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BIBLIOGRAPHIC REFERENCE

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EXECUTIVE SUMMARY

GNS Science was commissioned by Environment Southland to report on the suitability of in-situ denitrifying bioreactors and phosphorous (P)-sorbent filters for removing nitrate (NO_3^-) and P from tile drain discharge in the Southland region. Tile drains have been used in Southland for over a century as an important drainage tool in the conversion of swamps and wetlands into quality pastures. However, tile drains also work as direct conduits for nutrients to open waterways, particularly NO_3^- and P. This results in eutrophication and other negative environmental effects. Denitrifying bioreactors and P-sorbent filters have been used successfully worldwide to reduce nutrient loads to water resources, including open waterways and drains.

Briefly, the objectives of this study were:

- A literature review;
- The compilation of region specific tile drain N and P discharge;
- A site visit;
- The development of design options; and
- The development of an optimal pilot study design.

Denitrifying bioreactors (also known as denitrification beds) are trenches filled with a carbon energy source (e.g., woodchips) that remove NO_3^- through the enhancement of the natural denitrification process. P-sorbent filters contain media with high concentrations of Ca, Fe and/or Al (e.g., limestone gravels, crushed oyster shells) and remove P through precipitation and absorption processes. Denitrifying bioreactors and P-sorbent filters are cost-effective systems that can remove significant percentage (> 95%) of NO_3^- and P, and have lifespans of up to several decades if designed correctly. A bioreactor and filter design needs to be tailored to accommodate site-specific conditions that include the range of flow rates, influent NO_3^- and P concentrations, temperatures, hydrology, and removal goals. Correct installation, operation and maintenance of these systems are required to obtain maximum nutrient removal and mitigate any potential negative effects.

Limited information exists for tile drain discharge and nutrient concentrations in the Southland region. However, from the available information, tile drain discharge is estimated to be between 21-100% of annual rainfall, which is 1127 mm for Southland. Average tile drain discharge is estimated to be 55% of total rainfall. NO_3^- -N and P concentrations of sampled tile drain water ranged between 0.002 - 67 mg L⁻¹ and 0.004 - 2.4 mg L⁻¹, respectively.

Several design options were considered for the pilot trial design, that accommodated various substrates, flow rates, range of nutrient concentrations and tile drainage areas. The design goal was to remove as close to 100% of the N and P as possible from a component of tile drain flow. The final proposed pilot trial was designed to treat a 5 ha drainage area, with 'hotspot' nutrient concentrations of 5 mg L⁻¹ NO_3^- -N and 0.3 mg L⁻¹ P (75 percentile concentrations), at a flow rate of 85 m³ d⁻¹. It is proposed to use *Pinus radiata* woodchips as a carbon source for the bioreactor, and heat treated, crushed oyster shells as P-sorbent filter substrate. These materials were selected due to the local availability, performance, longevity and cost-effectiveness. An overflow bypass weir and line must be installed to prevent hydraulic overload of the system. Two monitoring regimes were proposed, with preference depending on available funding and final decision on the objectives of the pilot trial, which will be conducted in future work.

1.0 INTRODUCTION

GNS Science was contracted to Environment Southland (ES) through a Ministry for Science and Innovation (MSI) Envirolink Medium Advice Grant (1112-ESRC252). The aim of this project was to report on the suitability of in-situ denitrifying bioreactors and phosphorous-sorbent filters for removing nitrate (NO_3^-) and phosphorous (P), from tile drain discharge in the Southland region. The Grant was submitted to MSI by George Ledgard on behalf of ES. This report presents the findings of GNS Science and co-authors for the above contracted work including: a literature review of denitrification beds, P-sorbent filters and tile drain hydrology in Southland; a site visit; design of a pilot-trial bioreactor and filter to suit a typical Southland tile drain setting; and a recommended monitoring programme for the pilot trial.

1.1 What is the issue?

Tile drains have been an important drainage tool for farmers in Southland over the past century. They have been integral in the success of Southland as a productive farming region and have allowed large tracts of previously swampy and boggy land to be developed into quality pasture. However, tile drains also act as a direct conduit of soluble nutrients from the pasture to open waterways. In particular, nitrogen (N) and phosphorous (P) nutrients are elevated with the intensification of farming practices, and tile drains contribute a significant mass of N and P to waterways (Monaghan 2000; Monaghan and Smith 2004; Jaynes et al., 2008; Woli et al., 2010). An accumulation of N and P in waterways can contribute to hypoxia, eutrophication, loss of biodiversity, and habitat degradation of these environments.

Although nutrient losses can be minimised through a number of on-farm agricultural management practices (Dinnes et al., 2002; Jaynes et al., 2008), these practices often require 'precision' agricultural techniques that are typically out of scope under the current farming systems in Southland. As an alternative, mitigation of nutrient pollution of waterways can be obtained through the installation of in-situ tile drain bioreactors and filters. These techniques have proved to be successful overseas and in New Zealand (Kaspar et al., 2003; Jaynes et al., 2008; McDowell et al., 2008; Woli et al., 2010), and have the potential to be implemented in Southland to address the issue of N and P leaching. Specifically, this would entail installing bioreactors containing organic carbon (C) for the biological conversion of $\text{NO}_3\text{-N}$ to di-nitrogen gas (N_2) (denitrification), and filters containing P-sorbent material (e.g., wood chips, oyster shells) for the removal of P. Bioreactors and filters would be installed at the discharge end of tile drainage networks.

1.2 Project objectives and proposed future work

The objectives of this project were to:

- review state of current knowledge of N and P removal using bioreactors and filters, including media, removal rates and design (Sections 2.1 and 2.2);
- compile existing information on typical tile drain flow rates and $\text{NO}_3\text{-N}$ and P concentrations for the Southland region (Section 3.0);
- conduct a site visit (Stewart Cameron) to the Southland region to view hydrology of typical tile drain networks, meet with regional council staff and landowners to

ascertain if a bioreactor and filter pilot trial was feasible. Stewart Cameron visited Southland region on 27-28 June 2012 for this purpose;

- if a pilot trial was deemed feasible, develop an appropriate bioreactor and filter design based on anticipated tile drain flow rates, NO_3^- N and P concentrations, and tile drain hydrology (Section 4.0); and
- provide a recommended pilot tile design for future in-situ testing of an N-bioreactor and P-filter performance (Section 5.0).

If the outcome of this project indicated that a pilot trial was deemed feasible, ES indicated that the following additional work would be planned in subsequent projects:

- seek funding from internal and/or external sources to undertake this trial;
- if the trial is successful, expand installation to a regional scale with industry buy-in; and
- report findings in local media and incorporate into a toolbox and best practice advice if proven successful.

2.0 LITERATURE REVIEW OF SUITABLE METHODS FOR REMOVAL OF NITRATE AND PHOSPHOROUS FROM TILE DRAIN DISCHARGE USING PASSIVE BIOREACTOR AND FILTER TECHNOLOGIES

In New Zealand, N and P loss from farming activity to water bodies is considerable due to N fixation by pasture legumes, application of N and P fertilizer in excess of plant demand and high stocking rates (Parfitt et al., 2006; Ballentine and Tanner 2010). Leaching of excess N and P beneath farmland has caused elevated concentration of these nutrients in groundwater, eutrophication of surface waterways and algal blooms in nationally significant lakes and waterways (Larned et al., 2004; Vant and Smith, 2004).

Improved crop varieties and farm management practices hold promise for reducing N and P leaching (Cassman et al., 2002). For example, on farm management strategies can be the most cost effective method of reducing P loss, with costs estimates of \$0-200 kg of P conserved (McDowell and Nash 2012). However, N fertilizer use in New Zealand increased by almost 700% from 1989 to 2003 (Parfitt et al., 2006), and despite implementation of management strategies to reduce P losses (e.g., balancing P inputs and outputs and maintaining optimum Olsen P concentration in the soil), leaching still occurs in areas of wet soils, areas abutting streams, soils with limited P retaining capacity and in tile drained land. Reactive forms of N and P are soluble so that additional measures are needed to remove these nutrients after they leave the field and begin to flow through a catchment. In farmlands, nutrient sink areas (e.g., wetlands) have often been removed and/or bypassed by tile drains, which provide a direct conduit for transporting leached N and P from fields to streams (David et al., 2010).

Where natural nutrient sinks are absent, artificial sinks, such as carbon bioreactors (simple, wood-chip filled trenches; Schipper et al., 2010a) and P-sorbent filters (containments filled with porous materials rich in Ca, Al, and/or Fe; Vlahos et al., 1989; O'Reilly and Sims 1995; Haustein et al., 2000; Stout et al., 2000) are an effective method for reducing edge-of-field NO_3^- and P losses. The bioreactor and filter are positioned to intercept drainage and/or groundwater. Bioreactors are only used where edge-of-field N losses are dominated by NO_3^- as they do not remove other forms of N (Gentry et al., 1998; Goolsby et al., 1999; Nolan, 2001). There is evidence that P-filters are able to remove numerous forms of P (e.g., Soluble Reactive Phosphorous (SRP), Dissolved Reactive Phosphorus (DRP)) depending on the filter material (McDowell et al., 2008).

