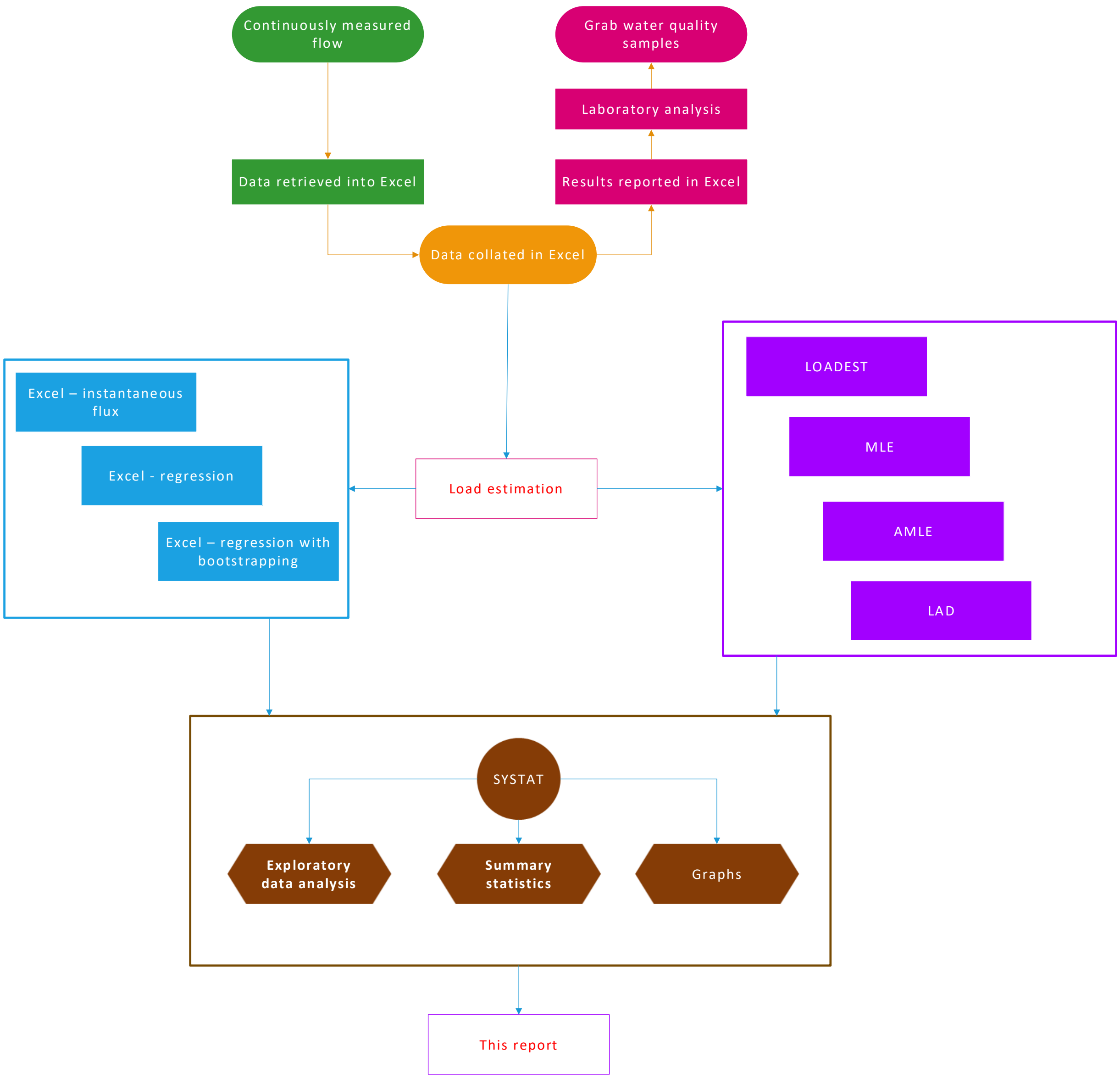


Figure 2-2: Data collation, processing and analysis sequence.

2.3 Load and removal efficacy estimation

Removal efficacy may be expressed in several ways, for example:

- as mass removed over time
- as mass removed proportional to treatment area or volume
- as mass removed proportional to catchment or source area.

A common requirement is an estimation of the mass load entering and leaving the treatment module in a unit of time. The mass of contaminant entering or leaving the treatment module is the product of concentration (units mass/volume) and flow rate (units volume/time). Often flow is measured continuously but concentration is only measured occasionally.

The simplest estimate of mass entering or leaving a treatment module is the product of grab sample concentration and the flow at the time of sampling. This will, however, provide information only at the time each grab sample is collected. No information regarding the times before or after sampling can be deduced even though flow is known.

This limitation may be overcome by applying suitable modelling techniques, such as regression models relating concentration to flow. These models allow the relationship between a continuously measured variable (in this instance flow) and an infrequently measured variable (contaminant concentration) to be established. This relationship may be used to estimate or predict the contaminant load during periods when no concentration data exist. The accuracy of these estimations depends on several factors – these are considered after the modelling and several statistical measures may be used to determine the suitability of the model (and therefore of the usefulness of the load estimations).

For the N filter, it was relatively easy to determine the inflow load – several regression techniques were found to be suitable, demonstrating a close relationship between instantaneous load estimated using grab samples and model predictions, and able to account for seasonal and associated flow and temperature effects. Estimation of the outflow load was more difficult however – the regression models (viz., the relationships between concentration and flow) were unable to account for the biological treatment process, which was not as closely related to flow. Consequently, all the regression techniques investigated were found to underestimate the nitrate-N removal rate. Attempts to improve the model fit by incorporating additional terms for which data were available (such as temperature, electrical conductivity and turbidity) failed to improve the model performance appreciably. Improvement of model fit are likely to require inclusion of data for additional water quality variables, such as organic carbon concentrations, which were not available for this project.

Despite this limitation, the modelling provides mass loads that may be used to estimate treatment efficacy, and allows the seasonal performance of the filters to be determined. Accordingly, these estimates may be regarded as fit for purpose.

For the P filter, a fraction of the water draining from the catchment paddocks was directed through the filter bed. This fraction varied dynamically as a function of total flow, with the flow entering the filter bed always less than a threshold defined by the pump rate. In this circumstance, the flow entering the unit would be the same as that leaving it. Estimation of treatment efficacy was simpler and was limited to a comparison between inflow and outflow concentrations. Consideration was also given to the effect of additional variables, such as temperature and turbidity.

3 Results

3.1 P filter performance

3.1.1 The climate context

Rainfall depths recorded at the site are compared with long-term precipitation records derived from the Invercargill Airport Automatic weather station in Figure 3-1. 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.

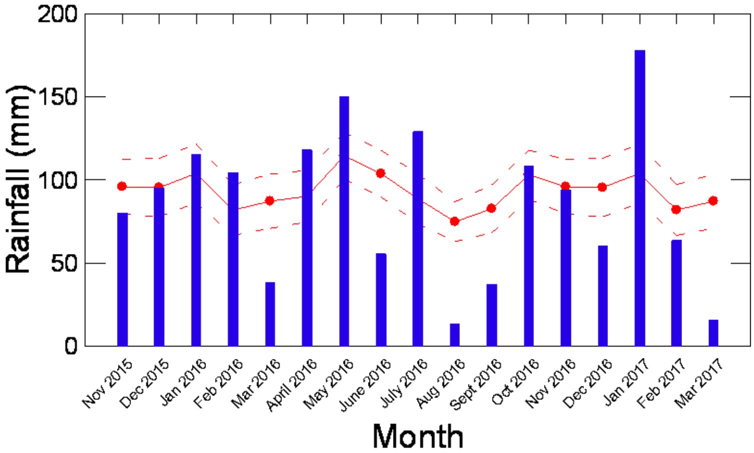


Figure 3-1: Comparison of on-site rainfall (blue bar) with long-term monthly average rainfall (red dot). The long-term average rainfall is from the Invercargill Automatic Weather Station site and represents the period January 1990 – December 2017. The upper and lower broken lines indicate the lower and upper 95th confidence interval of the long term monthly average value.

Rainfall were recorded at the P filter site. Monthly total rainfall data are summarised in Figure 3-2. The greatest average flow was recorded in May 2016. The reason for this is the high rainfall that had been measured in preceding months (data recorded at the N filter site (Figure 3-8) indicate that more than 100 mm of rain had fallen in April 2016). The earlier rain had presumably brought the soil to near field capacity, and additional rain in May 2016 resulted in considerable discharge to shallow groundwater and the tile drains. The bypass flow provides a good indication of the total discharge from the catchment. Figure 3-8 also indicates that the greatest inflow to the N filter occurred in May 2016. Entry of soil water into the tile drains declined over the winter period, despite at least some rain each month. As the soil profile dried, the tile drain response to discharge became weaker. For example, rainfall of approximately 100 mm in July 2016 increased the monthly average bypass flow from 0.3 to 0.4 L/s – rainfall of similar magnitude in October 2016 and November 2016 increased discharge by a smaller amount. Considerable rain fell in January 2017, which increased average tile drain discharge from almost nothing to more than 0.5 L/s

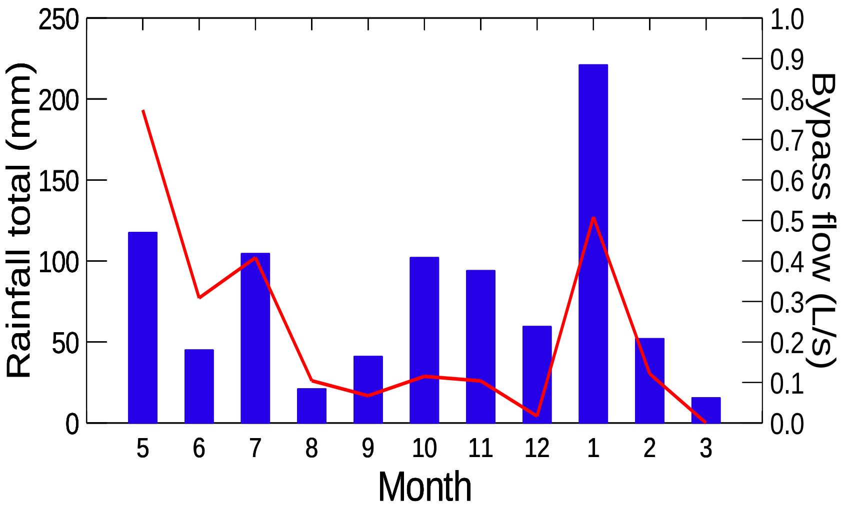


Figure 3-2: Monthly total rainfall and average monthly flow recorded at the P filter site. The period extends from May 2016 to March 2017.

Figure 3-3 shows monthly average flow, temperature and turbidity values. A seasonal increase in turbidity is evident in spring - the cause cannot be explained at this time, but it is potentially related to biomass growth in the tile drain system being flushed out by rainfall in October.

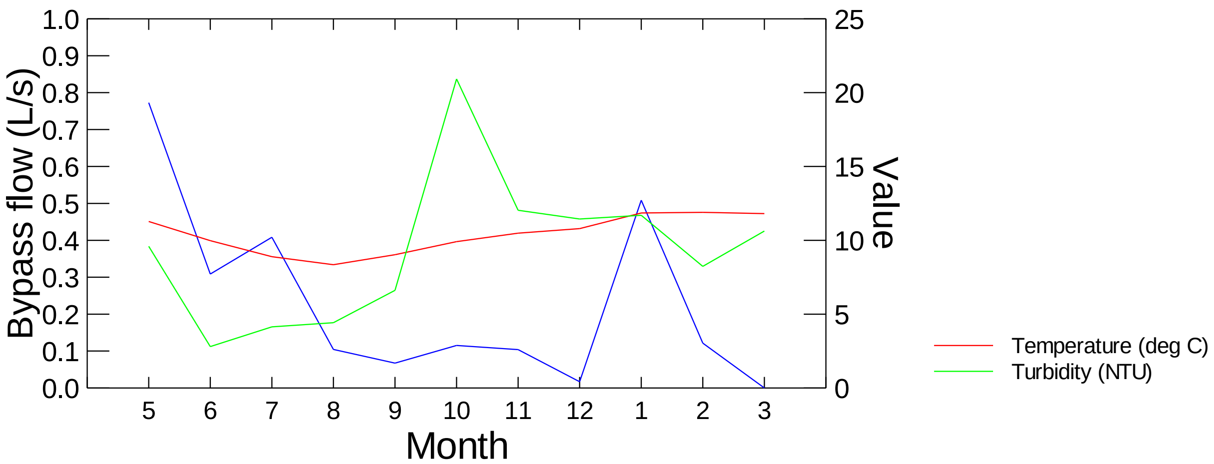


Figure 3-3: Monthly average flow, temperature and turbidity recorded at the P filter site. The period extends from May 2016 to March 2017.

3.1.2 P-filter inflow and outflow phosphorus concentrations

Inflow and outflow grab sample soluble and total phosphorus concentrations are summarised in Figure 3-4. Summary statistics are provided in Table C-1. Several features are common to both DRP and TP:

- inflow concentrations of DRP and TP are consistently greater and more variable than outflow concentrations
- both DRP and TP inflow concentrations increased markedly during spring and early summer.

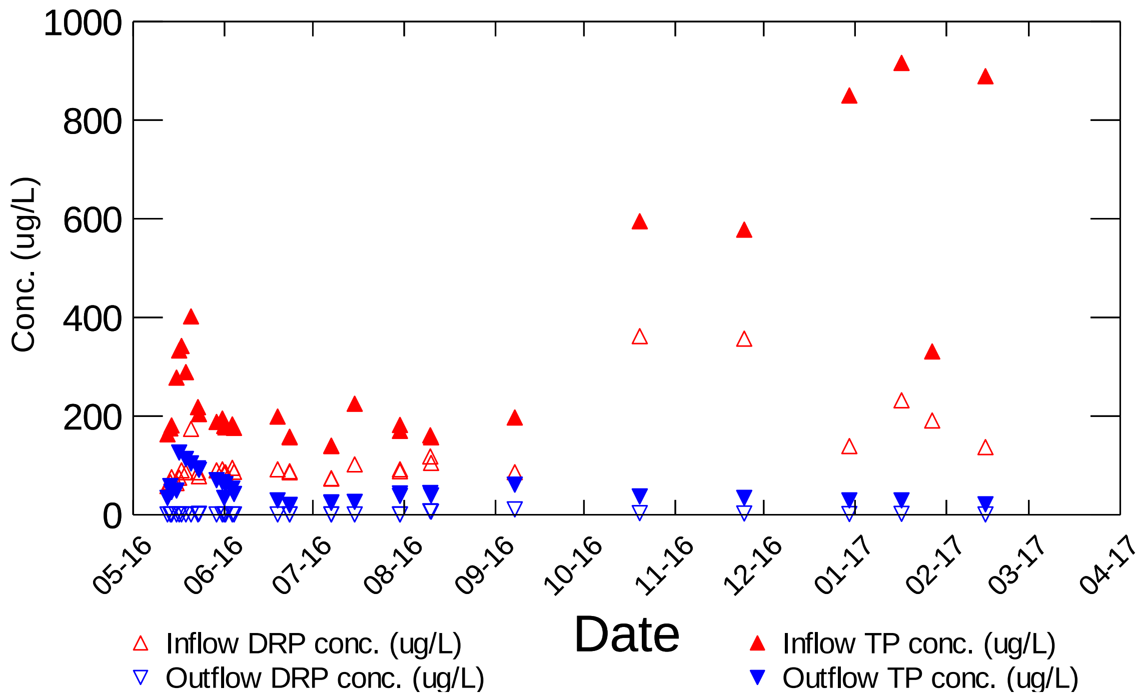


Figure 3-4: Grab sample inflow and outflow TP and DRP concentrations. The assessment period extends from May 2016 to March 2017.

Figure 3-5 indicates the relative proportion of DRP in TP in the inflow and outflow from the filter. Under most conditions, DRP comprised more than 40% of TP in the inflow, whereas in the outflow, the DRP component of TP was less than 20%. This demonstrates that the P- filter measurably reduced the proportion of readily available phosphorus from tile drainage.

Removal may also be accurately estimated in terms of mass removal rates, calculated as the product of concentration and flow. For the P filter, the inflow and outflow discharge rates were the same. Figure 3-6 presents DRP and TP removal efficiencies (viz., (inflow minus outflow)/inflow as mass fluxes) over the eleven-month assessment period, and under the tile drainage conditions that occurred during this time. Key points to note:

- Measured DRP removal efficiencies exceeded 95% except in September-October 2016.
- Measured TP removal efficiencies were far more variable, particularly early in the assessment period.
  - From the data available, it is not possible to determine whether this was because the P filter was in a "start-up phase" (which coincided with lower temperature conditions), persistently higher soil drainage conditions, or the form of phosphorus that entered the filter was less amenable to adsorption to the medium.
- The decline in DRP and TP removal efficiencies in September-October 2016 coincided with the increase in turbidity evident in Figure 3-3, but it is not possible to demonstrate that they are directly related.
- With the onset of drier conditions, and lower soil drainage rates over the summer, both DRP and TP removal efficiencies were consistently high. It is not possible to say

whether this was because of lower loading rates, or because the proportion of DRP in the drainage had increased.

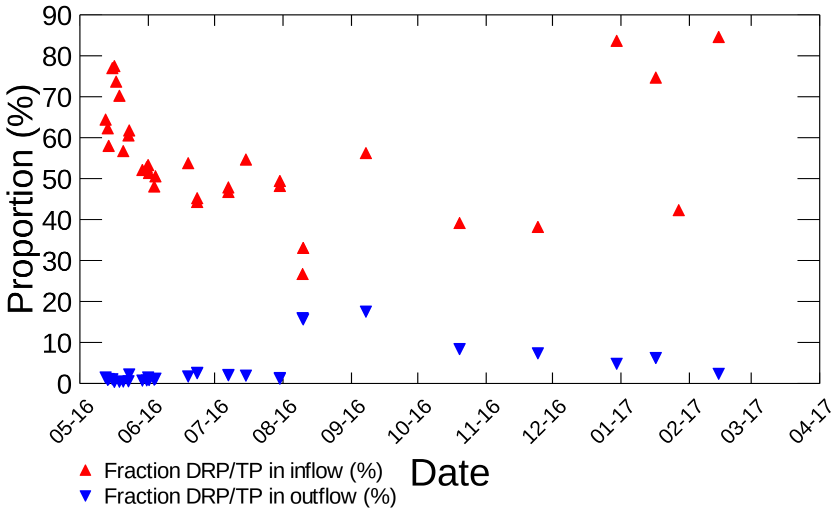


Figure 3-5: Proportion of DRP in inflow and outflow TP. The assessment period extends from May 2016 to March 2017.

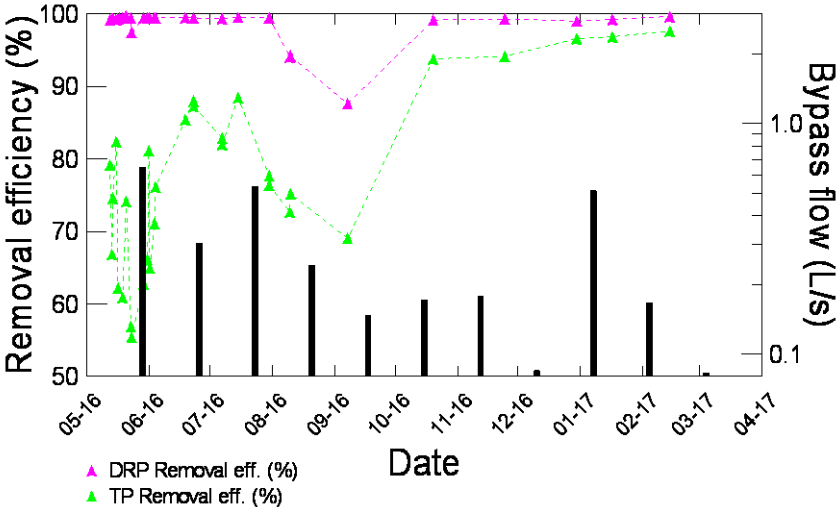


Figure 3-6: Phosphorus removal efficacy over time and under different tile drainage conditions. Bypass flow shown as monthly average values.

The inflow and outflow DRP and TP concentration data and monthly average bypass flow values are compared in Figure C-1 and Figure C-2, and removal efficacy in terms of DRP and TP concentration percentile values in Figure C-3.

3.1.3 P filter performance - discussion

It is not meaningful to estimate the performance of the filters in terms of catchment area because only a fraction of drainage from the catchment was pumped into the filer bed, and because of impairment in pump performance (ultimately leading to equipment failure), the proportion of total drainage treated in the filter could not be estimated accurately. The small volumes of flow that occurred when the discharge from the tile drain system declined in summer makes estimation of the mass load and mass removal rates during this period difficult. Despite these limitations, however, these data indicate that the P filter removed the bulk of the DRP and a substantial proportion of the TP load that was pumped into it.

The Aqual-P medium reduced P concentrations very effectively at the concentrations and loading rates to which the P filter was subject. Given that the concentration of DRP increased in the summer, when tile drainage is likely to be lowest, the potential exists to increase the hydraulic loading rate to better estimate the performance of the P filter, and to increase P removal from the site. This could be done by increasing the hydraulic loading rate generally (by increasing the pump sampling rate), or by targeting the summer period specifically.

Figure 3-3 indicated that the tile drainage contains a reasonably consistent load of suspended material (using turbidity as an index). It would be beneficial to characterise this material further to determine whether it is of organic or inorganic nature, whether the proportion of organic material alters seasonally, and whether the accumulation or particulate material at the bed surface constitutes a future load of soluble P that may be mobilised from the accumulated sediment should biogeochemical conditions in the bed alter. The effective life of the medium may be increased if a "potential P load" may be physically removed by strategically replacing the surface few centimetres of bed material. It would also be useful to determine the effect that this persistent load of particulate material has on permeability of the P filter.

Figure 3-5 indicates that DRP makes up a minor proportion of the TP load in the P filter outflow (generally less than 10%). This suggests that the primary P removal mechanism involves adsorption of soluble P to sites in the filter bed, rather than filtration of phosphorus-containing particulate material. It is possible that it may over time reduce permeability by clogging interstitial spaces, or reduce the number of active sites responsible for adsorbing the phosphorus at a significantly greater rate than the phosphorus in the inflow alone.

3.2 N filter performance

3.2.1 The climate context and impact on hydrology

Rainfall depths recorded at the site are compared with long-term precipitation records derived from the Invercargill Airport Automatic weather station in Figure 3-1. The impact of considerably lower or higher precipitation values (relative to long-term averages) on the discharge measured inflows are discussed below.

Filter inflow and outflow temperatures display a similar seasonal trend (Figure 3-7), suggesting that uniform conditions occurred across the wood chip filter bed. Total monthly rainfall exceeded 100 mm on nine of 15 months of the assessment period. Discharge appears to be a better indicator of soil moisture and capacity that does rainfall, and it is appears likely that immediacy and extent of discharge response to rainfall is related to soil moisture conditions.

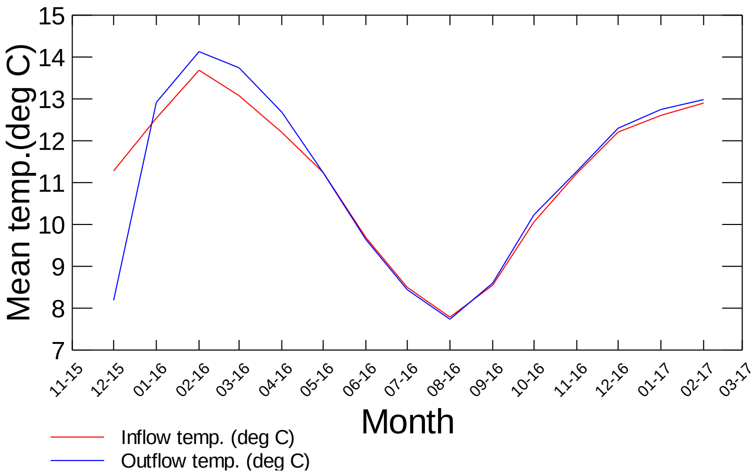


Figure 3-7: Seasonal variation in average inflow and outflow temperature. The monthly mean temperature is derived from data collected at five-minute frequency.

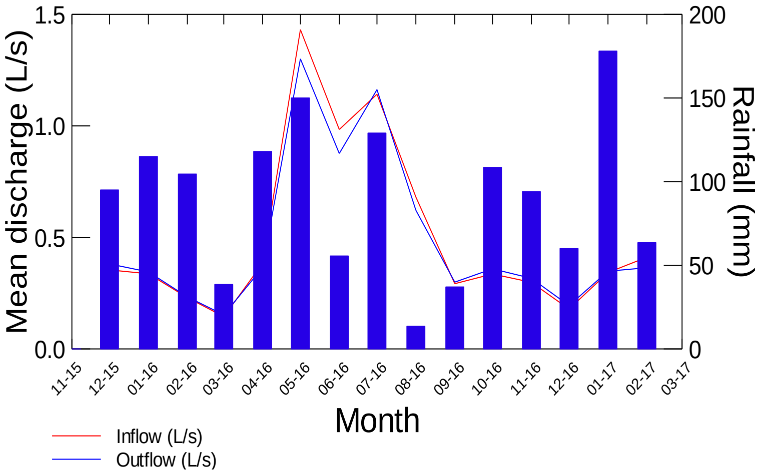


Figure 3-8: Monthly total rainfall depth and average N filter inflow and outflow.

Figure 3-9 indicates a complex relationship between turbidity and discharge during high rainfall event conditions, but the increase in turbidity observed during spring at the P filter site is less obvious. The range of turbidity observed and extensive periods of high turbidity values indicate that a measurable mass of particulate material may enter the filter. The data available does not allow this to be characterised – it would be useful to determine the nature of this material, because it could represent a source of organic carbon that may contribute to nitrate-N removal, either by encouraging anoxic conditions, or as a direct source of labile carbon.

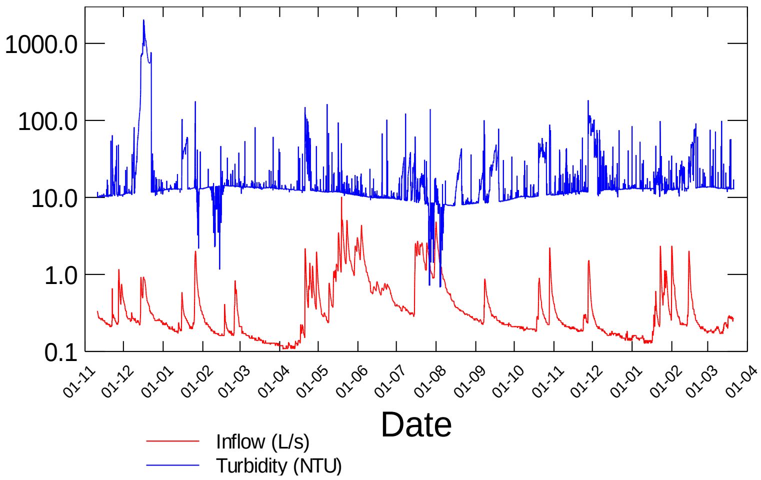


Figure 3-9: Time series of inflow and inflow turbidity.

As discussed in Section 3.1.1, although rainfall generally followed the long-term seasonal pattern, the well above- and below average rainfall was expected to influence tile drainage flow. Figure 3-10 provides a seasonal summary of inflow and outflow data. In all months, inflows and outflows are similar. From spring through early summer outflow is slightly larger than inflow – this could indicate a small amount of infiltration into the bed from shallow groundwater. In winter, it appears that water may leak from the filter bed to the surrounding soil until shallow groundwater levels rise in response to soil saturation. In general, inflows and outflows match rainfall, with some lag arising from less immediate changes in soil moisture.

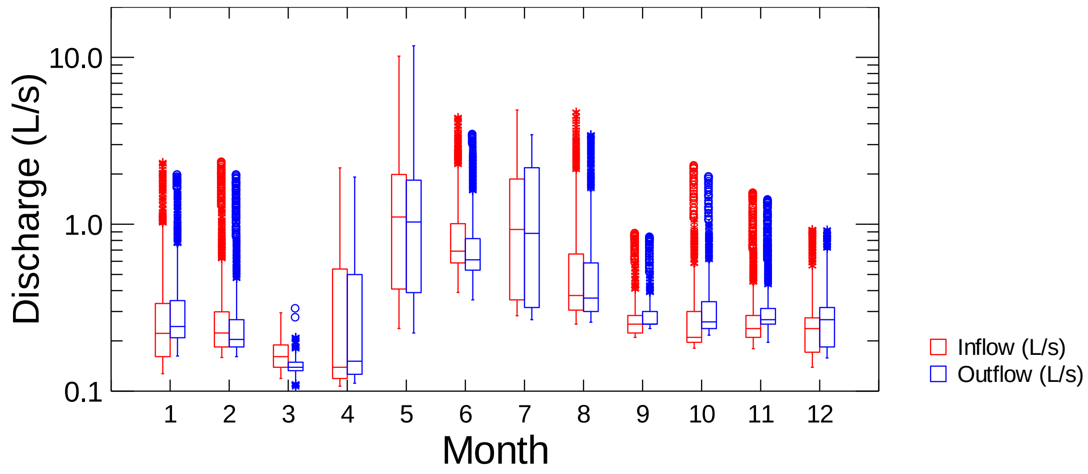


Figure 3-10: Seasonal summary of inflow and outflow.

Figure 3-11 A indicates the positive relationship between inflow electrical conductivity and inflow rate. In Figure 3-11 B, the relationship is clearer - daily average EC data for four months are used to represent seasonal differences.

- In summer, soil moisture is relatively low, soil drainage is low, and drainage has relatively low concentrations of dissolved salts and other ionic materials (indicated by low EC values).

- In early winter, rainfall causes soil drainage to increase and a larger range of flows occur from the tile drains.
- Electrical conductivity generally increases with flow, indicating increased mobilisation of solutes (likely to include nitrate-N and ammoniacal-N).

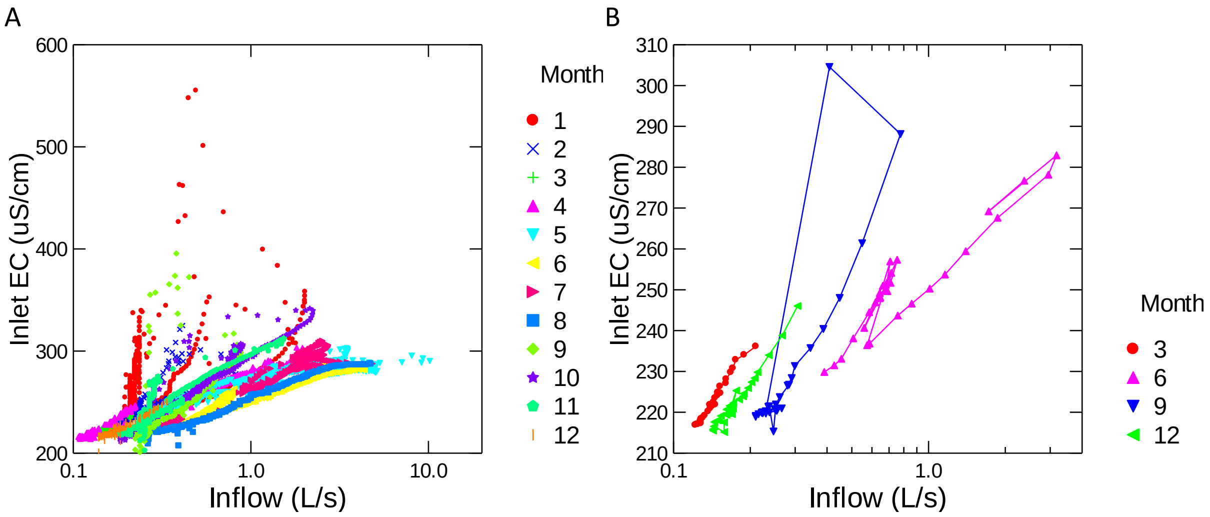


Figure 3-11: Relationship between inflow electrical conductivity and inflow discharge rate, 2016 calendar year. A – all months, hourly average EC, B seasonal representatives, daily average EC.

The relationship between turbidity and flow is more complex, as indicated in Figure 3-12A for 2016 using daily average values categorised by month, and seasonally in Figure 3-9 B. In spring and summer, a strong positive relationship between flow and turbidity is evident that is not as pronounced in autumn and winter. These data indicate that a greater proportion of the nutrient load may be in particulate nutrient forms in spring and autumn. No information exists regarding the amount of particulate material discharged from the filter bed – turbidity was not measured in the outflow.

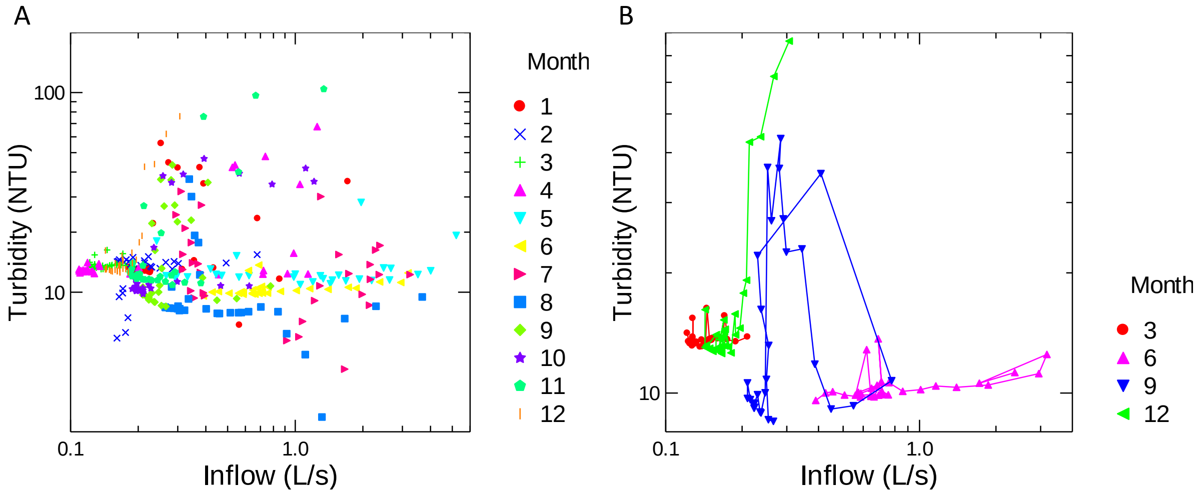
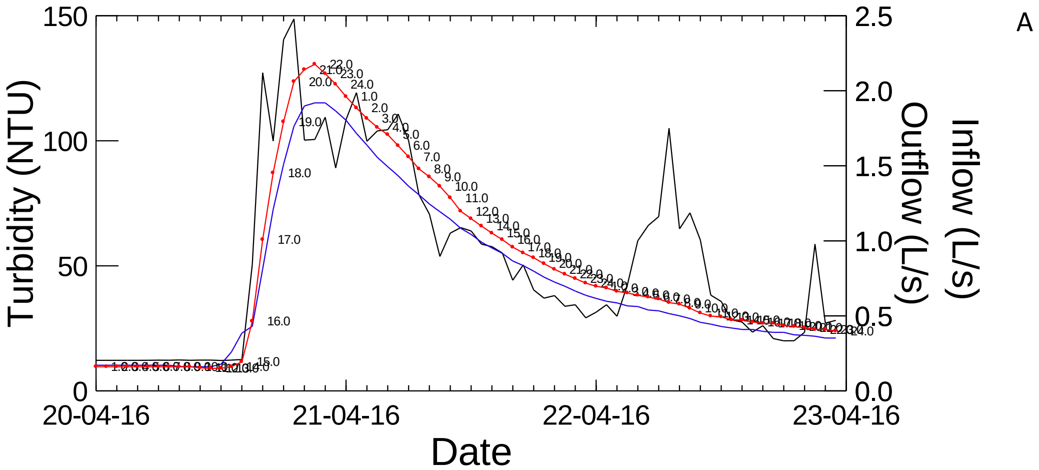


Figure 3-12: Relationship between inflow turbidity and inflow discharge rate, 2016 calendar year. A – all months, daily average turbidity, B seasonal representatives, daily average turbidity.