Bioreactors and filters are starting to be employed in an array of climatic, geophysical and agricultural settings. Field testing of bioreactors and filters has shown that these systems can be integrated into different land uses if designed to meet site-specific conditions. However, the success of bioreactors and filters requires informed adoption and placement. There are emerging general design principles and best management practice (BMP) guidelines for specific spatial/geographic conditions, climate regimes and agricultural practices. The following sections discuss relevant information on removal of NO_3^- using bioreactors, and removal of P-species using in-situ filters.

2.1 Removal of nitrate using denitrifying bioreactors

Denitrifying bioreactors are being used increasingly to reduce nitrate (NO_3^-) loads into receiving waters (Schipper et al., 2010a). Their design is based upon the principle of enhancing natural removal of NO_3^- from water via the denitrification process (e.g., wetlands, soils, sediments). Denitrification is the microbial conversion of NO_3^- -N to N gases under anaerobic conditions in the presence of a carbon (C) energy source (Seitzinger et al., 2006). In bioreactors, the C energy source is typically wood chips, although other materials such as corn cobs have been used (Cameron and Schipper 2010; 2011). In water-saturated conditions, such as within a bioreactor, the C source degrades slowly, creating anaerobic conditions and labile C that stimulates denitrification.

Two types of bioreactors have been developed and tested – denitrification beds and denitrification walls (Schipper et al., 2010a). A denitrification bed (Figure 2.1) is a lined cavity or container filled with a particulate organic C matter through which nitrate-bearing water is passed. Beds are used to treat nitrate-rich discharges such as tile drainage or effluents (e.g., Woli et al., 2010; Schipper et al., 2010b) but have also been installed in the base of streams (Robertson and Merkely 2009). A denitrification wall (Figure 2.2) is a trench of soil excavated perpendicular to the groundwater flow direction and back-filled with a wood chip/soil mix. The walls act to treat the nitrate-bearing groundwater before it flows into surface water or tile drains (e.g., Schipper et al., 2005; Jaynes et al., 2008; Moorman et al., 2010). A key requirement of bioreactor design is that it needs to be tailored to accommodate site-specific conditions that include the range of temperatures, flow rates, discharge points, NO_3^- concentrations and NO_3^- removal goals found at sites in different settings and regions.

2.1.1 Performance of denitrifying bioreactors

The performance of a bioreactor can be described by the percentage removal of NO_3^- between influent and effluent water flow. However, percentage removal does not take into account the volume of C incorporated in the bioreactor. For comparative purposes, it is more useful to describe denitrification performance as a reaction rate using mass of N removed per unit volume of organic media per unit time (e.g., $\text{g N m}^{-3} \text{d}^{-1}$). Such units are scalable and can be used for bioreactor design when influent volumes and concentrations are known. The rate also allows comparison of NO_3^- removal performance between different bioreactor designs and studies.

Performance monitoring of bioreactors has been undertaken in laboratory-, pilot- and field-scale studies. Laboratory studies generally focus on the comparison of the hydraulic performance and NO_3^- removal efficiency of different C sources (e.g., Gibert et al., 2008; Greenan et al., 2006; Cameron and Schipper 2010). Field-scale studies have generally been for “proof of concept” within agricultural landscapes or wastewater treatment. To date, field scale beds have used varying fractions and grain-sizes of wood particles as the carbon source. These wood-based beds are capable of sustained NO_3^- removal and hydraulic performance for more than 15 years (Robertson et al., 2010). The majority of field scale trials have been undertaken in Canada (Robertson et al., 2008; Robertson and Merkely 2009), New Zealand (Schipper et al., 2004; 2005; 2010b), Australia (Schipper et al., 2010a) and in several US States, including Iowa (Moorman et al., 2010), Illinois (Woli et al., 2010), Rhode Island (Addy and Gold 2008), and California (Leverenz et al., 2010). The wide range of

reported NO_3^- removal rates (2 to 32 g N m^{-3} d^{-1} in wood-based beds; 0.014 to 3.6 g N m^{-3} d^{-1} in walls) in field-based bioreactor studies reflect differences in C substrates, hydrologic setting, temperature, seasonal/site variation in influent NO_3^- -N concentration and hydraulic residence time (HRT) (Schipper et al., 2010a). Reported rates of NO_3^- removal can be conservative (less than the potential rate) if the bioreactor removes all incoming NO_3^- and is not accounted for in the monitoring regime. The reported higher NO_3^- removal rates (>10 g N m^{-3} d^{-1}) for operational wood based beds are from the start-up period of the first 6 to 10 months of operation (Blowes et al., 1994; Robertson et al., 2000; Schipper et al., 2010b) but are not sustained over the longer-term.

Table 2.1 provides a summary of long-term NO_3^- removal rates for a variety of bioreactors and C media. For beds containing wood materials, the long-term sustainable rate varies between approximately 2 to 10 g N m^{-3} d^{-1} for operating bed temperatures of 8 to 20°C and influent nitrate concentrations of 5 to 250 mg L^{-1} . Wood-based by-products sawdust and woodchip are the most commonly used C source in bioreactors. This is because these materials are commonly available, have a relatively low cost, support high permeability, have a high C:N ratio (e.g., 300:1; Robertson and Anderson, 1999), and have long durability. Wood based products do not support as high initial NO_3^- removal rates as more labile C sources, such as corn cobs, but they are likely to sustain removal for longer duration. The commercially available Nitrix filter used in bed trials by Robertson et al. (2005a), contained bark, sawdust and woodchips and provided removal rates of 2 to 5 g N m^{-3} d^{-1} , however, testing of this system was generally conducted under NO_3^- limiting conditions and maximum NO_3^- removal rates have the potential to be higher. While alternative and more labile C sources (e.g., maize cobs) may support higher denitrification rates (up to 23 g N m^{-3} d^{-1} for at least 2 years; Cameron and Schipper 2011), they may require more frequent replacement because of C depletion and decreased permeability in the bed as the C structure breaks down (Erikson et al., 1974; Fay, 1982). Similarly, removal rates for wheat straw are likely to be almost twice that of wood for several years but it will require more frequent replacement (Cameron and Schipper, 2010).

Reaction rates achieved in wall studies are generally an order of magnitude less than bed reaction rates, most likely due to the lower proportion of C media to wall matrix (typically 15–30% by volume), compared to 100% C media utilised in beds. Lower reaction rates achieved in column studies of wood media compared to operational beds are generally attributed to the presence of dissolved oxygen in the influent water that must first be removed by microbial activity before denitrification can occur (Tiedje, 1988).

Bioreactors containing C media of wood will remove NO_3^- for nine years or more (Robertson et al., 2008; Moorman et al., 2010; Long et al., 2011). Studies estimating future performance of bioreactors by examining decay rates of wood material in walls estimated C material half-lives of between 4.6 and 37 years depending on sample depth (Moorman et al., 2010) and 11 years (Long et al., 2011). Warneke et al. (2011) measured total C losses from a large bed and estimated a life time of up to 39 years. These studies suggest that NO_3^- removal in the bioreactor will be sustained for decades, although the efficiency is likely to decline over time.

Schipper et al. (2010a) estimated a removal cost of between US\$2.39-15.17 per kg of N for bioreactors based on functional life time of 20 years. This compared well with other agricultural management techniques for reducing N losses, such as controlled drainage,

wetlands and autumn cover crops. The cost estimate was calculated based on landowner constructed bioreactors using local materials and demonstrates the need for robust design criteria that are usable by the landowner.

2.1.2 Hydrologic Considerations

Incorporating the hydrologic conditions at an installation site is critical for the development of an appropriate bioreactor design. Walls are passive systems, restricted by construction constraints to installation of less than 1-2 meters below a shallow groundwater table. Wall bioreactors only treat groundwater that is intercepted. Figure 2.2 displays ideal treatment conditions in an aquifer with a shallow impermeable layer that causes all groundwater flow to intersect the wall. In deep aquifers, where the wall is not installed to the base of the aquifer, flow paths may go below the wall (Figure 2.3), which reduces the efficiency of the treatment. The hydraulic conductivity (K) of a wall versus the K of the surrounding aquifer may control groundwater flow paths and affect treatment performance. If the wall's K is lower than the surrounding media, which may occur if a fine-grained C media is used or if a labile C media's structure breaks down, flow will bypass the wall. Hydraulic efficiency can be maintained by using large wood chips without a decline in NO_3^- removal performance (Cameron and Schipper 2010) due to the large secondary porosity of wood chip material. Alternative C sources, such as maize cobs, can provide greater N removal, but there is concern that they may not support NO_3^- removal for as long and decline in hydraulic performance more quickly. Alternatively, walls with high K relative to the surrounding media have been found to induce groundwater upwelling (Figure 2.4), extending the treatment depth (Robertson et al., 2005b).