Figure 3-12 A indicated that a wide range of flow and turbidity values occurred in April 2016. Data for one relatively high flow discharge events are shown in Figure 3-13 A and B (turbidity and electrical conductivity respectively). Antecedent rainfall was moderate – 17.5 mm in the two-day period ending 18/04/17, and 16 mm on 20/4/16. During this event, the EC response is more closely related to the hydrograph than turbidity.



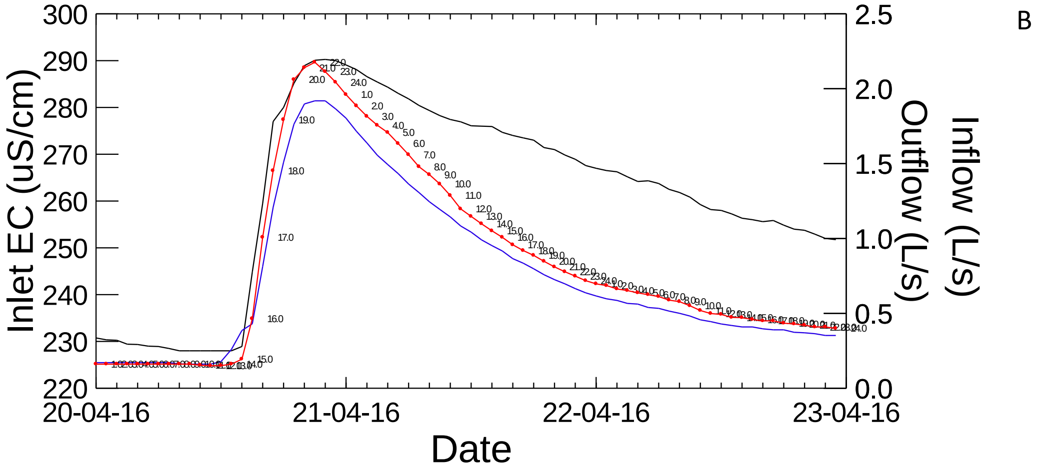


Figure 3-13: Relationship between inflow turbidity, electrical conductivity and inflow discharge, 20-22 April 2016. The red line is the inflow, the blue line is the outflow and the black line is either turbidity (A) or electrical conductivity (B). The labels indicate the time (hh.0) during each day.

3.2.2 N-filter inflow and outflow nitrogen concentrations

Inflow and outflow concentrations of three key nitrogen fractions are summarised in Figure 3-14 (B-D), which are shown in relationship to the inflow to the N filter (A). Between 99 (ammoniacal-N) and 199 (nitrate-N and TN) grab sample were collected from the N filter inflow and outflow over the 16-month assessment period). Summary statistics for forms of nitrogen and other water quality variables are included in Table E-1 through Table E-5 and in Table E-6 respectively. These tables also indicate the relative proportion of nitrate-N and ammoniacal-N of the TN concentration.

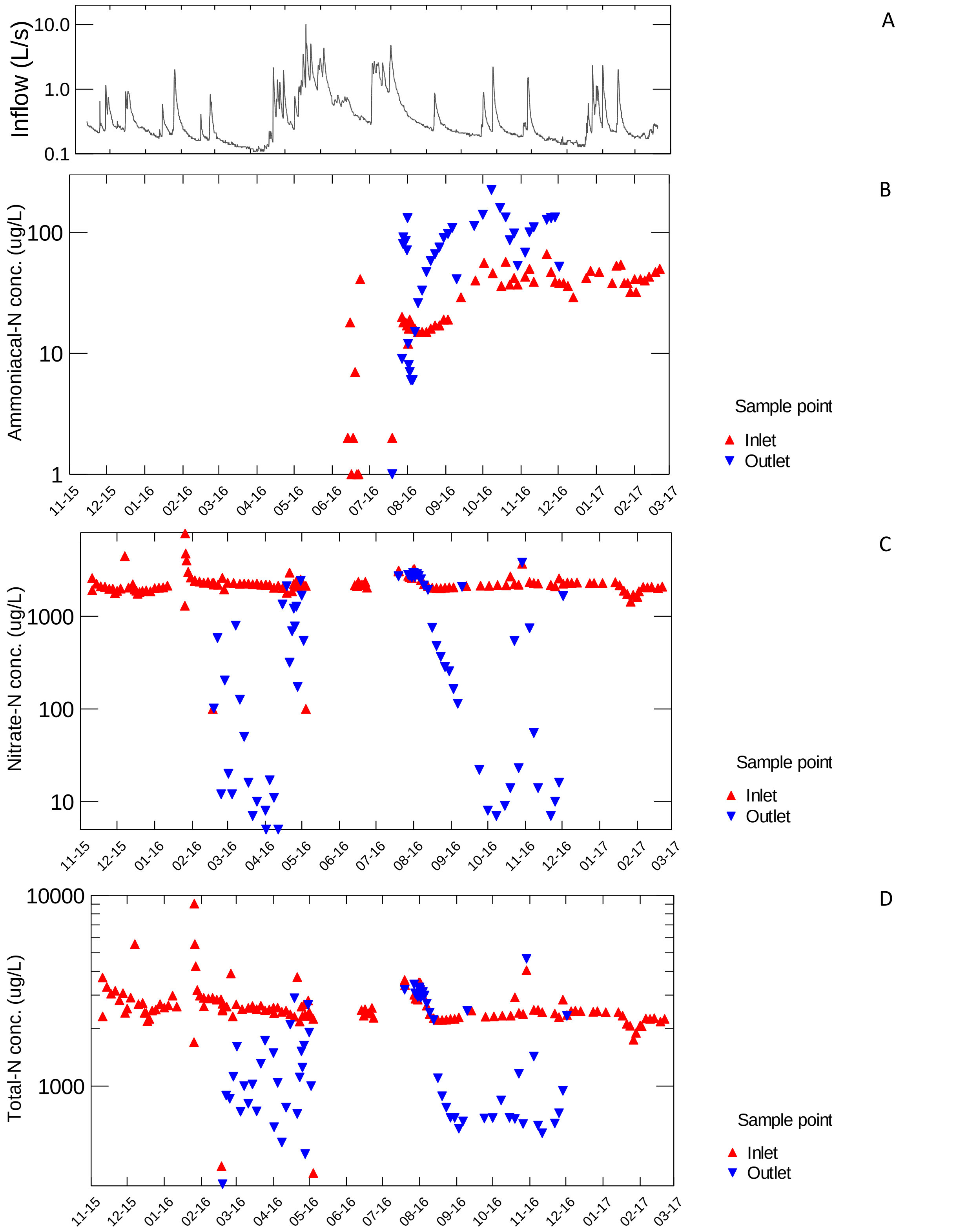


Figure 3-14: Time series of key nitrogen fraction concentrations. A) Inflow, B) ammoniacal-N, C) nitrate-N, D) Total-N. Fewer samples were collected for the outlet following two failures of automatic samplers.

3.2.3 N-filter inflow and outflow nitrogen flux estimates

The product of the measured concentrations in inflow and outflow samples with the inflow and outflow values at the time of sampling provides measures of instantaneous load, or flux. These estimates are summarised in Table 3-1, and in Figure 3-15. There are many more concentration samples in Figure 3-14 than instantaneous load estimates in Figure 3-15 – this occurs because the samples and not necessarily collected concurrently at the inlet and outlet. The number of concurrent samples was maximised by averaging concentration and flow data at hourly steps, but as Table E-2 shows, there are 66 samples each in the outlet for nitrate-N and TN – after averaging these data to maximise the number of pairs of inflow and outflow, 46 pairs remain for comparison. The number of pairs of samples is even smaller for ammoniacal-N (fewer samples were analysed for this variable). Despite the reduced number of samples, the load and removal efficacy estimates are not dissimilar to those derived from models (discussed in the following sections).

Table 3-1: Removal efficacy estimated from instantaneous load or flux estimates derived from grab samples. These estimates are derived from the inflow and outflow load estimates, expressed as proportion of inflow load (%).

Statistic	Removal efficacy (%)		
	Ammoniacal-N	Nitrate-N	Total N
N of Cases	25	46	46
Minimum	-503.9	-25.3	-33.0
Maximum	70.4	99.8	82.9
Median	-182.8	79.6	54.7
Arithmetic Mean	-179.5	64.1	44.1
Standard Error of Arithmetic Mean	36.0	5.7	4.2
95.0% LCL of Arithmetic Mean	-253.7	52.7	35.7
95.0% UCL of Arithmetic Mean	-105.3	75.5	52.6
Standard Deviation	179.8	38.4	28.6
Percentiles, Cleveland method			
1%	-503.9	-25.3	-33.0
5%	-492.6	3.9	-0.5
10%	-442.9	10.2	6.9
20%	-356.7	14.1	11.4
25%	-303.1	22.9	13.4
30%	-288.2	31.6	22.3
40%	-215.3	66.5	46.0
50%	-182.8	79.6	54.7
60%	-140.3	91.6	61.0
70%	-52.7	98.9	66.6
75%	-17.8	99.2	68.8
80%	34.6	99.4	69.6
90%	57.9	99.6	72.2
95%	67.0	99.7	77.3
99%	70.4	99.8	82.9

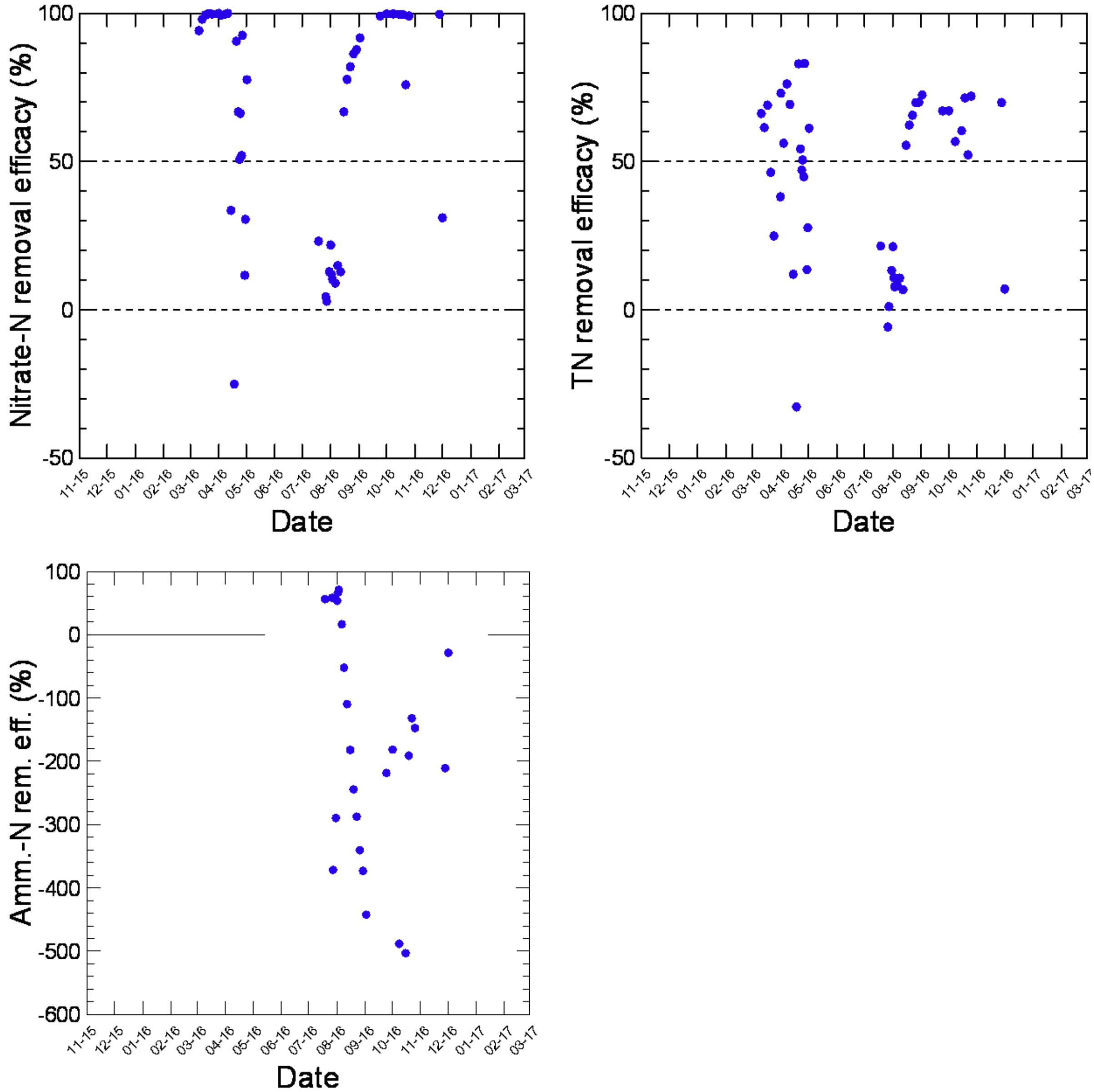


Figure 3-15: Time series of removal efficacies of key nitrogen fractions. These estimates were derived from the instantaneous inflow and outflow load estimates, expressed as proportion of inflow load (%).

3.2.4 Ammoniacal-N load estimation

Ammoniacal-N was the smallest fraction of nitrogen in the inflow and outflow, comprising 1.5% and 4.8% of inflow and outflow nitrogen respectively (median concentration values). The carbon-rich, reducing conditions within the N filter required for denitrification also favoured formation of ammoniacal-N. Consequently, outflow concentrations were generally greater than those in the inflow.

Several methods were trialled to estimate ammoniacal-N loads (simple regression, as well as 11 models incorporated in the LOADEST modelling suite). Ultimately, a regression method incorporating bootstrapping was selected. Although the fit between modelled and observed ammoniacal-N loads was not completely satisfactory (Figure 3-16), it was better than any of the other models and considered to be adequate for this exercise because ammoniacal-N made only minor contributions to the total nitrogen loads entering and leaving the N filter bed.

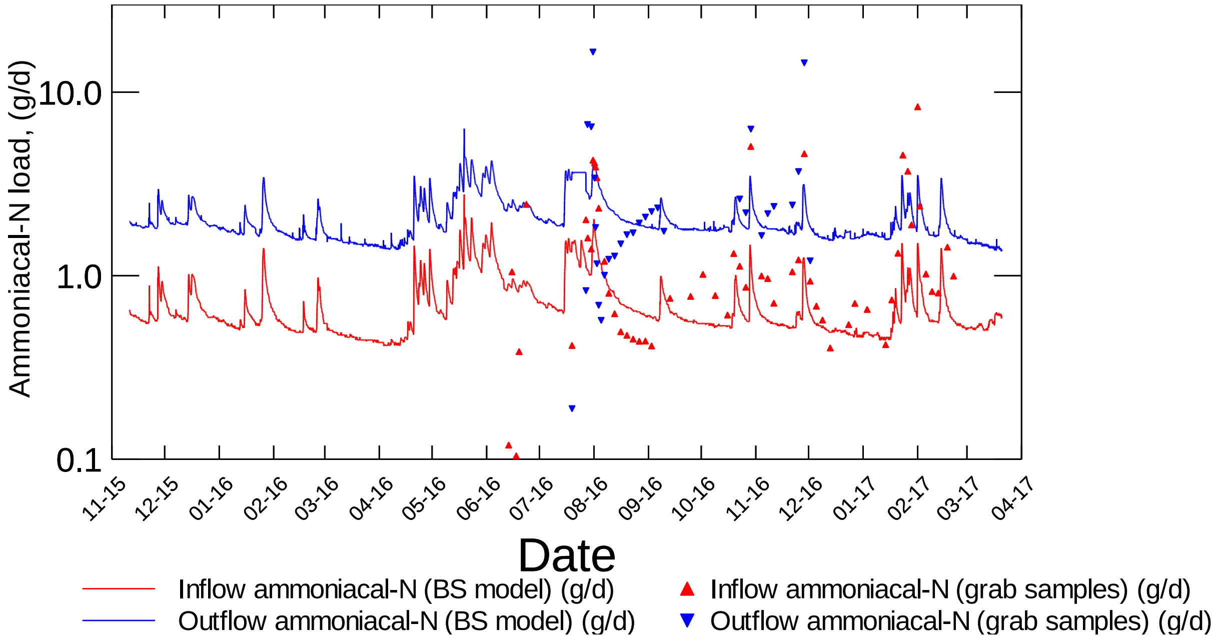


Figure 3-16: Time series of measured and predicted ammoniacal-N loads. "BS" indicates bootstrap regression model.

The performance of the N filter in terms of ammoniacal-N removal is summarised in Figure 3-17. As indicated by the relative concentrations on ammoniacal-N in inflow and outflow samples (Figure 3-14 A), biogeochemical processes within the N filter convert nitrogen fractions into ammoniacal-N form. Consequently, removal efficacy is never positive (i.e., there is always a greater load of ammoniacal-N leaving the N filter than entering it). There is a positive relationship between the magnitude of ammoniacal N leaving the N filter and detention time – as the residence time in the filter bed increases, ammoniacal-N concentrations increase (viz., the tendency for the N filter to become a source of ammoniacal N increases as flows decrease). This is consistent with relatively slow biogeochemical processes that convert inflow nitrogen fractions to ammoniacal-N.

Detailed summary statistics for ammoniacal-N inflow and outflow measured concentrations and estimated loads are included in Appendix E, Table E-1 and Table E-3.

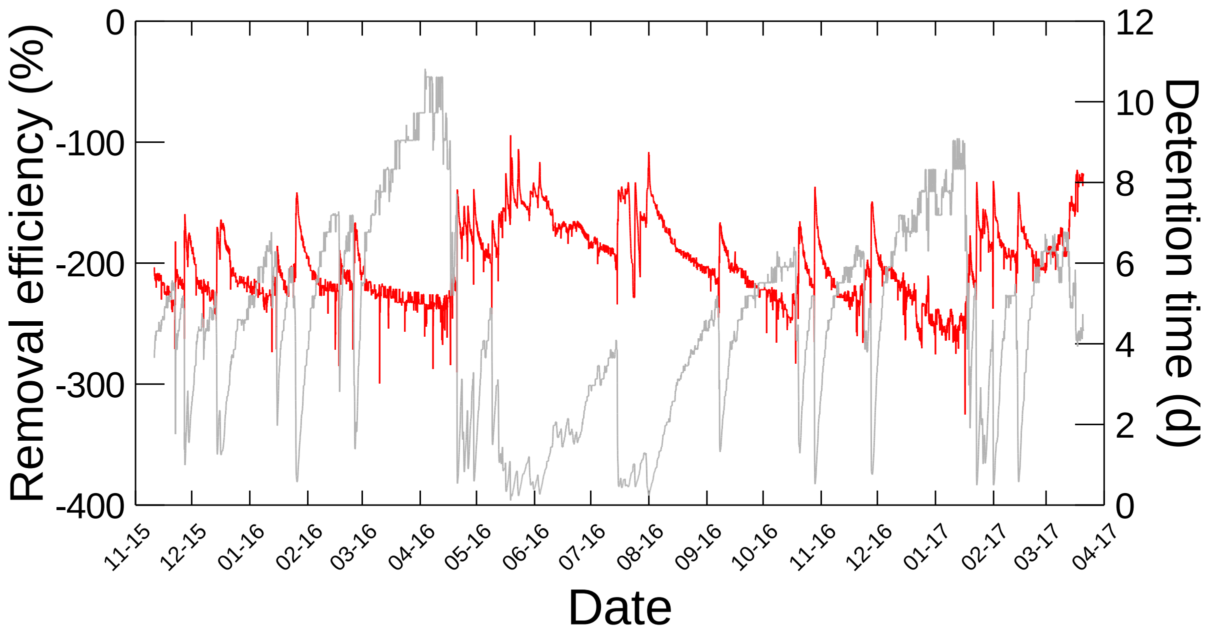


Figure 3-17: Time series of removal efficacy - ammoniacal-N loads. The red line indicates removal efficacy and the gray line indicates detention time within the filter bed.

Ammoniacal-N is of environmental concern owing to its toxicity (e.g., ANZECC/ARMCANZ 2000; MfE 2015a; MfE 2015b). Table 3-2 summarises ammoniacal-N concentrations at two locations in the Waituna Creek catchment, and compares these with the N-filter outflow concentrations and attribute state values from the NPS-FM (MfE 2014).

Table 3-2: Comparison of measured ammoniacal-N values with attribute state values from the NPS-FM. Data for Curran Creek provided by Environment Southland. Statistics for the Curran Creek sites calculated for the period January 2010 to March 2013 inclusive, values for the N-filter outflow from Table E-1. Blue shading indicates that annual median value falls within the NPS-FM "A" attribute state (≤0.03 mg/L), Green shading indicates that median values fall within the NPS-FM "B" attribute state (>0.03 - ≤0.4 mg/L).

Measurement site	Ammoniacal-N concentration (mg/L)					No. values
	Year	Average	Median	Min	Max	
Curran Creek at Waituna Road	2010	0.05	0.042	0.005	0.104	12
	2011	0.072	0.014	0.005	0.48	14
	2012	0.156	0.095	0.005	0.51	23
Curran Creek at Marshall Road	2010	0.041	0.036	0.005	0.17	12
	2011	0.213	0.186	0.005	0.59	82
	2012	0.079	0.088	0.005	0.34	30
N-filter discharge	2015-2017	0.076	0.077	0.001	0.224	38

The relatively small N-filter discharge rate (largest maximum flow 11.7 L/s) relative to the lowest minimum flow value available for Waituna Creek (58 L/s in 2012) indicates that the additional ammoniacal-N load discharged from the N-filter is unlikely to impair water quality. Sampling of Waituna Creek at key times would confirm this assessment, or indicate whether caution is required.

3.2.5 Nitrate-N load estimation

Figure 3-14 C demonstrates that inflow nitrate-N concentrations were typically 2000 mg/m3 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 appeared to be influenced by season (as well as other factors).

Several methods were explored to predict loads based on continuous flows and grab samples of concentration. For inflows, several flow-concentration models gave a satisfactory fit and hence predicted inflow loads reasonably well. However, models were less successful in predicting outflow loads. An example is shown in Figure 3-18. A very good model fit was obtained for the predicted inflow loads but predicted outflow loads are less satisfactory, particularly in the late summer/autumn, and again in late spring. Model calibration was likely to have been adversely affected by the malfunction of the automatic samplers over the spring/summer of 2016/2017. The model predictions are, however, adequate for estimating nitrate-removal performance of the filter over the assessment period.

Points to note from Figure 3-18 include:

- the model over-predicts nitrate-N outflow loads in the late summer/autumn period (i.e., removal is likely to be under-predicted)
- both models appear to represent peak nitrate-N loads adequately.

A comparison of nitrate N loads estimated from grab samples and alternate models is shown in Figure E-4.

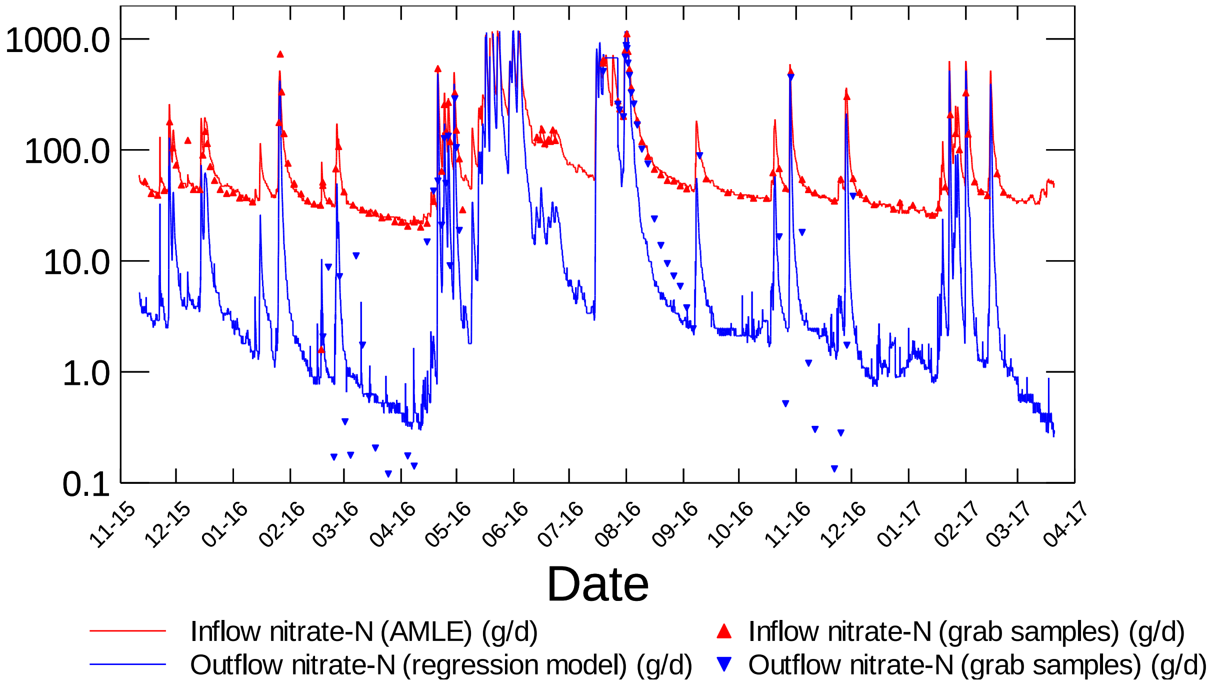


Figure 3-18: Time series of measured and predicted nitrate-N loads.

Nitrate-N removal efficacy is summarised in Figure 3-19. Except for periods of high inflow (i.e., over periods of short detention time), removal efficacy is generally high. As Table 3-3 indicates, 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 detention time is apparent in Figure 3-19, and Figure 3-20 A confirms the strong correlation. Figure 3-20 B shows that there is a seasonal variation in removal rate which is likely to be related to temperature.

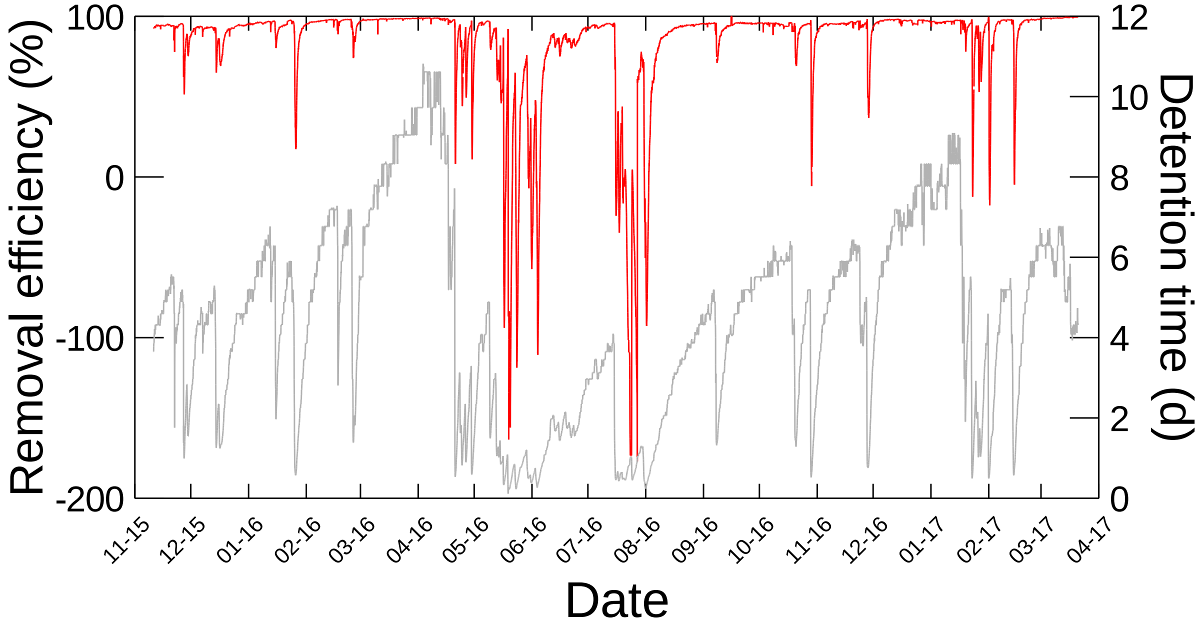


Figure 3-19: Time series of removal efficacy - nitrate-N loads. The red line indicates removal efficacy and the gray line indicates detention time within the filter bed.

Table 3-3: Summary statistics for inflow and outflow nitrate-N loads, and removal efficacy.

Statistic	Inflow nitrate-N (AMLE model) (g/d)	Outflow nitrate-N (regression model) (g/d)	Removal efficacy (%)
N of Cases	11904	11894	
Minimum	21.3	0.3	98.8
Maximum	5056	53462	-957
Median	47.1	2.9	93.8
Arithmetic Mean	110	78.6	28.7
Mode	394	2.5	95.3
Standard Deviation	198	688	33.8
Percentiles of time (Cleveland method)			
1%	21.8	0.3	98.4
5%	24.9	0.5	98.1
10%	28.6	0.6	97.8
20%	34.3	1.1	96.8
25%	35.8	1.3	96.4
30%	36.9	1.5	95.8
40%	41.7	2.4	94.3
50%	47.1	2.9	93.8

Statistic	Inflow nitrate-N (AMLE model) (g/d)	Outflow nitrate-N (regression model) (g/d)	Removal efficacy (%)
60%	54.6	3.9	92.9
70%	70.0	6.5	90.7
75%	89.2	10.6	88.2
80%	117	19.5	83.3
90%	240	92.8	61.4
95%	460	439	4.7
99%	929	1380	-48.5

In Figure 3-21, outflow nitrate-N load, removal efficacy and outflow are plotted against temperature for selected months (incorporating the greatest temperature range). The close relationship between flow and nitrate outflow load is confirmed (A and C). It is also apparent that removal efficacy varies firstly with flow, and secondly with temperature. In general, removal efficacy increases with temperature, and decreases with flow (viz., detention time). In Figure 3-21 A, for example, data for April and July 2016 falls into two different groups, differentiated by relatively low- or elevated nitrate-N load. Figure 3-21 C indicates that this differentiation is related to flow, rather than temperature. In any month, nitrate removal is greater under low flow conditions. This is further confirmed by comparing data for March 2016 and March 2017 (Figure 3-22 A-C). Removal efficacy was lower in March 2016, even though the temperature was slightly higher, because detention time in the filter bed was shorter (due to slightly higher flow rates).

In Figure E-2 and Figure E-3, the distribution of nitrate-N removal rates is summarised by month, which confirms that lower removal rates occur in winter when residence times are shorter and temperatures lower.

These data show that removal is strongly affected by residence time (viz., contact time between nitrate-N, the microbial biomass and the organic carbon that acts as electron donor). The inflow to the filter bed is determined by soil moisture balance. During periods of rainfall and high soil moisture, the detention time is approximately one day, which is inadequate for substantial nitrate-N removal. Improving the nitrate-N removal performance during months of high rainfall will require implementation of additional facilities, such as balancing ponds or bypass arrangements. These are discussed briefly in Section 4.3.

A

B

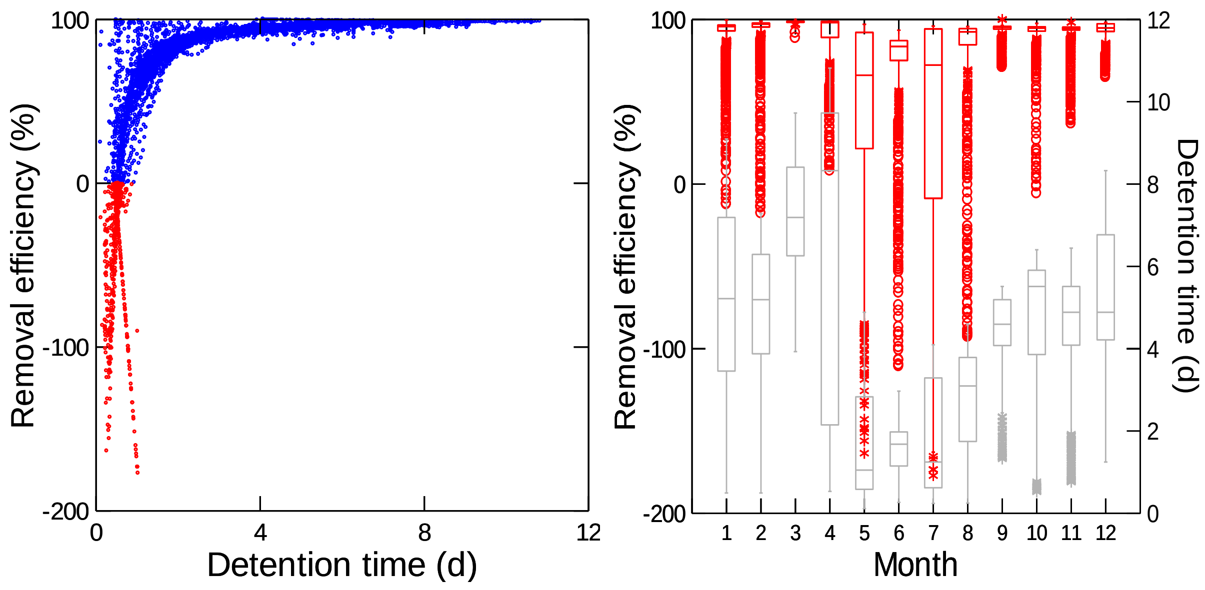


Figure 3-20: Relationship between removal efficacy and flow through the N filter. In A), the red coloured dots constitute approximately 4.5% of 12,000 hourly estimates made during the assessment period. In B), removal efficacy (red) has a strong seasonal component, inversely related to detention time within the filter bed (gray).

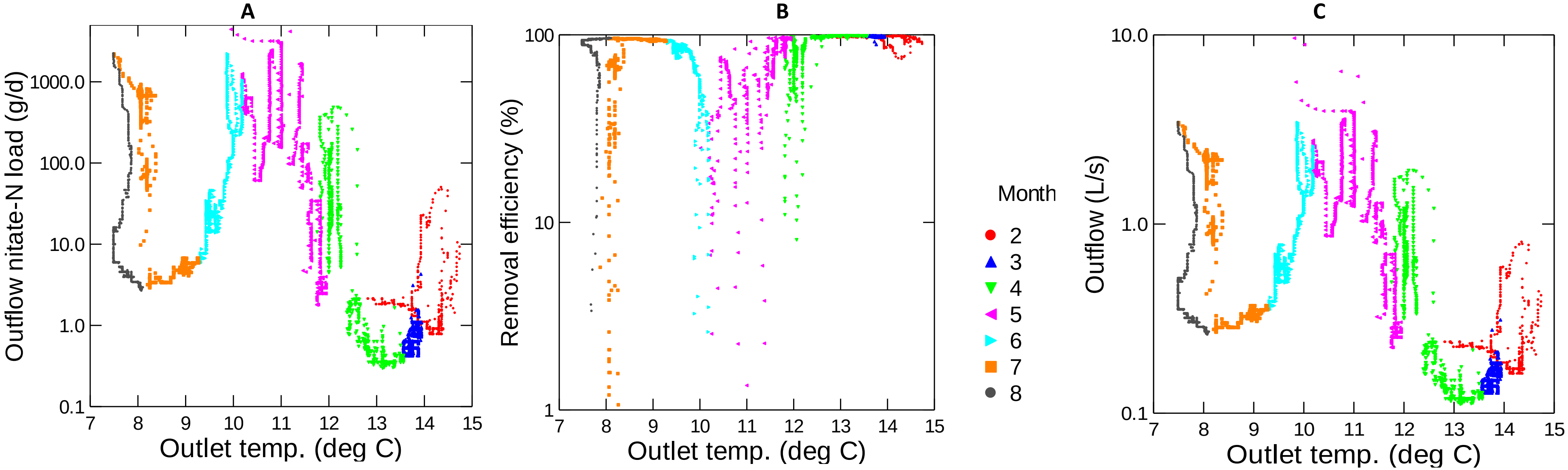


Figure 3-21: Relationship between outflow load, removal efficacy and flow through the N filter with temperature for selected months. Data for 2016 calendar year only.