Beds are positioned to intercept channelized flow or tile drainage. Because NO_3^- removal in beds can be limited by HRT and NO_3^- load, sizing of the beds requires understanding of void space volume within the bed and temporal and spatial variation of inputs. Consideration of seasonal variation and rainfall event pulses of hydrologic and NO_3^- inputs also need to be incorporated in bed design to optimise cost versus performance.

2.1.3 Effect of temperature

Seasonal and annual temperature differences between regions are likely to account for some of the variability observed in bioreactor performance. Temperature is an important factor controlling the rate at which denitrification occurs (Sedlak, 1991). This is because enzyme activity generally increases with increasing temperature, typically over the range 0°C to about 40°C, due to increase in kinetic energy of molecules. Enzyme activity begins to decline above an optimum temperature limit as enzymes start to denature. Generally, the change in reaction rate with change in temperature can be approximated by the temperature coefficient (Q_{10}) which is the factor by which a reaction rate increases for every 10°C rise in temperature (Anderson and Janssen, 2006).

In the Schipper et al. (2010a) review of non-nitrate limiting bioreactors, a general trend of increasing denitrification rates with increasing average annual temperature was noted. In an as yet unpublished later review by Schipper (2012) of denitrification bed studies that incorporated a temperature component, it was found that removal rates increased from approximately 0.5 to 2.5 g N m³ d⁻¹ at 1 - 3°C, up to 4 to 12 g N m³ d⁻¹ at 22°C (Figure 2.5).

Alternatively, some studies suggest that long-term N removal may be lowered in warmer climates since the C substrate may decompose more rapidly (Cameron and Schipper 2010).

While many studies of denitrification beds have acknowledged the temperature dependence of denitrification reaction rates, few studies have quantified the temperature-reaction rate relationship in field conditions. The variability of other factors that influence nitrate removal (e.g., influent nitrate concentration and chemistry) makes it difficult to isolate the effect of temperature under field conditions. Canadian-based studies (van Driel et al., 2006a; Robertson et al., 2000) reported a range of denitrification rates achieved in denitrification beds for water temperatures up to 22°C. The removal rates reported in these studies increased from 2–7 g N m⁻³ d⁻¹ at 2–5°C up to 4–30 g N m⁻³ d⁻¹ at 10–22°C. However, in the study of van Driel et al. (2006a), precise calculation of nitrate removal rates at temperatures >12°C were compromised, as the denitrification processes had removed all of the available nitrate.

Temperature dependence of denitrification rate is also indicated in denitrification wall studies. Schipper and Vojvodic-Vukovic (2001) reported reaction rates of 0.014 to 0.43 g N m⁻³ d⁻¹ for temperature range of 11°C to 22°C. In comparison, Fahrner (2002) measured higher removal rates of 15 g N m⁻³ d⁻¹ when soil temperatures regularly exceeded 30°C.

2.1.4 Adverse environmental effects

Adverse effects of denitrification beds have been identified, including: leaching of biochemical oxygen demand (BOD), ammonium (NH₄) and organic-nitrogen; and production of greenhouse gases (GHG), methyl mercury (CH₃Hg), nitrous oxide (N₂O), methane (CH₄) and hydrogen sulphide (H₂S).

BOD, NH₄ and organic-nitrogen

The leaching of BOD, NH₄ and organic-nitrogen into bed effluent can be problematic depending on the discharge environment.

Dissolved labile C leaching from the bioreactor causes elevated BOD in the bed discharge water. This can affect dissolved oxygen concentrations of receiving waters. The BOD is an important test used in protecting aquatic life from oxygen deficiency. It is the primary regulatory tool used in limiting discharge of wastewaters to water (McCutcheon et al., 1993). Water quality guidelines used in New Zealand (Ministry of Health, 2005) limit BOD₅ to <5 g m⁻³ to protect against nuisance bacterial slime growth (Davis-Colley and Wilcock, 1994). Denitrification beds typically have high BOD (>100 mg L⁻¹) during start-up and for up to several months thereafter as the soluble organic constituents (tannic acids etc.) are leached from the reactive media (Robertson et al., 2005a; Schipper et al., 2010a). As these soluble constituents comprise only 1% to 2% of the wood mass (Robertson et al., 2005a), BOD subsequently stabilises at much lower values (10 to 40 mg L⁻¹) after several months operation depending on factors such as HRT, temperature, and wastewater characteristics. It is likely that use of more labile carbon sources in denitrification beds for the purpose of increasing nitrate removal rate will result in higher rates BOD leaching during the first months of operation. Careful start-up procedures may be able to minimize losses of dissolved C, such as pre-leaching of wood chips.

Similarly, denitrification beds have elevated NH_4 and organic-nitrogen during the start-up period as these mobile compounds are flushed from the C media. However, concentrations decline to less than environmental guideline values (ANZECC, 2000) within two months of operation (Cameron and Schipper, 2010).

Management practices could be employed to reduce leaching of BOD, NH_4 and organic-nitrogen. For example: commissioning beds during the winter when leaching rates are reduced due to slower decomposition of the carbon media at lower bed temperatures; reducing HRT at start-up to decrease the dissolution concentrations of carbon from the media; pre-leaching of media; or recirculation. However, the effectiveness of these practices has not been tested.

GHG

Production of N_2O and CH_4 (both potent greenhouse gases) and CH_3Hg is from either incomplete denitrification or prolonged retention times that promote highly reducing conditions within the bioreactor. Controlling these pollutants depends on scaling bioreactors to ensure that the N concentrations within the bioreactor are appropriate to ensure complete denitrification while not allowing methanogenesis (Warneke et al., 2011) or sulphate reducing conditions (Shih et al., 2011).

Techniques to maintain gases in solution are needed to encourage complete denitrification to N_2 to resolve this issue. Integration of bed covers to maintain gas emissions in solution with the solar heating of beds is one possible technique that could be explored. Bioreactors are currently being developed to reduce H_2S emissions from geothermal power plant discharges (Bacon, 2011; Rattanapan et al., 2009). Adaptation of this technology could potentially be used to reduce H_2S emissions from denitrification bed discharge.

Sulfate reducing environments in natural wetlands and lakes are known to promote the methylation of inorganic mercury (Gilmour et al., 1992; Eckley and Hintelmann, 2006; Todorova et al., 2009). Methylation allows Hg to bio-accumulate in the food chain with potentially detrimental consequences. Sulfate reducing conditions occur in denitrification beds when nitrate removal is complete (Robertson and Merkley, 2009; Elgood et al., 2010).

2.2 Removal of phosphorous using P-sorbent filters

The loss of P through soil processes in tile drained land with an agricultural land use is essentially inevitable. However, recent applications of in-situ filters containing P-sorbing material have been effective at reducing P concentrations in runoff and groundwater (McDowell et al., 2008). A P-sorbent filter can be positioned in a location to intercept water flow from tile drainage (Figure 2.6). Material within the filter then acts to remove various forms of P from the water through adsorption and chemical precipitation. Studies on the detrimental effects of excess P in the aquatic environment generally consider dissolved reactive P (DRP) or total P (TP). DRP is the most mobile and abundant form in the agricultural landscape. Studies on P filters have shown they are able to remove forms of P other than DRP (McDowell et al., 2012) although the removal mechanisms are somewhat dependent on the form of P (i.e., ortho-P, poly-P or organic-P). Therefore, it is appropriate to

consider TP concentrations of the influent water for filter design and when determining performance of the filter.

There are many factors to consider when designing a P-filter, and in particular substrate material has a considerable influence on filter performance. More than 20 containment materials have previously been used in P-filters, including natural materials, industrial by-products and man-made materials (Vohla et al., 2011). Phosphorous removal by filters is closely associated with the physical-chemical and hydrological properties of the filter media because precipitation and absorption processes dominate. P removal by biomass (i.e., algae) assimilation also occurs, although it is typically a minor component in filters. The important constraints in filter design are P-sorption capacity, longevity of the P-sorption capacity, particle size, hydraulic conductivity availability and cost of the filter media. In addition, adverse environmental effects, including environmental toxicity from the substrate material are important to consider.

2.2.1 Performance of P-sorbent filters

The performance of the in-situ P-filter can be determined by the percentage of P-sorbed to containment material, and therefore removed from inflow water. P removal is generally described as a rate using mass of P removed per volume of filter material (e.g., g P kg⁻¹). Such units allow comparison between different media type, although specific gravity of the media needs to be considered for volumetric comparison. The performance of a P-filter is highly dependent on the type, size and porosity of media used, and the hydraulic loading rate and retention time.

An understanding of the influent P concentration is required during P-filter design and analysis of performance. On agricultural land, P losses of up to 1 kg P ha⁻¹ are common but this can increase to > 2 kg P ha⁻¹ where soils have limited capacity to sorb P (Sims et al., 1998). The ability of a soil to retain P can be limited in three primary ways: in organic-matter rich soils that block the sorption sites; in leached soils with little Fe and Al; or where P has been applied in excess of crop demand (McDowell et al., 2008). Many soils which have elevated levels of P also have increased P concentration in drainage waters, particularly over tile drain networks. The loss of P from soils in many locations is inevitable, especially under intensive farming and/or where the water table is shallow.