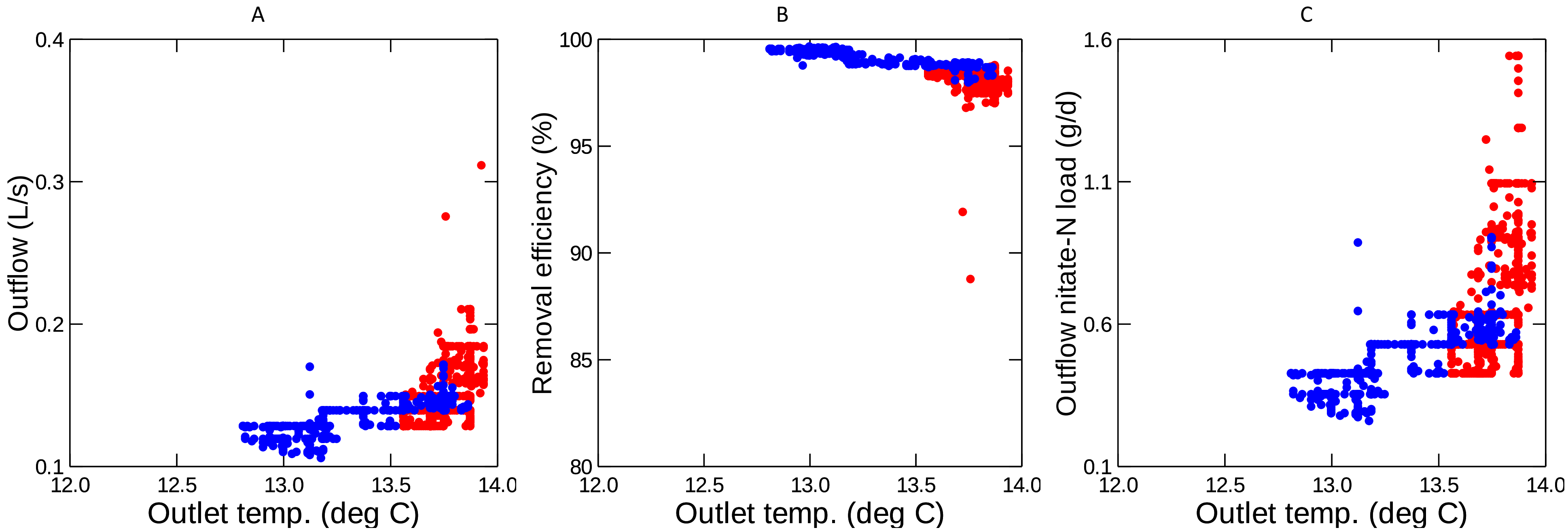


Figure 3-22: Relationship between outflow load, removal efficacy and flow through the N filter with temperature. Data for March 2016 (red) and March 2017 (blue).

3.2.6 Total-N load estimation

Figure 3-14 D shows that TN concentrations did not vary much during baseflow but increased during runoff events (varying by a factor of approximately two to five). There are two anomalously low inflow TN measurements. The outflow concentrations varied more widely (approximately one order of magnitude), showed seasonal variations and may have been influenced by other factors.

Several methods were explored to predict inflow and outflow loads. All were able to predict inflow loads very well, and most predicted outflow loads tolerably well. An example of measured and predicted TN inflow and outflow loads is shown in Figure 3-23. A very good model fit was obtained for the predicted inflow loads. Predicted outflow loads were less well represented by the model, particularly in the late summer/autumn, and again in late spring. Model calibration was likely to have been adversely affected by the malfunction of the automatic samplers during spring/summer 2016/2017. The model predictions were, however, adequate for estimating filter performance in terms of TN removal during the assessment period.

Points to note from Figure 3-23 include:

- The model over-predicted outflow TN loads in the late summer/autumn period (i.e., removal is likely to be under-predicted).
- The model appeared to over-predict outflow TN loads in late winter, although fewer data are available to estimate this – the removal efficacy is likely to be under-predicted at these times as well.
- Models appear to represent peak inflow and outflow TN loads tolerably well.

TN removal efficacy is summarised in Figure 3-24. Removal efficacy is lower than that estimated for nitrate-N. For example, the 90th percentile removal efficacy for nitrate-N was approximately 60%, which was similar to the TN 10th percentile removal efficacy.

Figure 3-24 and Figure 3-25 both indicate a negative relationship between inflow and TN removal efficacy, as was observed for nitrate-N.

Nitrate-N is the dominant fraction of TN, comprising approximately 90% and 47% of TN in the inflow and outflow respectively (median values). The magnitude of the median inflow nitrate-N load is similar to that of the median TN inflow load (47 g/d and 55 g/d, respectively), whereas the median outflow TN load is considerably larger than the median inflow nitrate-N load (28 g/d and 2.9 g/d, respectively).

The median nitrate-N concentration was reduced from 2,190 µg/L to 421.5 µg/L (viz., by 80%), whereas the median TN inflow and outflow concentrations were 2,495 µg/L and 1,105 µg/L respectively (viz., a reduction by 55%). Although these values indicate that more nitrate-N is removed than suggested by inflow and outflow TN concentrations, this may be an artefact arising from the poor model fit for the outflow rates, and the smaller number of outflow samples that were analysed.

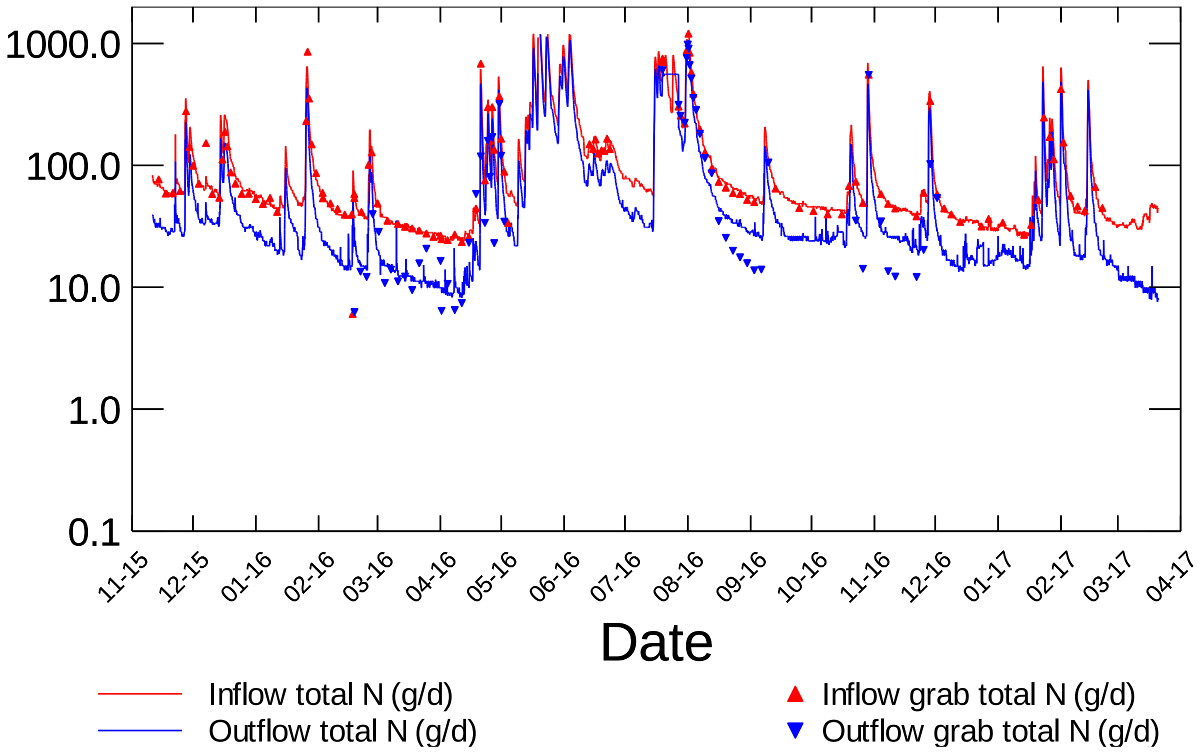


Figure 3-23: Time series of measured and predicted total N loads.

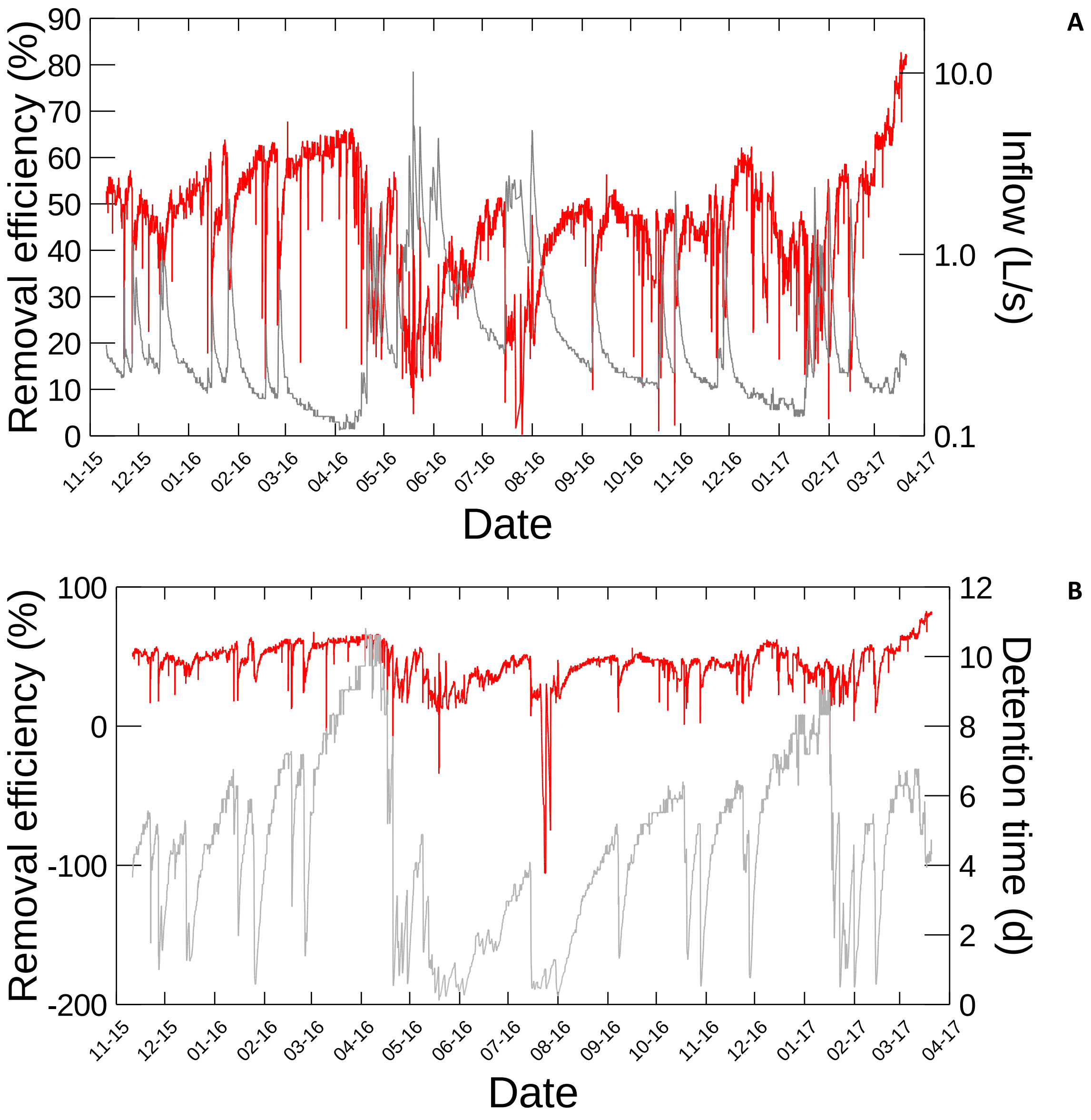


Figure 3-24: Time series of total N load removal efficacy. A) removal efficacy vs. inflow, and B)) removal efficacy vs. detention time. In each case the red line indicates removal efficacy.

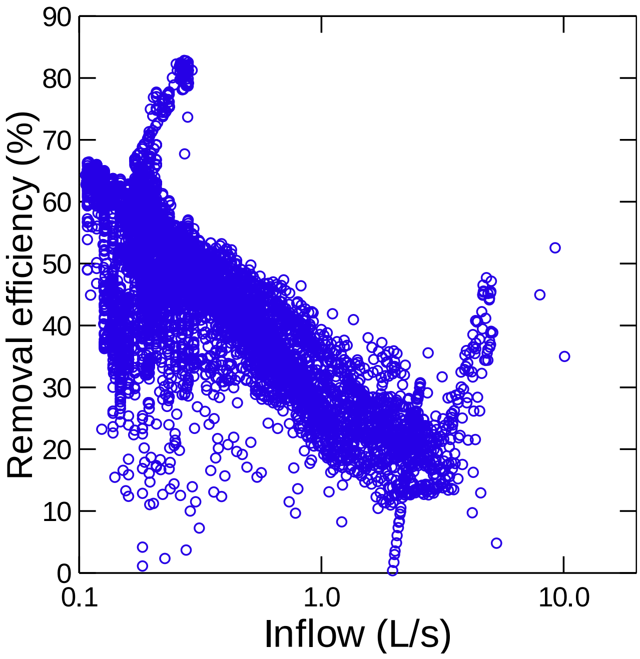


Figure 3-25: Relationship between inflow and total N removal efficacy.

Table 3-4: Summary statistics for inflow and outflow total-N loads, and removal efficacy.

Statistic	Inflow total N (g/d)	Outflow total N (g/d)	Removal efficacy (%)
N of Cases	11904	11894	11894
Minimum	24.0	7.6	-105.8
Maximum	6208.9	5975.9	82.7
Median	55.6	28.5	46.9
Arithmetic Mean	122.2	83.3	44.9
Standard Deviation	223.4	176.8	15.7
Percentiles (Cleveland method)			
1%	24.4	8.9	2.5
5%	28.1	10.7	21.1
10%	30.8	12.3	27.3
20%	36.2	16.7	35.3
25%	39.6	18.3	38.6
30%	42.5	20.2	41.1
40%	47.8	25.5	44.5
50%	55.6	28.5	46.9
60%	63.8	33.5	49.1
70%	77.9	44.3	52.3
75%	99.3	57.8	54.0
80%	130.0	80.9	55.8
90%	258.3	189.5	60.6
95%	498.4	441.5	63.2
99%	1015.1	823.0	75.2

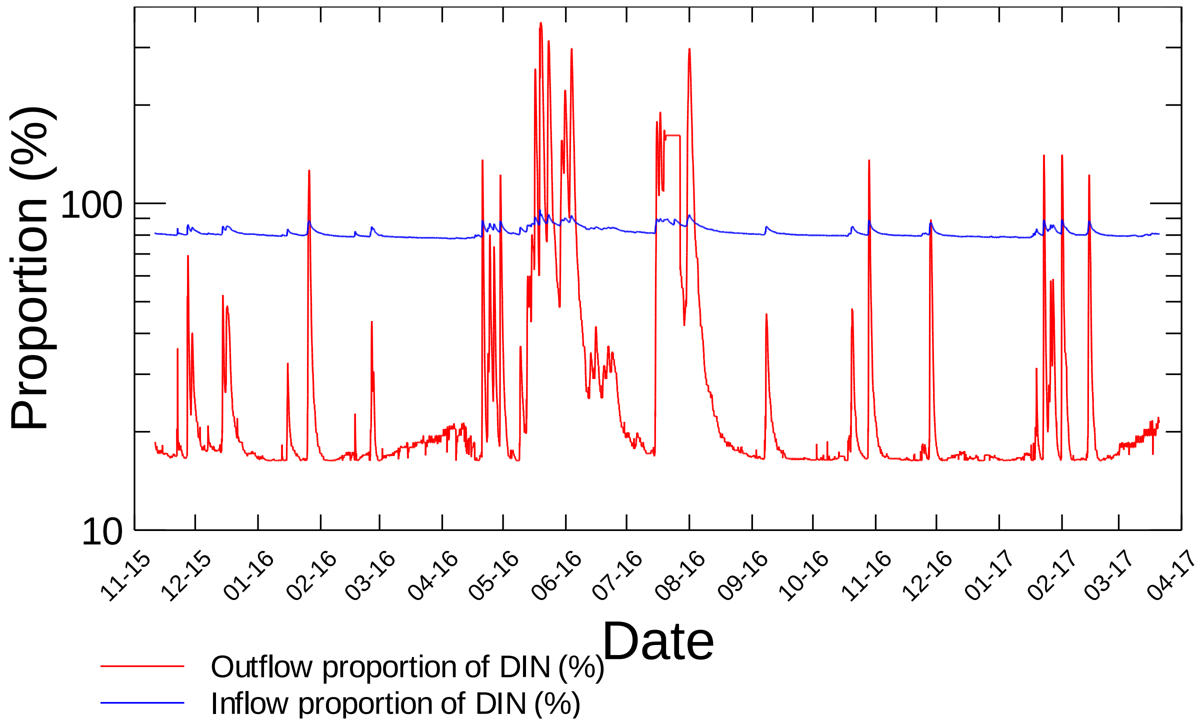


Figure 3-26: Time series of proportion of DIN in TN in inflow and outflow respectively.