As there are a wide variety of potential materials and various methods used for determining the P-removal capacity, it is often difficult to compare studies from the literature (Vohla et al., 2011). In many cases, the P-removal rates of a particular material also vary greatly between studies. Potentially the most important consideration is that in some cases materials which prove to be highly efficient in the laboratory do not operate as well in field systems. Batch experiments provide an effective way to compare the P-removal ability of media, in particular column studies allow for the comparison using similar hydraulic conditions. Caution should be applied when basing a field design on the maximal results of laboratory or pilot studies as the P-removal ability could be overestimated due to differences in study design (e.g., duration, water/material ratio, influent water composition) (Drizo et al., 2002).

It is also extremely difficult to estimate the lifespan of a P-filter, as so much depends on site specific conditions (e.g., material, influent flow, pH conditions). In many cases, P-removal

using natural sands and gravels has been efficient for the first 6 - 24 months, after which retention is highly ineffective due to P-saturation of the media (Vohla et al., 2011). Premature loss of performance can occur if the media is saturated in a short time frame. In particular this occurs when the selected media does not contain enough sorbing ions (e.g., Ca, Al and Fe) to continue P-retention. In many cases, the retention ability of the material is significantly reduced after 5 years of operation, however this timeframe will be filter volume, material, grain-size, inflow P concentration and chemistry dependent.

Another potential issue for the P-filter performance arises from the clogging of material in the system. Influent water can contain high levels of suspended solids and organic material which can cause the pore spaces to be clogged, in return decreasing the hydraulic performance of the filter. This can be resolved through appropriate pre-treatment of the influent water if necessary, which in turn is likely to increase the lifespan of the filter media (Vohla et al., 2011).

2.2.2 P-filter materials

In a review of materials for P-filters by Vohla et al. (2011) categorised over eighty studies into three broad material groups: natural materials, industrial by-products and man-made materials. Comparison of the P-removal capacity of different filter media identified that industrial by-products had the highest removal rate of up to 420 g P kg⁻¹. However, not all of these materials are readily available in New Zealand. For the purposes of this review, a subset of 21 materials reviewed by Vohla et al. (2011) that are of relevance to the New Zealand context is provided in Table 2.2. Of these materials, six are of particular relevance for the Southland area (bolded) including: **bauxite, limestone, zeolite, modified zeolite, red mud (unheated) and oyster shells (heated)**.

Bauxite, commonly known as aluminium ore, is a naturally occurring mixture of minerals which are rich in hydrated Al and Fe oxides (Vohla et al., 2011). Studies have indicated that activated bauxites can remove higher concentrations of P compared to raw forms of bauxite (Altundogan and Tumen, 2003). In addition, studies found that ortho-phosphate removal was highest at a pH of 4.5. A potential local source of bauxite is the Tawai Aluminium Smelter near Bluff, where bauxite is used for aluminium production.

Limestone is a common sedimentary rock composed primarily (> 50%) of calcite minerals (CaCO₃), and is often formed in conjunction with other minerals, sand and silt. Limestone is formed in a depositional environment, primarily from calcium rich remains of organisms (e.g., shellfish, corals), and is therefore rarely pure in calcite. Due to its high P-adsorption properties, crushed or powdered limestone is often applied to agricultural soils to reduce acidity and increase P-retention (Ballantine and Tanner, 2010). P-filter experiments have indicated that a limestone rock filter system can remove from 20% (Strang and Wareham, 2006) to 46% (DeBusk et al., 2004) of TP. Furthermore, a retention capacity of 0.3 - 20 g P kg⁻¹ of limestone was reported by Johansson (1999a). Studies in New Zealand have indicated that a batch experiment in the laboratory had significantly higher P-removal than in the field (Shilton et al., 2005). In the same study, it was found that increased temperatures correlated with increased P-retention. Extensive deposits of limestone occur throughout New Zealand, and local limestone could be sourced from the Fernhill limeworks in Kauana, Winton. Care is required when selecting limestone material, as each limestone has unique

physical and chemical characteristics and therefore different P-retention properties (Ballantine and Tanner, 2010).

Zeolite is a mineral composed of hydrated aluminosilicates. There are currently 40 known frameworks that occur naturally, although zeolites can also be synthesised artificially (Ballantine and Tanner, 2010). The crystal structure of zeolites is fine-grained with open meshes, and accordingly the material has a low bulk density and high porosity. Zeolites have a high adsorption capacity, an affinity for ammonium and a high cation exchange capacity (Ballantine and Tanner, 2010). Studies have indicated that zeolite has a maximum P retention of 0.01 - 0.05 g P kg⁻¹, which is dependent on the pH (Vohla et al., 2011). Several studies of P-retention in zeolite have been conducted in New Zealand (e.g., Nguyen, 1997; Nguyen and Tanner, 1998; Sukias 2005; and Srinivasan 2008). From these studies, mordenites have been documented as the most effective zeolites for P-retention (Sukias, 2005), although comparatively lower values for coarse zeolite and fine zeolite were reported by Nguyen (1997). In addition, Srinivasan et al. (2008) found that a mixture of zeolite and limestone materials resulted in a higher P-sorption. Furthermore, zeolite has previously been trialled in New Zealand for its suitability for P removal from Lake Rotorua, and is generally available from lower temperature geothermal zones (McDowell et al., 2008).

Red mud is created as a by-product of the Bayer Process which is necessary during aluminium production (Vohla et al., 2011). Difficulty exists in the disposal of red mud due to the large volumes which are produced and the high alkalinities of the by-product. Therefore, environmental and economic concerns have led to research into the use of red mud to facilitate P removal. Studies have indicated that red mud can be effective to adsorb 114 g P kg⁻¹ for raw red mud to 345 g P kg⁻¹ from extra heated red mud (Vohla et al., 2011). As red mud is a by-product of aluminium production, it is likely to be locally available from Tawai Smelter, Bluff.

Oyster shells are another material with the potential to effectively absorb P in an in-situ filter. An experiment conducted by Kwon et al. (2004) indicated a P removal rate of between 68% - 98% for heat treated oyster shells. Shells were heated to between 750 - 800°C. Highest removal rates (98%) were achieved under pyrolysed (nitrogen) conditions, whereas heating under natural air conditions corresponded to a lower P absorption of 68% (Kwon et al., 2004). High temperatures are required to achieve decomposition of the carbonates to allow for the high rates of P-adsorption. In comparison, raw oyster shells that received no heat treating, removed almost no P during the same experiment. The local Bluff Oyster industry is a likely local source of oyster shells.

While there are a large number of P-sorbing materials that could potentially be used in filters, chosen materials should ideally be low cost, available locally to reduce transport costs, available in sufficient quantity, non-toxic and reusable once P-saturated or able to be safely and locally disposed. Another consideration to make when selecting material is the longevity of the media, which depends on the sorption characteristics, hydraulic loading rate, flow dynamics and influent P concentration. Regardless of the P-sorption capacity of a media, sorption will be a finite process as the media becomes saturated.

2.2.3 Hydraulic considerations

There is very limited information available regarding the hydraulic loading rate (HLR) and retention times in P-filters. Therefore, no significant correlation exists between HLR and P-retention, although results from several trials indicate that there is most likely an optimal HLR for P-filters (Vohla et al., 2011). In the case of a hydrated oil shale ash filter, a long HLR caused chemical clogging, which reduced the P-removal capacity of the filter. The optimal HLR of a particular P-filter is likely to be primarily dependant on the type of material used (Vohla et al., 2011).

There is a high variation in the particle size of materials available for use in P-filters. The most suitable materials are generally those which are fine textured, and therefore exhibit a high surface area (Ballantine and Tanner, 2010). It is also important that the media has sufficient permeability for water to flow through, as permeability decreases with decreasing particle size there is a trade-off. Generally, a particle size of > 5 mm, and preferably 10 - 20 mm, is recommended (Drizo et al., 1999).

2.2.4 Adverse environmental effects

When deciding on a suitable material to be used in the P-filter, it is important to consider potential toxicity to the environment (Ballantine and Tanner, 2010; McDowell et al., 2008). For example, many of the readily available materials for P-filters (e.g., steel slag) are also high in heavy metals such as arsenic (As), cadmium (Cd), mercury (Hg) and Boron (B). The concentration of some heavy metals is likely to exceed the respective limits set for New Zealand soils, and may also impair stock health (McDowell et al., 2008). In comparison to industrial by-products, natural materials (e.g., oyster shells, limestone, bauxite) are less likely to pose toxicity issues. Furthermore, following utilisation in a P-filer, natural materials have a higher potential to be further recycled (e.g., as aggregate, on farm races) then industrial by-products or manmade materials (Ballantine and Tanner, 2010).

3.0 NITRATE AND PHOSPHOROUS CONCENTRATIONS AND FLOW RATES OF TILE DRAIN DISCHARGE IN SOUTHLAND REGION

The direct measurement of tile drain flow rates and/or nutrient concentrations in the Southland region are limited to three studies: two by Monaghan et al. (2005, 2009) and unpublished sampling results by ES (Hodson, 2012). Monaghan et al. (2005) reported nitrogen concentrations and flow rates of tile drainage water for an area in eastern Southland in a study that looked at the effects of increased N fertiliser inputs and associated cattle stocking rates on pasture, soil and nutrient losses. In the latter study (Monaghan et al., 2009) they assessed the effectiveness of dicyandiamide in limiting NO_3^- leaching from a grazed dairy pasture from a site near Invercargill and reported monthly drainage volumes and N losses. The ES nutrient concentration data was from one-off and routine sampling of tile drains, open drains and waterways in the upper Waihopai River catchment, for the period February 2007 to June 2012.

Monaghan et al. (2005) reported volume-averaged $\text{NO}_3\text{-N}$ and TP concentrations of 6.9 mg L^{-1} and 0.074 mg L^{-1} , respectively. These concentrations are consistent with the ES results presented in the cumulative frequency distributions. For example, in the ES dataset, $\text{NO}_3\text{-N}$ concentrations varied from 1.5 mg L^{-1} (25 percentile) to 5 mg L^{-1} (75 percentile) (Figure 3.1), and TP concentrations varied from 0.015 mg L^{-1} (25 percentile) to 0.3 mg L^{-1} (75 percentile) (Figure 3.2). The 50 percentile cumulative frequency distribution for $\text{NO}_3\text{-N}$ and TP are 1.8 mg L^{-1} and 0.075 mg L^{-1} , respectively, for the ES data.

Monaghan et al. (2009) reported an average annual rainfall of 1127 mm and calculated annual tile drainage totals of between 375 - 421 mm for the four year study period from 2004 - 2007. Tile drains monitored during the study intercepted on average 55% of rainfall received at the sites, but there was considerable variation in drainage recoveries of individual plots ranging from 21-100%. The highest mean monthly drainage volumes were recorded in the late autumn and winter months of June (93 mm), May (67 mm) and August (53 mm) (Figure 3.3).

Estimates of daily flow rates and mass of NO_3^- and TP discharging from tile drain networks are required for the design and sizing of a bioreactor and filter for the pilot trial. ES requested that flow rates and nutrient mass loadings be calculated for drainage areas of 2, 5, 10, 20 and 50 ha. Tile drain flow rates for these areas were estimated using the minimum (21%), mean (55%) and maximum (100%) drainage recovery percentages reported by Monaghan et al. (2009) (Table 3.1; Figure 3.4). The mean daily flow rates were calculated from an annual rainfall of 1,127 mm distributed evenly over the year at 3.1 mm d^{-1} . The estimated flow rates range from $13 \text{ m}^3 \text{ d}^{-1}$ for a 2 ha drainage area at 21% drainage recovery, up to $1544 \text{ m}^3 \text{ d}^{-1}$ for a 50 ha drainage area at 100% drainage recovery. As the mean 55% drainage recovery scenarios are more representative of average drainage conditions, they will most likely be used for the pilot trial design. For 55% drainage recovery, flow rates ranged from $34 \text{ m}^3 \text{ d}^{-1}$ for a 2 ha drainage area up to $849 \text{ m}^3 \text{ d}^{-1}$ for a 50 ha drainage area.

A suitable drainage area for a pilot scale treatment system is in the range of 5 to 20 ha, which corresponds to a mean daily flow rate of 85 to $340 \text{ m}^3 \text{ d}^{-1}$, when NO_3^- loading and flow rates are considered. NO_3^- loading rates from the 2 ha area are likely to be too small for a pilot trial and unnecessarily large from a 50 ha area. Even for a pilot trial of a 5 to 20 ha area,

bypass flow structures will need to be incorporated in the design to restrict the maximum instantaneous flow rate to the bioreactor and filter to less than several litres per second to avoid exceeding the hydraulic capacity of the system. It is likely that drainage recovery will exceed 55% and the hydraulic design of the bed during winter, following prolonged rainfall, or under saturated soil conditions.

Estimated daily mass of NO_3^- and TP that will discharge from the drainage areas has been calculated by:

$$\text{mass (in g d}^{-1}\text{)} = \text{flow rate (in m}^3 \text{ d}^{-1}\text{)} \times \text{concentration (in g m}^{-3}\text{)}$$

The mass loadings of $\text{NO}_3\text{-N}$ (Table 3.2a and Figure 3.5a) and TP (Table 3.2b and Figure 3.5b), were based on the flow rates presented in Table 3.1, and the lower (25%), upper (75%) and mean (50%) percentile $\text{NO}_3\text{-N}$ and TP concentrations. The considered pilot trial options are designed for 55% drainage recovery, incorporating 75 percentile concentrations for $\text{NO}_3\text{-N}$ and P, with a drainage area of 5 to 20 ha. These values are based on the request from ES to develop a pilot trial to reflect a nutrient hotspot, e.g., runoff from a tile drain network. The daily mass loadings for these scenarios range from 425 g N d^{-1} and 25.5 g P d^{-1} for 5 ha up to 1,698 g N d^{-1} and 102 g P d^{-1} for 20 ha (Table 3.2).

4.0 PROPOSED BIOREACTOR AND FILTER DESIGNS FOR THE PILOT TRIAL

A bioreactor and filter design needs to be tailored to accommodate site-specific conditions that include the range of flow rates, $\text{NO}_3\text{-N}$ and TP concentrations, temperatures, discharge points, and removal goals. The size of the bioreactor and filter are dictated by a combination of these conditions. For the pilot trial design in this study, the following conditions have been considered in the described manner:

- Mass load of N and P in tile drain discharge that were calculated from measured concentrations and estimated flow rates. Mass loads were based on the lower, upper and 50 percentile $\text{NO}_3\text{-N}$ and TP concentrations of tile and open drain water provided by ES (Hodson, 2012). Tile drain flow rates were estimated for 2, 5, 10, 20 and 50 ha drainages areas using the minimum (21%), mean (55%) and maximum (100%) drainage recovery values reported by Monaghan (2009).
- Carbon and p-sorbent filter media, which controls the removal rate. Bioreactor size has been estimated for three C media (wood chip, maize cobs and wheat straw) and filter size for six P-sorbent media (limestone, bauxite, zeolite, modified zeolite, red mud (unheated) and oyster shell).
- Temperature, as the biological denitrification process is temperature dependent. A mean annual temperature of 10°C (NIWA, 2013) has been assumed for the Southland region for bioreactor design.
- Treatment goal of $\text{NO}_3\text{-N}$ concentration in the bioreactor discharge and replacement interval of the P-sorbent filter material. The proposed bioreactor and filter size are based on >95% $\text{NO}_3\text{-N}$ removal and 20 year replacement interval for the P-sorbent material, respectively.
- Given the relatively low average $\text{NO}_3\text{-N}$ and TP concentrations, conditions in the bioreactor and filter may often be N and P limited. As such, removal rates will be lower than rates for non-limiting conditions due to reduced nutrient contact with reactive sites. In nutrient limited conditions it is important to maximise nutrient-media contact by minimising by-pass flow with appropriate bioreactor/filter size, dimensions, inlet-outlet structure and HRT. NO_3^- removal rates for wood-based bioreactors for N limited conditions (2-5 g N $\text{m}^3 \text{d}^{-1}$) are about half those for non-limited N conditions (5-10 g N $\text{m}^3 \text{d}^{-1}$), with the range a function of variation in influent N concentration and operating temperature (Schipper et al., 2010a). Removal rates for other C and P-sorbent media under nutrient limiting conditions have not been identified by researchers. For the purposes of this study removal rates for media in nutrient-limiting conditions are assumed to also be half of the reported literature values.

The required volume of C and P-sorbent medias for the bioreactor and filter have been calculated for the considered range of NO_3^- and TP loading rates, drainage areas (2, 5 10, 20 and 50 ha) and media removal rates (Figures 4.1 and 4.2). P-sorbent filter volumes include reported literature values of specific gravity, for conversion of P-sorbition rate (in g P kg^{-1}) to media volume.

As noted in Section 3, a suitable drainage area for a pilot trial is between 5 to 20 ha. Typically systems are designed to treat average loading conditions as designing for peak loading conditions requires a prohibitively large system. However, ES requested that the pilot trial be designed to treat the 75 percentile nutrient loading rate under average flow conditions for placement in a “nutrient hotspot”. Reactor and filter volumes for these criteria are presented in the centre 55% drainage recovery plots (Figures 4.1 and 4.2), with the upper bars representing the 75 percentile $\text{NO}_3\text{-N}$ and TP concentrations of 5 N mg L^{-1} and 0.3 P mg L^{-1} , respectively. Bioreactor and filter volumes for other drainage areas and nutrient loading rates are not further discussed.