4 Summary and conclusion

4.1 P filter performance

Earlier work undertaken by NIWA demonstrated the P binding capacity of Aqual-P at pH 7 was 23 g P/kg (Gibbs et al. 2008). Subsequent work indicated the P binding capacity used as a capping agent in lakes was in a range from 1.6 – 2.3 (w/w %) (Hickey and Gibbs 2009). Using a continuous flow incubation system, Gibbs et al. (2011) demonstrated a P-binding capacity of 21 g P/kg Aqual-P at pH 6.1 – 7.0. These trials demonstrated that P removal and retention were influenced by changes in oxic conditions within the filter bed. Cycling that included anoxia led to P release from the bed.

In this work, the P filter demonstrated high removal of dissolved reactive phosphate, and moderate removal of TP. Although the redox conditions within the bed were not measured during the field trial, it is likely that the low hydraulic loading rate and non-sealed nature of the system favoured input of sufficient oxygen to avoid anoxia developing. An increase in P concentrations were observed in spring, but this was limited to the filter inflow, and there was no evidence of liberation from the filter material.

There is room for speculation regarding the actual hydraulic loading rate. The system allowed estimation of inflow, outflow and bypass flows. There were periods during which measured flows were very small. The pump unit was disassembled for maintenance during April 2017, which revealed sediment accumulation within the pump inlet. This was likely to have influenced pumping efficiencies and pump rates over the life of the project. It was not possible with the data available to identify to what extent the P loading varied over the trial life. This was a major (unanticipated) flaw in the experimental design. It is instructive however. Pumped systems should probably be avoided in what are intended to be low-cost, low-maintenance, passive treatment systems. Future design and implementation should favour gravity-fed systems to reduce the likelihood of blockages or inconsistent pump performance. A gravity-fed system could incorporate an inlet buffer tank with facilities to regulate the inflow and allow excess water to bypass the filter. Options for achieving this might include:

1. An orifice plate that would allow the inflow to be capped. This would allow:
   i. all flows less than the orifice plate design capacity to enter the treatment system, and when total tile drainage flow exceeded the orifice plate,
   ii. excess flow would be diverted over a second weir, bypassing the treatment unit.
2. An adjustable inlet and bypass weir, that would allow:
   i. Seasonal adjustment of the inflow by reducing the height of the inlet weir relative to the outlet weir
   ii. In all seasons, allow excess water to bypass the treatment unit.

Potentially many other options exist – additional trials and design work is required to define the dimensions of these and other inlet and bypass options. Design work should also consider how sediment entering the system would be trapped upstream of the filter unit.

Available data indicate:

- DRP removal efficacy consistently exceeded 97% of influent DRP concentration.

- This performance occurred over inflow DRP concentrations that ranged approximately six-fold, from approximately 60 µg/L to 360 µg/L.
- TP removal efficacy was more variable, ranging from 78% to 87% removal of influent P concentration.
- Inflow TP concentrations also ranged approximately six-fold, from approximately 140 µg/L to over 900 µg/L.
- Although influent DRP and TP concentrations varied seasonally, there was little evidence of seasonal variation in DRP or TP concentrations in the outflow.

A proposal for further assessment of the P filter has been submitted for consideration. The programme of work proposed will provide an opportunity to refine and confirm these preliminary results regarding:

- P removal rates at increased hydraulic loading rates, and
- to determine the P removal capacity of the filter medium by increasing the hydraulic loading rate, as well as
- determining the role that factors such as dissolved oxygen concentration and pH have on P removal under various seasonal conditions.

4.2 N filter performance

Ammoniacal-N was the smallest fraction of nitrogen in the inflow and outflow, comprising 1.5% and 4.8% of inflow and outflow nitrogen respectively (median concentration values). Inflow and outflow ammoniacal-N loads comprised approximately 1% and 6.3% of the TN loads respectively. Ammoniacal-N concentrations and loads increased following treatment in the N-filter by 113% and 204% respectively. Although the ammoniacal-N fraction generated in the N-filter appears modest in comparison with the inflow nitrogen load, it would be prudent to assess the impact that this potentially toxic material could have on the receiving environment. From the data available, concerns regarding adverse effect on receiving waters appear unfounded.

Median nitrate-N concentrations were 2190 µg/L and 421 µg/L in the inflow and outflow respectively, indicating a decrease of 80%. Modelling allowed the seasonal performance of the N-filter to be estimated, and results indicated that on average annual nitrate-N loads decreased from 47 g/d to 2.9 g/d, a reduction of almost 94%.

Median total N loads decreased from 55.5 g/d to 28.5 g/d, a reduction of almost 47%.

For both nitrate-N and TN, removal performance was influenced strongly by hydraulic loading rate, with a negative relationship between inflow rate and treatment efficacy.

Nitrate-N removal performance appears to vary seasonally, but this is primarily the result of seasonal variations in retention time. Temperature does influence nitrate-N removal (removal rates increase with temperature), but is less of a factor than detention time in the filter bed.

4.3 Modifications that may improve N-filter performance

The requirement to improve the nitrate-N removal performance warrants further consideration. From the data available, mitigation is impaired principally by inadequate residence time, which is determined by the hydraulic loading to the filter bed. The periods of poor nitrate-N removal will tend to occur in the winter, when flows across the catchment will be higher, and residence times in the lagoon will be shorter. The timing of delivery and magnitude of these loads in the receiving environment may not lead to undesirable ecological outcomes, and in these circumstances may be tolerable. Consideration of the magnitude and timing of the delivery of these loads could be done using catchment models and ecological surveys.

Should further improvement in nitrate-N removal be required, several options exist, including those identified for the P-filter in Section 4.1:

- inclusion of a flow-balancing basin on the inflow to the N-filter:
  - this would allow the inflow to be controlled within a narrow range
  - a system of this nature would probably require a pump as well, because there would be few situations where an entirely gravity-fed system could be implemented
- managing the additional hydraulic load within the paddock and tile drain system as a "temporary storage" system, as undertaken in various controlled drainage strategies (e.g., Christianson et al. 2016):
  - this would probably require tolerance of higher soil moisture across the field being drained, and careful management of soil moisture to ensure that agronomic performance is maintained
  - the tile drain itself could possibly provide "distributed storage", but this would also require careful management – reducing outflow from the drains would create additional hydraulic pressure, and could cause excess leakage within the drainage system, and potentially rupturing of the tile drain and formation of "pot-holes" within the paddock.
- Inclusion of a simple weir and bypass facility, which would allow flow that exceeds a target threshold to bypass the filter bed:
  - a system of this nature could be incorporated within the existing tile drainage system more simply than a flow balancing detention basin
  - the bypass could be achieved by installing the discharge pipe from the tile drain system, the inflow to the filter bed and the bypass pipe within a simple manifold (such as a 200 L drum), where the relative heights of the three pipes could be adjusted independently to achieve the levels and flow rates required.

5 Acknowledgements

The efforts of Dr Lucy McKergow in organising the site establishment and getting the monitoring programme under way.

The hardworking technical team (NIWA and Dairy Green) for keeping things going.

Assistance from NIWA Instrument Systems team, Christchurch.

Timely sample processing by the Water Quality Laboratory, Hamilton.

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Appendix A Schematics of the N and P filters and associated instrumentation

P filter

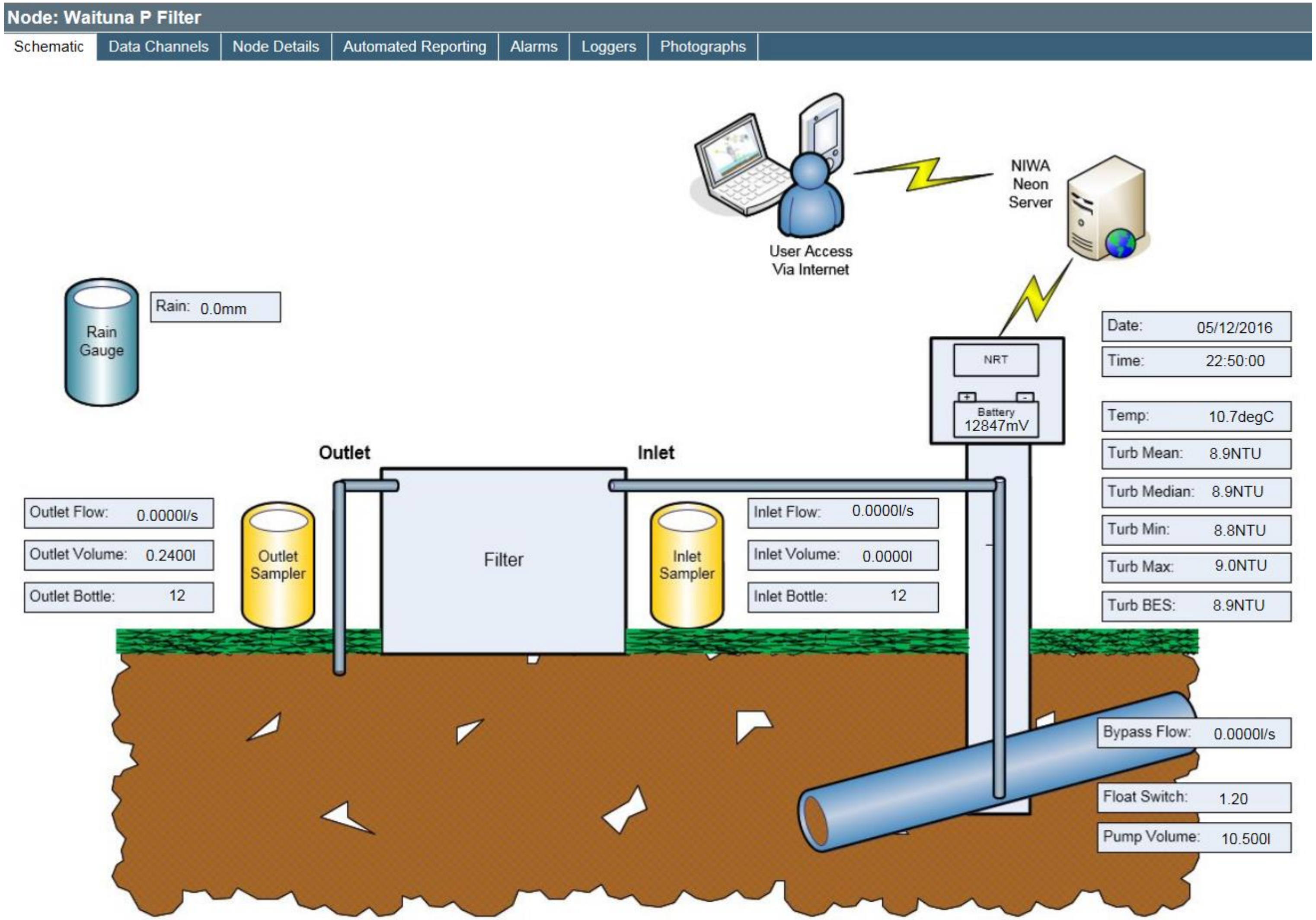

Figure A-1: Schematic of P filter, showing the relative location of equipment, measuring sensors and sampling equipment.

N filter

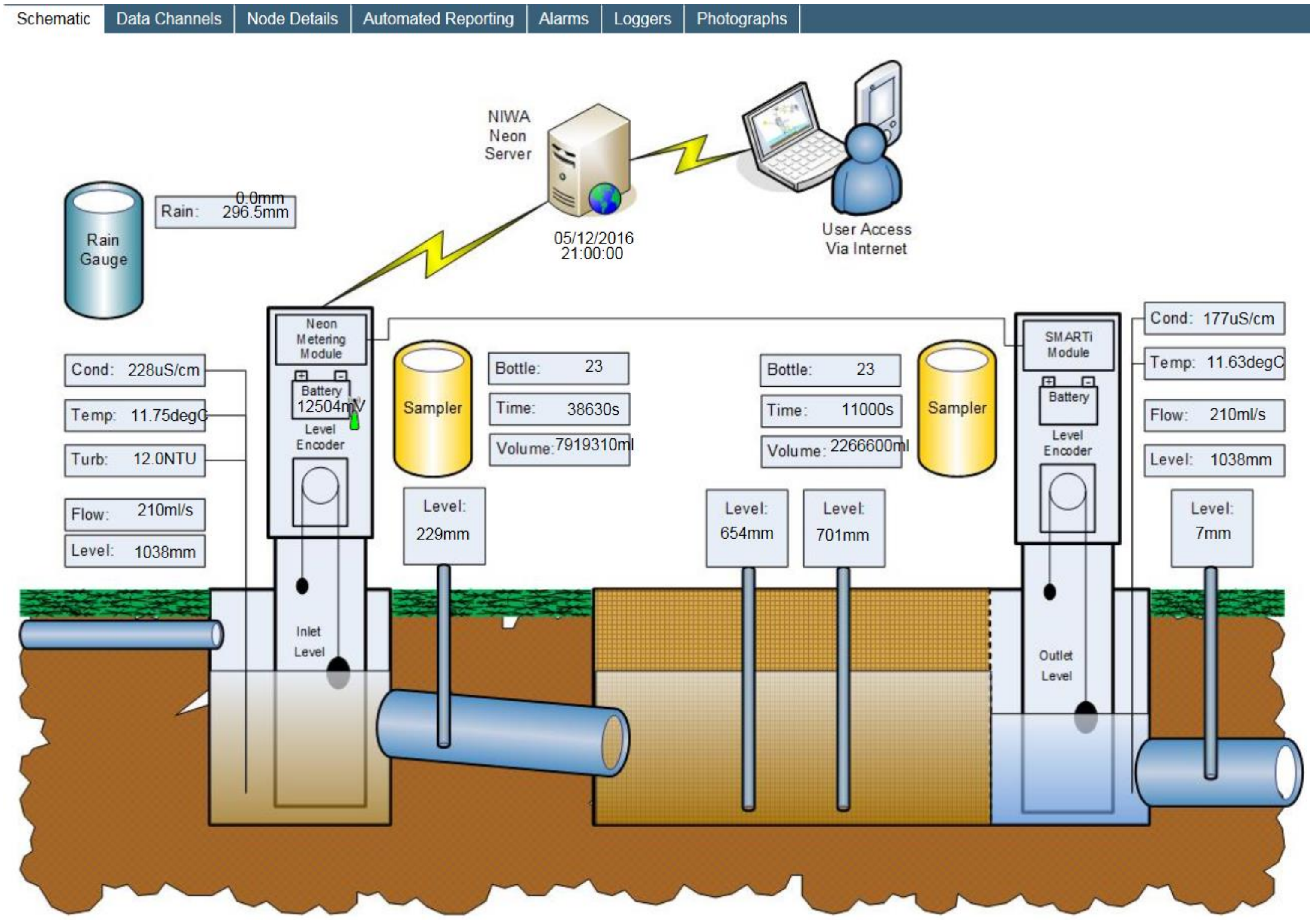

Figure A-2: Schematic of N filter, showing the relative location of equipment, measuring sensors and sampling equipment.

Appendix B Water quality analyses

Table B-1: Analytical methods and limits of detection.

Variable	Description	Detection limit	Units	Method
Electrical conductivity (EC)	Electrical conductivity meter, measured at 25°C	0.01	µS/cm	APHA 2510B
Automated cadmium reduction	Conversion of nitrite-N to nitrate-N, Flow injection analyser (FIA)		N/A	
Nitrate, nitrite nitrogen (Nitrate-N)	FIA	1	mg/m^3 or µg/L	Lachat
Ammonium nitrogen (Ammoniacal-N)	FIA	1	mg/m^3 or µg/L	Lachat
Total nitrogen (TN)	Persulphate digest, auto cadmium reduction, FIA	10	mg/m^3 or µg/L	Lachat
Dissolved Reactive Phosphorus (DRP)	Molybdenum blue colorimetric method FIA	1	mg/m^3 or µg/L	Lachat
Total Phosphorus (TP)	Persulphate digest, molybdenum blue colorimetric method, FIA	1	mg/m^3 or µg/L	Lachat

Appendix C P filter water quality statistics and additional figures

Table C-1: Concentrations of DRP and TP in P filter inflow and outflow and removal efficacy. Data summarised over the entire 10-month assessment period. The detection limit for DRP was 1.0 µg/L – values reported as "below detection limit" were replaced with 0.5 to calculate statistics and estimate loads.

Statistic	Inflow conc. (µg/L)		Outflow conc. (µg/L)		Removal efficacy (%)	
	DRP	TP	DRP	TP	DRP	TP
N of Cases	33	33	32	31	-	-
Minimum	58.0	139.0	0.5	19.0	99.1	86.3
Maximum	362.0	916.0	10.7	126.0	97	86.2
Median	88.0	195.0	0.5	42.0	99.4	78.5
Arithmetic Mean	115.4	294.8	1.5	50.3	98.7	82.9
Standard Deviation	73.0	219.6	2.3	28.6	-	-
Percentiles (Cleveland method)						
1%	58.0	139.0	0.5	19.0	99.1	86.3
5%	64.3	142.5	0.5	20.0	99.2	85.9
10%	71.6	157.0	0.5	22.8	99.3	85.5
20%	76.2	163.7	0.5	28.1	99.3	82.8
25%	82.5	173.7	0.5	29.0	99.4	83.3
30%	86.0	176.4	0.5	33.0	99.4	81.3
40%	86.1	181.7	0.5	36.7	99.4	79.8
50%	88.0	195.0	0.5	42.0	99.4	78.5
60%	92.0	208.2	0.5	46.3	99.5	77.8
70%	99.2	284.6	0.5	58.6	99.5	79.4
75%	108.2	331.5	0.95	61.7	99.1	81.4
80%	135.1	341.1	1.82	67.2	98.7	80.3
90%	199.2	646.0	4.0	98.0	98	84.8
95%	338.2	883.1	6.9	112.5	98	87.3
99%	362.0	916.0	10.7	126.0	97	86.2

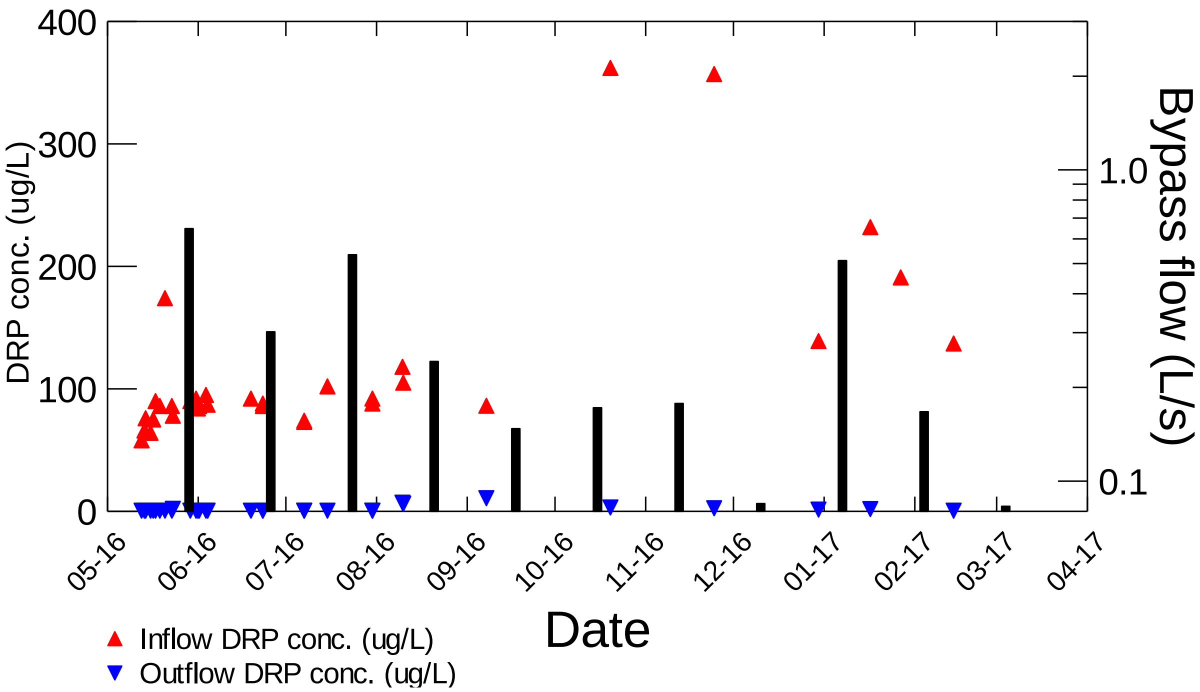

Figure C-1: Grab sample inflow and outflow DRP concentrations and monthly average bypass flow values. The period extends from May 2016 to March 2017.

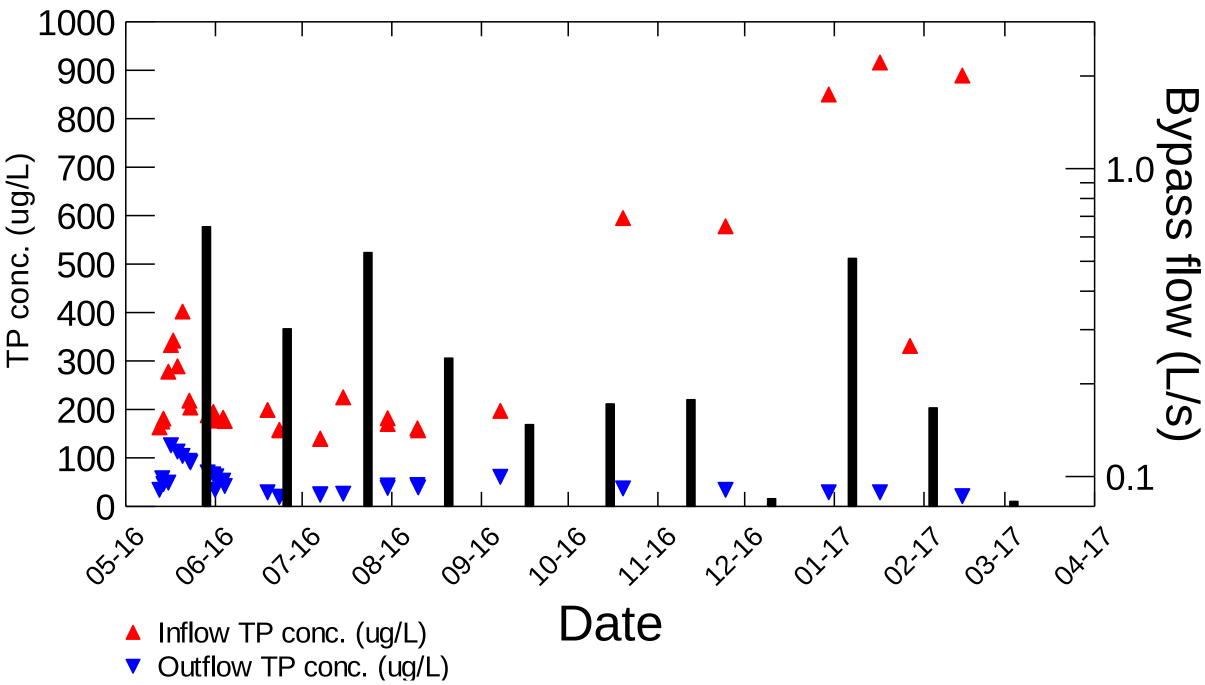

Figure C-2: Grab sample inflow and outflow TP concentrations and monthly average bypass flow values. The period extends from May 2016 to March 2017.

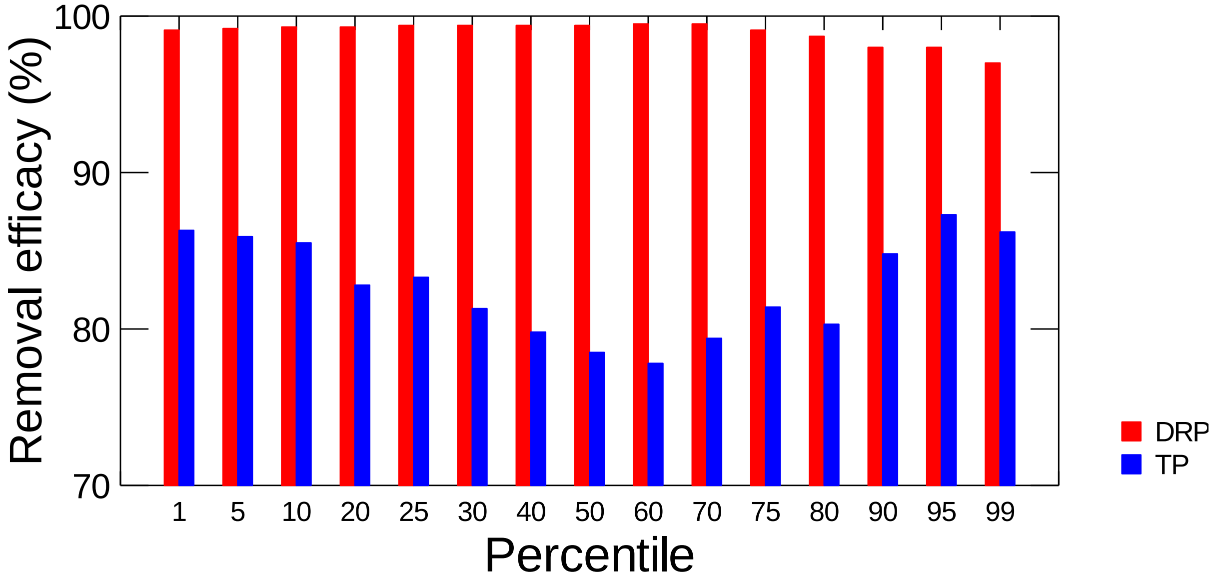

Figure C-3: Removal efficacies for DRP and TP in terms of grab sample concentration percentiles. This figure indicates that removal performance is practically invariant across a wide concentration range (DRP), but more variable in terms of TP concentration. No explanation exists for the latter situation.

Appendix D N filter flow statistics

Table D-1: Summary N filter inflow and outflow statistics. Data for period May 2015 to April 2017.

Statistic	Entire period	
	Inflow (L/s)	Outflow (L/s)
N of Cases	11904	11904
Minimum	0.1	0.3
Maximum	10.16	11.72
Median	0.25	0.26
Arithmetic Mean	0.48	0.47
Standard Deviation	0.63	0.57
Percentiles (Cleveland method)		
1%	0.11	0.12
5%	0.13	0.13
10%	0.15	0.15
20%	0.18	0.18
25%	0.19	0.19
30%	0.19	0.21
40%	0.22	0.25
50%	0.25	0.27
60%	0.29	0.3
70%	0.37	0.36
75%	0.47	0.44
80%	0.59	0.55
90%	1.10	1.01
95%	1.84	1.845
99%	3.12	2.87

Table D-2: Seasonal summary statistics for N filter inflow and outflow – January-March. Data for period May 2015 to April 2017.

Statistic	January		February		March	
	Inflow (L/s)	Outflow (L/s)	Inflow (L/s)	Outflow (L/s)	Inflow (L/s)	Outflow (L/s)
N of Cases	1488	1488	1368	1368	1224	1224
Minimum	0.13	0.16	0.16	0.16	0.12	0.10
Maximum	2.348	1.969	2.348	1.969	0.29	0.31
Median	0.222	0.244	0.223	0.204	0.16	0.14
Arithmetic Mean	0.340	0.347	0.319	0.297	0.17	0.14
Standard Deviation	0.339	0.286	0.305	0.266	0.04	0.02
Percentiles (Cleveland method)						
1%	0.128	0.168	0.161	0.161	0.119	0.111
5%	0.138	0.174	0.161	0.169	0.128	0.119
10%	0.139	0.188	0.171	0.171	0.128	0.128
20%	0.160	0.199	0.184	0.179	0.128	0.128
25%	0.161	0.209	0.184	0.184	0.138	0.133
30%	0.184	0.210	0.188	0.184	0.139	0.139
40%	0.196	0.223	0.210	0.196	0.149	0.139
50%	0.222	0.244	0.223	0.204	0.161	0.139
60%	0.237	0.268	0.232	0.223	0.172	0.148
70%	0.3	0.316	0.268	0.251	0.184	0.149
75%	0.335	0.349	0.299	0.268	0.185	0.149
80%	0.407	0.398	0.335	0.3	0.198	0.160
90%	0.665	0.612	0.574	0.527	0.236	0.171
95%	1.031	0.925	0.833	0.770	0.268	0.184
99%	1.975	1.754	1.957	1.721	0.284	0.210

Table D-3: Seasonal summary statistics for N filter inflow and outflow – April-June. Data for period May 2015 to April 2017.

Statistic	April		May		June	
	Inflow (L/s)	Outflow (L/s)	Inflow (L/s)	Outflow (L/s)	Inflow (L/s)	Outflow (L/s)
N of Cases	720	720	744	744	720	720
Minimum	0.107	0.111	0.237	0.223	0.390	0.352
Maximum	2.177	1.919	10.164	11.723	4.378	3.436
Median	0.139	0.151	1.104	1.031	0.690	0.612
Arithmetic Mean	0.384	0.365	1.432	1.301	0.984	0.876
Standard Deviation	0.436	0.386	1.297	1.171	0.736	0.652
Percentiles (Cleveland method)						
1%	0.109	0.115	0.237	0.223	0.390	0.352
5%	0.109	0.118	0.252	0.230	0.429	0.370
10%	0.109	0.119	0.284	0.267	0.471	0.415
20%	0.113	0.119	0.370	0.335	0.569	0.515
25%	0.119	0.126	0.409	0.389	0.586	0.531
30%	0.119	0.128	0.493	0.471	0.628	0.562
40%	0.125	0.137	0.914	0.865	0.658	0.586
50%	0.139	0.151	1.104	1.031	0.690	0.612
60%	0.186	0.207	1.370	1.248	0.727	0.663
70%	0.409	0.372	1.645	1.514	0.805	0.730
75%	0.539	0.499	1.986	1.838	1.8	0.820
80%	0.655	0.606	2.210	2.052	1.290	1.065
90%	1.021	0.948	3.023	2.822	2.063	1.890
95%	1.358	1.216	4.234	3.510	2.674	2.461
99%	1.963	1.766	5.067	4.401	3.939	3.352

Table D-4: Seasonal summary statistics for N filter inflow and outflow – July-September. Data for period May 2015 to April 2017.

Statistic	July		August		September	
	Inflow (L/s)	Outflow (L/s)	Inflow (L/s)	Outflow (L/s)	Inflow (L/s)	Outflow (L/s)
N of Cases	744	744	744	744	720	720
Minimum	0.283	0.268	0.252	0.259	0.210	0.3
Maximum	4.836	3.436	4.707	3.436	0.879	0.835
Median	0.930	0.882	0.374	0.361	0.252	0.252
Arithmetic Mean	1.142	1.162	0.681	0.622	0.292	0.299
Standard Deviation	0.889	0.884	0.722	0.614	0.126	0.119
Percentiles (Cleveland method)						
1%	0.292	0.284	0.259	0.265	0.210	0.3
5%	0.306	0.284	0.268	0.268	0.210	0.237
10%	0.317	0.285	0.284	0.284	0.223	0.237
20%	0.335	0.317	0.3	0.3	0.223	0.249
25%	0.352	0.317	0.306	0.3	0.223	0.252
30%	0.370	0.335	0.317	0.313	0.228	0.252
40%	0.390	0.357	0.345	0.335	0.237	0.252
50%	0.930	0.882	0.374	0.361	0.252	0.252
60%	1.165	1.630	0.471	0.429	0.253	0.267
70%	1.723	2.181	0.586	0.539	0.284	0.284
75%	1.867	2.181	0.663	0.586	0.284	0.3
80%	2.095	2.181	0.865	0.775	0.3	0.317
90%	2.424	2.181	1.412	1.251	0.429	0.410
95%	2.549	2.291	2.187	2.032	0.583	0.563
99%	3.806	3.192	4.130	3.364	0.843	0.805

Table D-5: Seasonal summary statistics for N filter inflow and outflow – October-December. Data for period May 2015 to April 2017.

Statistic	October		November		December	
	Inflow (L/s)	Outflow (L/s)	Inflow (L/s)	Outflow (L/s)	Inflow (L/s)	Outflow (L/s)
N of Cases	744	744	12	12	1488	1488
Minimum	0.181	0.217	0.180	0.196	0.139	0.158
Maximum	2.236	1.919	1.534	1.398	0.930	0.927
Median	0.210	0.260	0.237	0.268	0.237	0.268
Arithmetic Mean	0.336	0.359	0.310	0.332	0.266	0.288
Standard Deviation	0.309	0.264	0.219	0.194	0.153	0.143
Percentiles (Cleveland method)						
1%	0.184	0.222	0.184	0.203	0.139	0.161
5%	0.189	0.230	0.184	0.210	0.149	0.170
10%	0.196	0.237	0.196	0.227	0.149	0.171
20%	0.196	0.237	0.207	0.246	0.163	0.184
25%	0.196	0.237	0.210	0.252	0.171	0.184
30%	0.196	0.241	0.210	0.252	0.171	0.186
40%	0.210	0.252	0.224	0.268	0.196	0.220
50%	0.210	0.260	0.237	0.268	0.237	0.268
60%	0.223	0.268	0.252	0.284	0.252	0.293
70%	0.269	0.3	0.268	0.3	0.268	0.306
75%	0.3	0.343	0.283	0.311	0.275	0.317
80%	0.370	0.394	0.3	0.323	0.3	0.335
90%	0.651	0.612	0.506	0.508	0.429	0.429
95%	0.865	0.835	0.746	0.741	0.618	0.604
99%	1.977	1.754	1.449	1.325	0.891	0.865

Appendix E N-filter summary statistics – water quality analyses

Table E-1: Summary statistics for key nitrogen species concentration data - N filter inflow and outflow.

Statistic	Inflow			Outflow		
	Ammoniacal-N (µg/L)	Nitrate-N (µg/L)	Total N (µg/L)	Ammoniacal-N (µg/L)	Nitrate-N (µg/L)	Total N (µg/L)
N of Cases	61	133	133	38	66	66
Minimum	1	1	350	1	5	306
Maximum	66	7840	9050	224	3780	4650
Median	36	2190	2495	77.5	421.5	1105
Arithmetic Mean	30.34	2290.04	2668.31	76.08	998.21	1571.62
Standard Deviation	16.55	731.57	838.7	51.70	1149.09	1040.9
Percentiles (Cleveland method)						
1%	1	1	374.9	1	5	327.44
5%	1.55	1741.5	2071.5	6	7	554
10%	5	1860	2220	7.3	8.1	622.2
20%	16	2010	2301	16.1	13.4	680.7
25%	17	2030	2330	33	16	714
30%	18	2058	2374	46.4	22.3	747.2
40%	19.9	2137	2440	63.6	160.2	891.3
50%	36	2190	2495	77.5	421.5	1105
60%	38	2243	2573	90.3	754.5	1493
70%	41	2290	2686	1.9	1664	2184
75%	42	2332.5	2840	110	2110	2480
80%	43.9	2399	29	125.6	2536.5	2894.5
90%	50	2960	34	133	2799.5	3110
95%	54.9	3158.	3727	151.4	2903	3263
99%	65	5275	6145	224	3646.4	4451.6

Table E-2: Summary statistics for key water quality variables - N filter inflow.

Statistic	Inflow proportion of TN (%)		Outflow proportion of TN (%)	
	Nitrate-N	Ammoniacal-N	Nitrate-N	Ammoniacal-N
N of Cases	133	58	66	37
Minimum	26.3	0.1	0.5	0.2
Maximum	95.5	2.8	92.4	26.6
Median	89.5	1.5	47	4.8
Arithmetic Mean	85.38	1.33	44.81	8.74
Standard Deviation	10.31	0.7	36.93	8.21
Percentiles (Cleveland method)				
1%	28.21	0.1	0.5	0.2
5%	68.46	0.18	0.6	0.2
10%	73.68	0.5	1.11	0.3
20%	80.02	0.6	1.60	1.05
25%	82.3	0.7	2.2	1.4
30%	83.98	0.79	3.32	2.0
40%	85.87	1.11	26.47	2.93
50%	89.5	1.5	47	4.8
60%	90.8	1.7	63.94	10.28
70%	91.26	1.8	79.54	15.74
75%	91.65	1.9	83.9	16.32
80%	92.1	1.99	87.73	16.85
90%	93.02	2.2	90.59	19.88
95%	93.58	2.36	91.12	22.35
99%	95.33	2.78	92.25	26.6

Table E-3: Summary statistics for N filter ammoniacal-N inflow and outflow loads and removal efficacy.