The estimated bioreactor volume for the design criteria varies from 21 m^3 for a 5 ha drainage area using maize cobs as the carbon source, up to 340 m^3 for a 20 ha drainage area using woodchip (Table 4.1a and Figure 4.1). The estimated P-filter volume for the same conditions and drainage area varies from 0.03 m^3 for oyster shells up to 20 m^3 for limestone (Table 4.2 and Figure 4.2).

The length of the bioreactor and filter is a function of cross-sectional area. The typical cross sectional area of a denitrifying bioreactor is about 4 m wide x 1.5 m deep (6 m^2). These dimensions allow the excavation to be completed by a moderate size excavator (e.g., 12 t) that is commonly available for hire. This size also reduces the required battering of the trench walls to achieve wall stability and meet health and safety requirements of working in open pits during construction. Calculated bed and filter lengths for the media volumes presented in Tables 4.1a and b and Tables 4.2a and b, are based on a 6 m^2 cross sectional area. However, as the required volume of both carbon and P-filter media for a 5 ha area is relatively small (c. $< 85 \text{ m}^3$), and a longer excavation with smaller cross-sectional area may be more appropriate in this setting (e.g., Figure 4.3). The cross sectional area and length of the pilot trial bed/filter should be based on the hydrology, topography and layout of the site.

5.0 PROPOSED PILOT TRIAL

5.1 Pilot trial design

The pilot trial design needs to consider the site specific hydrology, landform and tile drain layout. However, some fundamental concepts and design components are common to all installations and are presented in this section. While the system can be scaled to suit a site specific drainage area and nutrient concentrations, for the ease of discussion an example drainage area of 5 ha, NO₃-N concentration of 5 mg L⁻¹ and TP concentration of 0.3 mg L⁻¹ is assumed. ES indicated that the pilot trial will likely be sited in a “nutrient hotspot”, so the above high nutrient concentrations have been used as design criteria (rather than median concentrations). These values represent 75 percentile concentrations that have been measured in tile drain discharge by ES (Hodson, 2012), and correspond to a mean hydraulic loading rate of 85 m³ d⁻¹ and mass nutrient loads of 425 g N d⁻¹ and 25.5 g P d⁻¹.

The general placement and design of the system is similar to that shown in Figure 5.1. It should be placed parallel to, and near the downstream end of, the tile drain discharge line. An inlet chamber/weir is required to direct all tile drain flow up to 1 L s⁻¹ into the bed. Excess flow would by-pass the bed via the existing tile drain. The outlet chamber/weir maintains water level in the bed and stops return flow into the bed during periods of by-pass flow. It is recommended that the bioreactor and filter be placed in the same excavated trench with C and P-sorbent media separated (Figure 5.1). The C media should be placed at the upstream end of the system to act as a filter for removal of suspended sediment (SS) from the influent water. As a result the upstream 2-3 m section of the C media may need to be replaced at regular interval depending on SS load, due to loss of permeability.

Pinus Radiata woodchip media of greater than 15 mm chip size is recommended as the C media. A volume of 85 m³ is required, which is the equivalent of about 205 t. The woodchip would occupy a 28 m length of the bed assuming a cross section area of 6 m² (4 m W x 1.5 m D; Figure 5.2 and 5.3). Crushed oyster shells of between approximately 10 mm to 20 mm grain-size are recommended as the P-filter media. A volume of 0.2 m³ is required based on published P removal rates for heated oyster shells. However, it is recommended to increase this volume substantially to 12 m³, to increase the HRT of water through the media. Oyster shells would occupy a 2 m length assuming a cross section area of 6 m². Oyster shells can be sourced locally from Bluff, and can be heat treated to 750°C - 800°C in an electric or gas oven at Bradken Engineering, Dunedin.

The recommended hydraulic gradient within the bed is 1:100, or 300 mm for the proposed 30 m long bed. This would provide a hydraulic load capacity of between about 120-360 m³ d⁻¹ (~1-4 L s⁻¹), when potential variation in permeability of the woodchip media is considered. The design load for the system is 85 m³ d⁻¹, so this gradient also provides a reasonable safety margin to reduce risk of hydraulic failure. The hydraulic gradient in the bed can be maintained (and adjusted if necessary) by the relative difference in elevation between the inlet and outlet pipe and/or elevation of the outlet chamber weir (Figures 5.1, 5.2 and 5.3).

An impermeable liner will be installed in the excavated trench between the media and trench wall to prevent water loss from the system. A range of liner materials are available (e.g., HDPE, PVC, polythene), all of which vary considerably in price and durability. To minimise

installation cost for the pilot trial, it is suggested to use a relatively cheap option of double or triple layer of 25 micron silage wrap, which can be installed by a resourceful untrained field team. Silage wrap can be purchased for several hundred dollars as opposed to \$5,000 - \$6,000 for more durable HDPE, which would require professional installation. Sharp objects and stones need to be removed from the base of the excavated trench before the liner is installed, to prevent puncture of the liner. A layer of sand (or similar) may be required if the base material is not suitable (e.g., gravel). Also, a protective layer(s) of "geotextile" material (e.g., bideem cloth) may need to be laid between the oyster shells and the liner to prevent the liner from the risk of punctures from the crushed shells.

The inlet and outlet structures should be designed to reduce potential for preferential flow and maximise the volume of "active" media in the system. One way to achieve this is to have the inlet and outlet structure occupy the cross sectional area of the bed and be located at the upstream and downstream limits of the bed (Figures 5.1, 5.2 and 5.3). The suggested inlet and outlet design is a vertical array of three (or more) perforated, 100 mm diameter, PVC pipes, capped at the base and connected by a horizontal pipe. The perforations (e.g., hand drilled holes) must be sufficiently large and frequent to easily accommodate the maximum design flow of 1.4 L s^{-1} , but not allow media to enter the pipes. The inlet and outlet feeder pipe can be sealed against the impermeable liner material with an appropriately sized "tank" fitting.

The top of the system can be installed below ground level and overlain by soil and sod, if hydrological conditions (e.g., drain and water levels and available hydraulic head) at the site permit. This would allow the area to be grazed. A layer of permeable geotextile fabric should be placed between the woodchip/oyster shell media and the soil to prevent contamination of the media and associated loss of permeability. It is recommended that the pilot trial system is not buried until it is known that the system is working correctly.

5.2 Recommended monitoring schedule

Monitoring of the system for NO_3^- removal, P removal and flow rate is required for performance monitoring purposes. A measurement of system flow rate needs to be taken at the same time as nutrient sampling. Flow rate can be measured either upstream or downstream of the bed/filter system by recording the time taken to fill a container of known volume, and repeated several times. Nutrient sampling needs to be undertaken at the inlet and outlet, and at sampling piezometers installed within the bed. Inlet and outlet water samples can be collected by grab sampling at the inlet and outlet chambers. Five PVC sampling piezometers (~ 25 mm diameter) should also be installed along gradient in the middle of the bed (Figures 5.1 and 5.2).

The recommended monitoring regime is provided in Table 5.1. This program contains a recommended minimum monitoring regime and a more comprehensive monitoring regime that could be undertaken depending on available funding and the objectives of the pilot trial. The minimum monitoring regime is designed to minimise sampling and analytical costs while still providing adequate nutrient removal information for future system design purposes. However, it does not address leaching of nutrients from the bed during the start-up period (~0-6 months), which may be of importance depending on discharge environment. The comprehensive sampling regime addresses leaching during start-up.

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TABLES

Table 2.1 Nitrate removal rates for various carbon sources in denitrifying bioreactors.

Carbon media	NO ₃ -N removal rate (g N m ³ d ⁻¹)	Comment	Reference
<i>Pinus radiata</i> woodchip	3.0 - 4.9	200 L barrel trial. Rates are likely conservative due to effects of DO in influent water	Cameron and Schipper (2010)
	2 - 10	Lower rates limited by nitrate concentrations	Schipper et al. (2010a)
	7.6		Warneke et al. (2011)
Eucalyptus woodchip	3.3 - 4.4	200 L barrel trial. Rates are likely conservative due to effects of DO in influent water	Cameron and Schipper (2010)
Maize cobs	23.3		Cameron and Schipper (2011)
Wheat straw	5.8		Cameron and Schipper (2010)
Green waste	7.8		Cameron and Schipper (2010)

Bolded media have been included in this study for estimating potential options for pilot trial design.

Table 2.2: TP removal rate for various materials used in P-sorbent filters.

Class	Material	Process	Rate (g P kg ⁻¹)	Comment
Natural products	Apatite	Adsorption	0.4 - 4.8	
	Bauxite	Adsorption	2.5 - 40	
	Dolomite	Adsorption	9.7 - 52.9	
	Gravel	Adsorption, precipitation and biological assimilation	25.8 - 47.5	Wetland
	Limestone	Adsorption	0.3 - 20	
	Sands	Adsorption and precipitation	0.117	Rate dependent on content of Ca, Al and Fe
	Peat	Adsorption	0.081	
	Shell sand	Adsorption	0.5 - 17	
	Soils	Adsorption	0.001 - 0.005	Rate dependent on content of Al and Fe
	Zeolite	Adsorption	0.01 - 0.462	
	Modified zeolite		45 - 100	
By products				
	Coal fly ash	Adsorption	9.7 - 42.6	
	Red mud (unheated)	Adsorption	113.9	By product of Al production
	Red mud (heated)	Adsorption	345	By product of Al production
	Slag	Adsorption	22 - 25	
Man-made products	Alunite	Adsorption		
	Filtra-P		11.0 - 12	
	Filtralite		209 - 2,210	
	Filtralite P		3,887	small scale
			4,500	meso scale
			52	full scale
	Light expanded clay aggregates		0 - 10	
Oyster shell	Adsorption	833 - 7,925	P concentration dependent?	

Bolded materials have been included in this study for estimating potential options for pilot trial design.

Table 3.1 Estimated mean daily tile drain flow rate for specified drainage areas and rainfall recovery. Values are based on drainage recovery values from Monaghan et al. (2009).

		Drainage area (ha)				
		2	5	10	20	50
		Mean daily tile drain flow rate (L s ⁻¹)				
Rainfall recovery (%)	21	0.2	0.4	0.8	1.5	3.8
	55	0.4	1.0	2.0	3.9	9.8
	100	0.7	1.8	3.6	7.1	17.9
		Mean daily tile drain flow rate (m ³ d ⁻¹)				
Rainfall recovery (%)	21	13	32	65	130	324
	55	34	85	170	340	849
	100	62	154	309	618	1544

Table 3.2 Estimated mean daily mass (in grams) of, (a) NO₃-N and (b) TP, discharging from 2, 5, 10, 20 and 50 ha drainage areas, with lower (21%), mean (55%) and upper (100%) drainage recovery rates.

(a)

		NO ₃ -N concentration in drainage water (mg L ⁻¹)		
		1.5	1.8	5
Rainfall intercepted (%)	Area	2 ha		
	21	19	23	65
	55	51	61	170
	100	93	111	309
	Area	5 ha		
	21	49	58	162
	55	127	153	425
	100	232	278	772
	Area	10 ha		
	21	97	117	324
	55	255	306	849
	100	463	556	1,544
	Area	20 ha		
	21	195	233	648
	55	509	611	1,698
100	926	1,112	3,088	
Area	50 ha			
21	486	584	1,621	
55	1,274	1,528	4,246	
100	2,316	2,779	7,719	

(b)

		TP concentration in drainage water (mg L ⁻¹)		
		0.015	0.075	0.3
Rainfall intercepted (%)	Area	2 ha		
	21	0.2	1.0	3.9
	55	0.5	2.5	10.2
	100	0.9	4.6	18.5
	Area	5 ha		
	21	0.5	2.4	9.7
	55	1.3	6.4	25.5
	100	2.3	11.6	46.3
	Area	10 ha		
	21	1.0	4.9	19.5
	55	2.5	12.7	50.9
	100	4.6	23.2	92.6
	Area	20 ha		
	21	1.9	9.7	38.9
	55	5.1	25.5	101.9
100	9.3	46.3	185.3	
Area	50 ha			
21	4.9	24.3	97.3	
55	12.7	63.7	254.7	
100	23.2	115.8	463.2	

Table 4.1: Bioreactor (a) volume, and (b) length, required to treat NO₃-N load for specified drainage area, concentration and C media, under average (55%) flow conditions. Length assumes a cross sectional bed area of 6 m² (4 m W x 1.5 m D).

a) Bioreactor volume (m³)

Drainage area	5 ha		10		20	
	1.8 mg L ⁻¹	5.0 mg L ⁻¹	1.8 mg L ⁻¹	5.0 mg L ⁻¹	1.8 mg L ⁻¹	5.0 mg L ⁻¹
Woodchip	31	85	61	170	122	340
Wheat straw	26	73	53	146	105	293
Maize cobs	8	21	15	42	31	85

b) Bioreactor length (m)

Drainage area	5		10		20	
	1.8 mg L ⁻¹	5.0 mg L ⁻¹	1.8 mg L ⁻¹	5.0 mg L ⁻¹	1.8 mg L ⁻¹	5.0 mg L ⁻¹
Woodchip	5.1	14.2	10.2	28.3	20.4	56.6
Wheat straw	4.4	12.2	8.8	24.4	17.6	48.8
Maize cobs	1.3	3.5	2.5	7.1	5.1	14.2

Table 4.2: (a) Filter volume and (b) length required to treat TP load for specified drainage area, concentration, and P-sorbent media, under average (55%) flow conditions. Length assumes a cross sectional area of 6 m² (4 m W x 1.5 m D).

a) Filter volume (m³)

Drainage area	5 ha		10 ha		20 ha	
	0.075 mg L ⁻¹	3.0 mg L ⁻¹	0.075 mg L ⁻¹	3.0 mg L ⁻¹	0.075 mg L ⁻¹	3.0 mg L ⁻¹
Limestone	1.27	5.09	2.55	10.19	5.09	20.38
Zeolite	1.27	5.09	2.55	10.19	5.09	20.38
Modified zeolite	0.08	0.33	0.16	0.66	0.33	1.31
Oysters	0.01	0.03	0.02	0.06	0.03	0.12
Bauxite	0.34	1.34	0.67	2.68	1.34	5.36
Red mud (unheated)	0.06	0.22	0.11	0.45	0.22	0.89

b) Filter length (m)

Drainage area	5 ha		10 ha		20 ha	
	0.075 mg L ⁻¹	3.0 mg L ⁻¹	0.075 mg L ⁻¹	0.075 mg L ⁻¹	3.0 mg L ⁻¹	0.075 mg L ⁻¹
Limestone	0.2	0.8	0.4	1.7	0.8	3.4
Zeolite	0.2	0.8	0.4	1.7	0.8	3.4
Modified zeolite	0.014	0.054	0.027	0.109	0.0548	0.027
Oysters	0.001	0.005	0.003	0.010	0.005	0.020
Bauxite	0.056	0.223	0.112	0.447	0.223	0.894
Red mud (unheated)	0.009	0.037	0.019	0.075	0.037	0.149

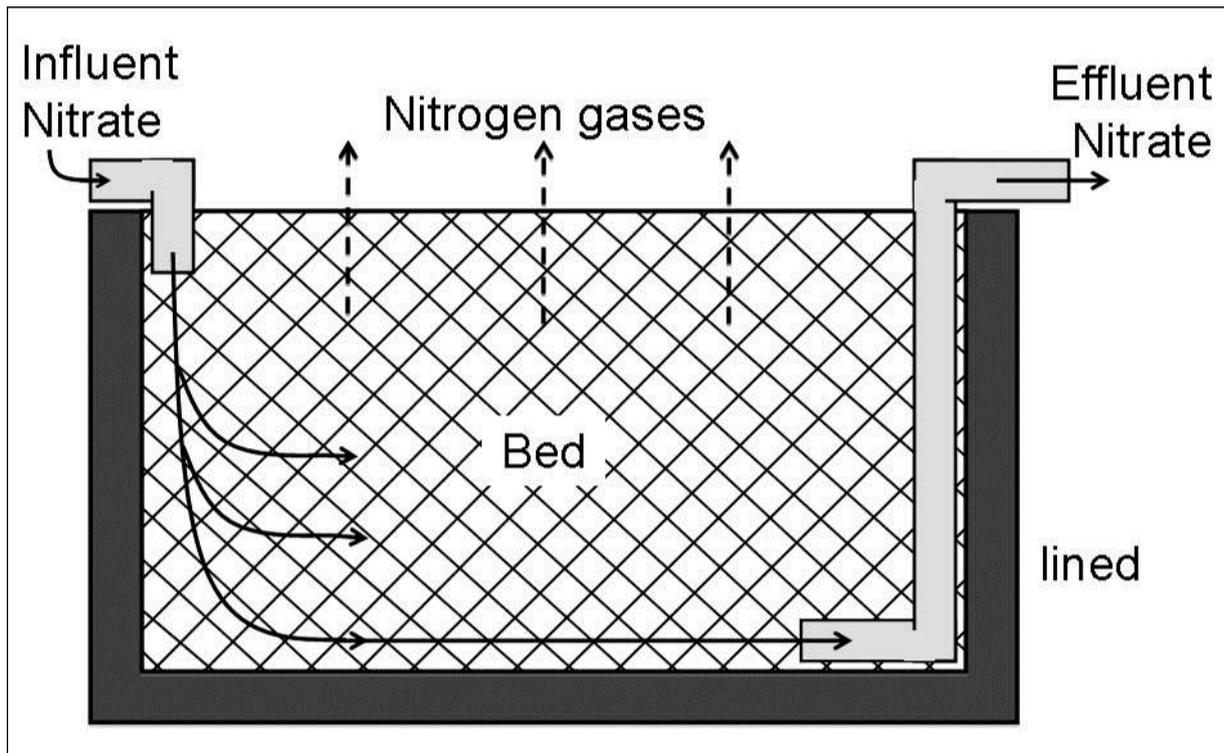
Table 5.1: Recommended minimum and comprehensive monitoring regimes.