Statistic	Ammoniacal-N load (g/d) Inflow (BS model)	Ammoniacal-N load (g/d) Outflow (BS model)	Removal efficacy (%)
N of Cases	11904	11904	
Minimum	0.414	0	100
Maximum	2.772	6.322	-128.1
Median	0.592	1.838	-210.5
Arithmetic Mean	0.698	2.025	-190.1
Standard Error of Arithmetic Mean	0.003	0.005	
Mode	0.548	1.801	
95.0% LCL of Arithmetic Mean	0.693	2.014	
95.0% UCL of Arithmetic Mean	0.703	2.035	
Standard Deviation	0.283	0.58	
Percentiles (Cleveland method)			
1	0.418	1.409	-237.1
5	0.446	1.463	-228
10	0.475	1.517	-219.4
20	0.516	1.625	-214.9
25	0.526	1.659	-215.4
30	0.533	1.697	-218.4
40	0.562	1.789	-218.3
50	0.592	1.838	-210.5
60	0.629	1.907	-203.2
70	0.696	2.034	-192.2
75	0.765	2.16	-182.4
80	0.848	2.333	-175.1
90	1.095	2.839	-159.3
95	1.359	3.454	-154.2
99	1.693	3.99	-135.7

Table E-4: Summary statistics for N filter nitrate-N inflow and outflow loads and removal efficacy.

Statistic	Nitrate-N load (g/d)		Removal efficacy (%)
	Inflow (AMLE model)	Outflow (Reg. model)	
N of Cases	11904	11894	
Minimum	21.27	0.258	98.8
Maximum	5056.5	53462.541	-957.3
Median	47.068	2.902	93.8
Arithmetic Mean	110.191	78.567	28.7
Standard Error of Arithmetic Mean	1.816	6.305	
Mode	39.391	2.473	
95.0% LCL of Arithmetic Mean	106.63	66.207	
95.0% UCL of Arithmetic Mean	113.751	90.926	
Standard Deviation	198.171	687.643	
Percentiles (Cleveland method)			
1	21.758	0.352	98.4
5	24.898	0.483	98.1
10	28.554	0.631	97.8
20	34.344	1.092	96.8
25	35.799	1.287	96.4
30	36.873	1.554	95.8
40	41.748	2.365	94.3
50	47.068	2.902	93.8
60	54.553	3.891	92.9
70	69.97	6.501	90.7
75	89.222	10.561	88.2
80	116.976	19.541	83.3
90	240.16	92.793	61.4
95	460.085	438.604	4.7
99	928.678	1378.708	-48.5

Table E-5: Summary statistics for N filter -total N inflow and outflow loads and removal efficacy.

Statistic	Total-N load (g/d)		Removal efficacy (%)
	Inflow (AMLE model)	Outflow (AMLE model)	
N of Cases	11838	11828	
Minimum	24.056	7.558	68.6
Maximum	6208.9	5975.9	3.8
Median	55.569	28.526	48.7
Arithmetic Mean	120.912	80.658	33.3
Standard Error of Arithmetic Mean	2.052	1.597	
Mode	42.967	26.132	
95.0% LCL of Arithmetic Mean	116.889	77.527	
95.0% UCL of Arithmetic Mean	124.935	83.789	
Standard Deviation	223.299	173.696	
Percentiles (Cleveland method)			
1	24.458	8.966	63.3
5	28.148	10.668	62.1
10	30.762	12.357	59.8
20	36.183	16.695	53.9
25	39.623	18.268	53.9
30	42.412	20.155	52.5
40	47.82	25.226	47.2
50	55.569	28.526	48.7
60	63.624	33.494	47.4
70	77.776	42.558	45.3
75	97.117	55.705	42.6
80	125.83	77.046	38.8
90	246.074	175.54	28.7
95	497.538	400.09	19.6
99	1016.536	826.34	18.7

Table E-6: Summary statistics for automatically measured and laboratory measured electrical conductivity - N filter inflow.

Statistic	Inflow Electrical conductivity (µS/cm)	Grab sample electrical conductivity (µS/cm)
N of Cases	99	37
Minimum	159	158
Maximum	242	229
Median	178	185
Arithmetic Mean	185.73	184.34
Standard Deviation	20.4	18.526
Percentiles (Cleveland method)		
1%	160	158
5%	162	158.7
10%	165	162.2
20%	167	164.8
25%	168	166
30%	168	169.2
40%	172.1	173.2
50%	178	185
60%	187	194.7
70%	206	198.2
75%	207	199.75
80%	207	202.55
90%	208	204.4
95%	216.05	206.95
99%	240.53	229

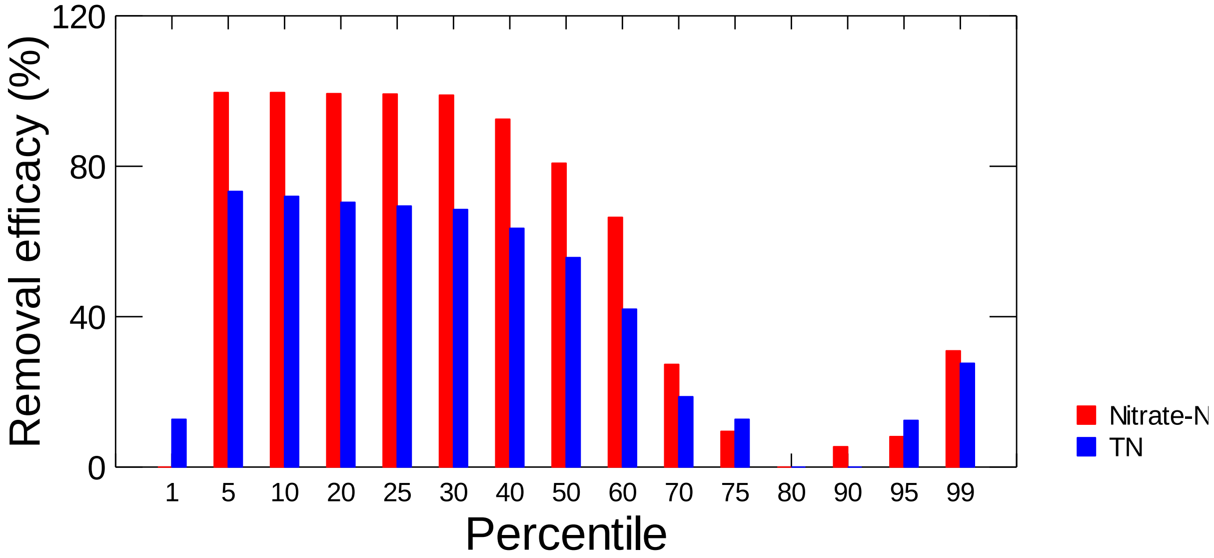

Figure E-1: Removal efficacies for nitrate-N and TN in terms of grab sample concentration percentiles. This figure indicates that removal efficacy decreases with TN and nitrate-N concentration. This behaviour may be an artefact of the relationship between flow, load and retention time.

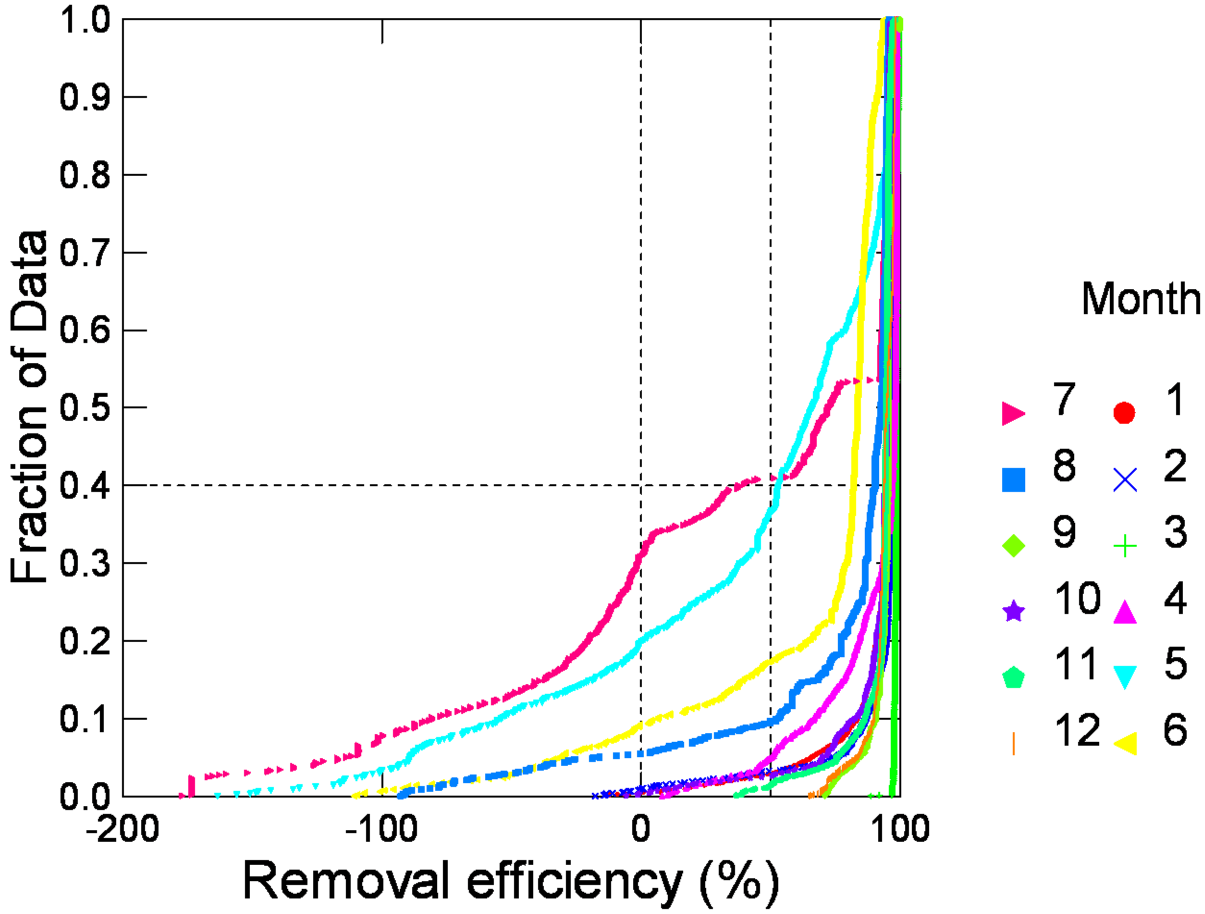

Figure E-2: Monthly nitrate-N removal efficacies. The vertical dashed lines indicate zero and 50% net nitrate-N removal respectively, and the horizontal dashed line indicates the 40th percentile value.

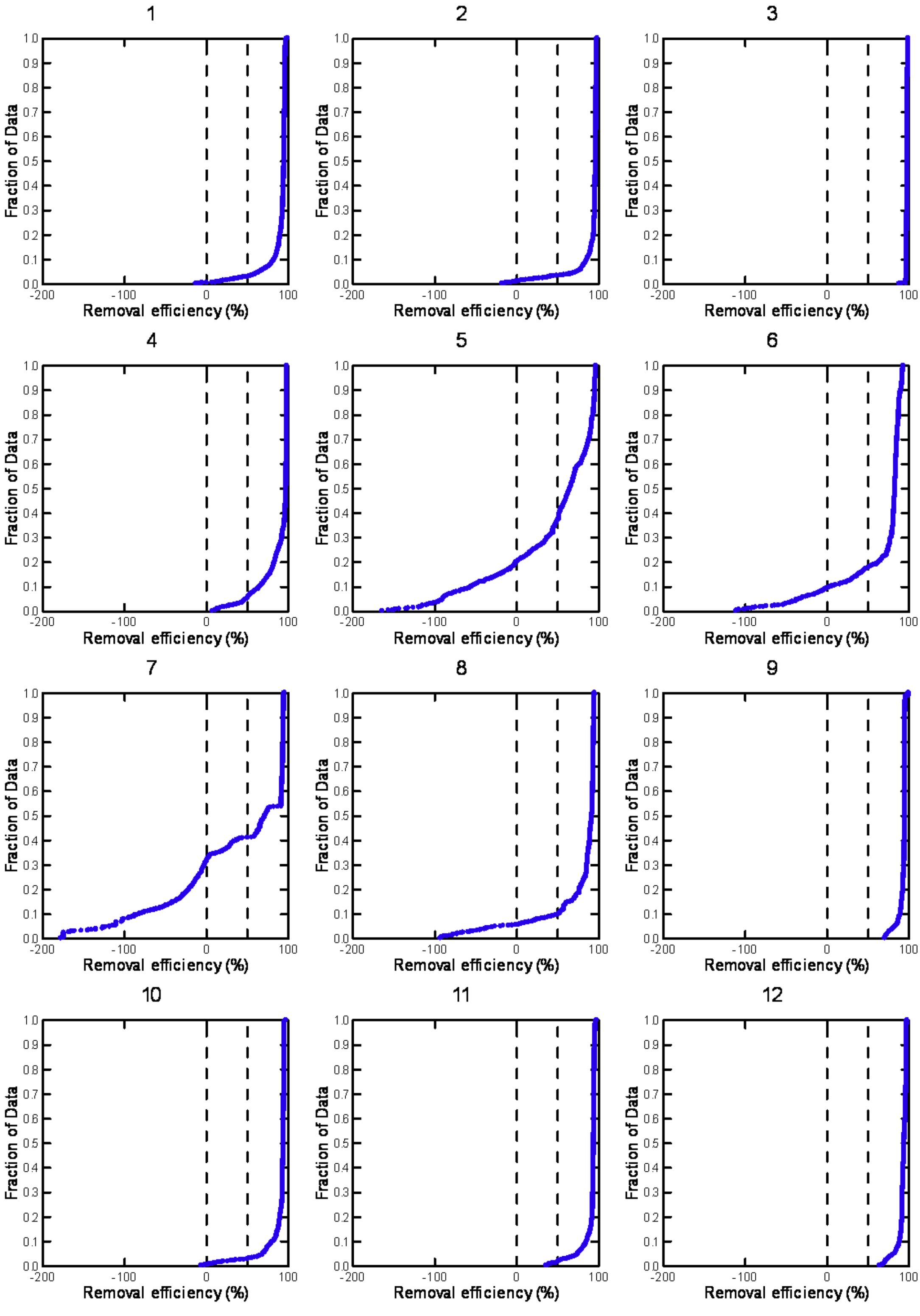

Figure E-3: Monthly nitrate-N removal efficacies. The vertical dashed lines indicate zero and 50% net nitrate-N removal respectively, and the horizontal dashed line indicates the 40th percentile value.

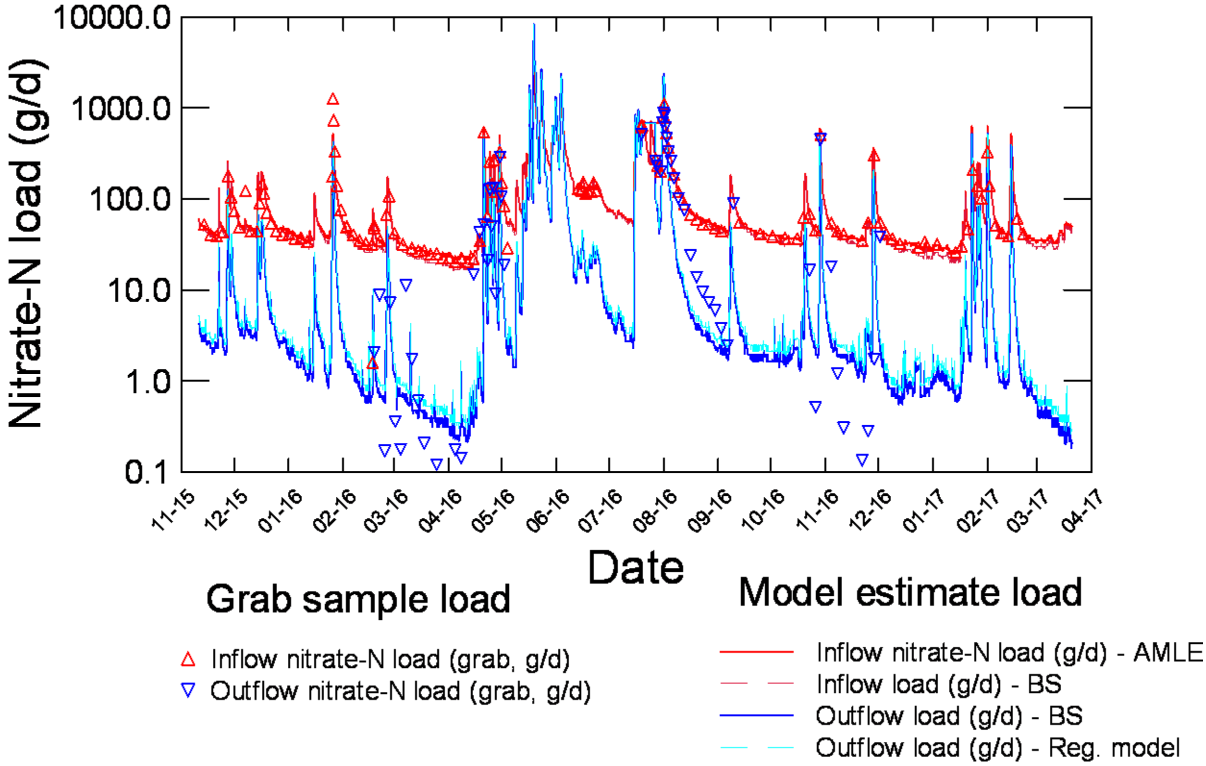

Figure E-4: Comparison of measured and modelled nitrate-N load estimates – alternate models.