Minimum sampling regime			
Sampling location	Parameter	Frequency	Duration
Inlet and outlet	NO ₃ -N, TP, DRP, TN	Monthly	
Piezometers 1-4	NO ₃ -N	Quarterly	
Piezometer 5	TP, DRP	Quarterly	
Inlet or outlet	Flow rate	Monthly	real-time**
Comprehensive sampling regime*			
Inlet and outlet	NH ₄ , BOD, TN	Monthly	1st six months

* In addition to minimum sampling regime

** Real-time monitoring is preferable if budget allows, otherwise measure at time of sampling

FIGURES



(a)



(b)

Figure 2.1: Schematic cross section (a) from Schipper et al., 2010a and (b) photograph, of a denitrification bed.

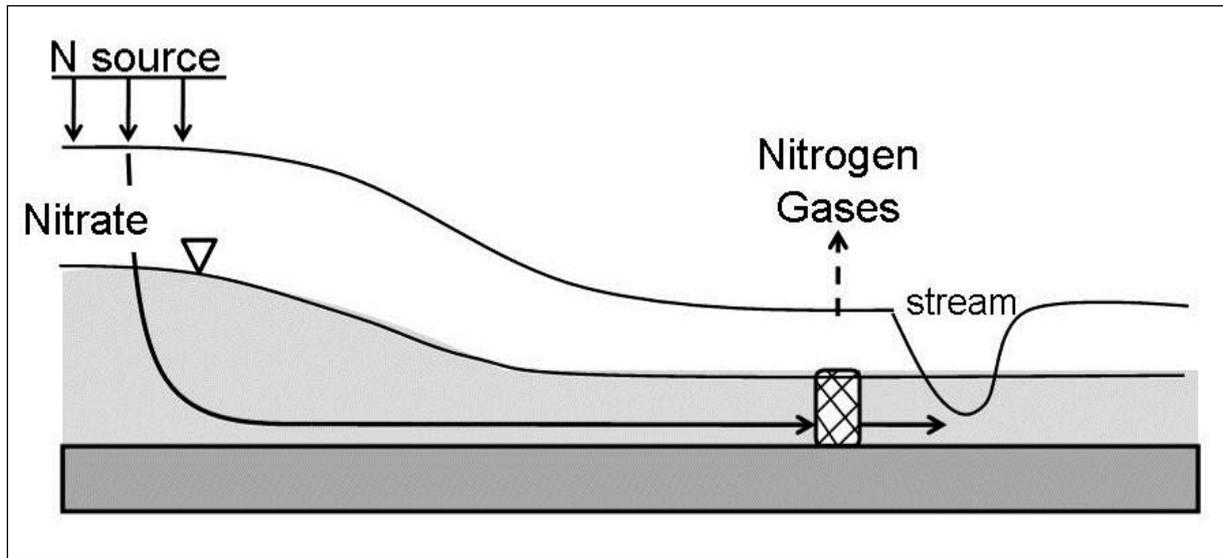


Figure 2.2: Schematic of denitrification wall in cross section (from Schipper et al., 2010a).

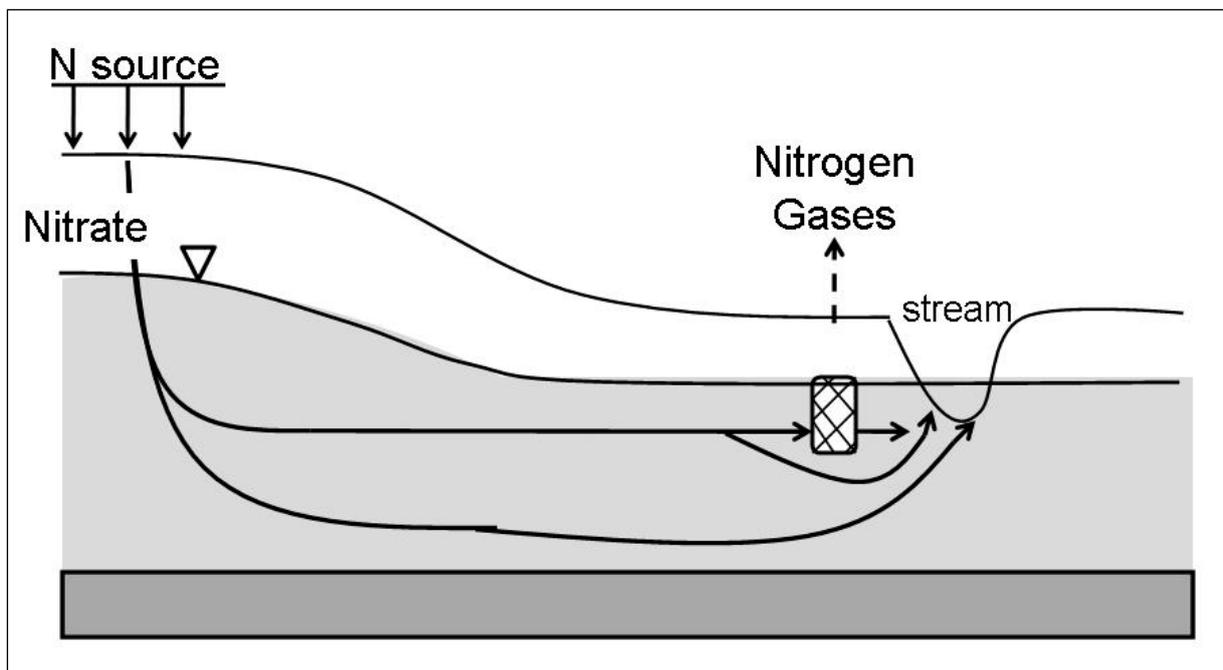


Figure 2.3: Schematic of denitrification wall in cross section where the ground water flow paths may bypass the wall where saturated hydraulic conductivity in the deep aquifer is higher than denitrification wall (from Schipper et al., 2010a).

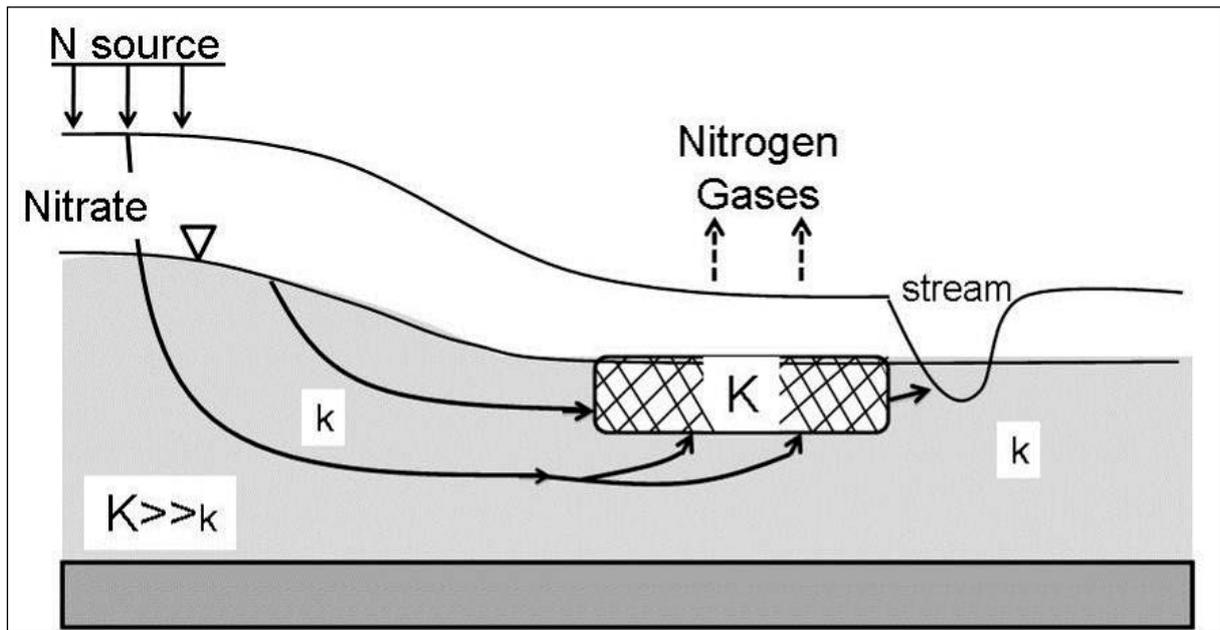


Figure 2.4: Cross-section of a denitrification wall where the wall's K is higher than the surrounding media resulting in upwelling of ground water into the wall (Schipper et al., 2010a).

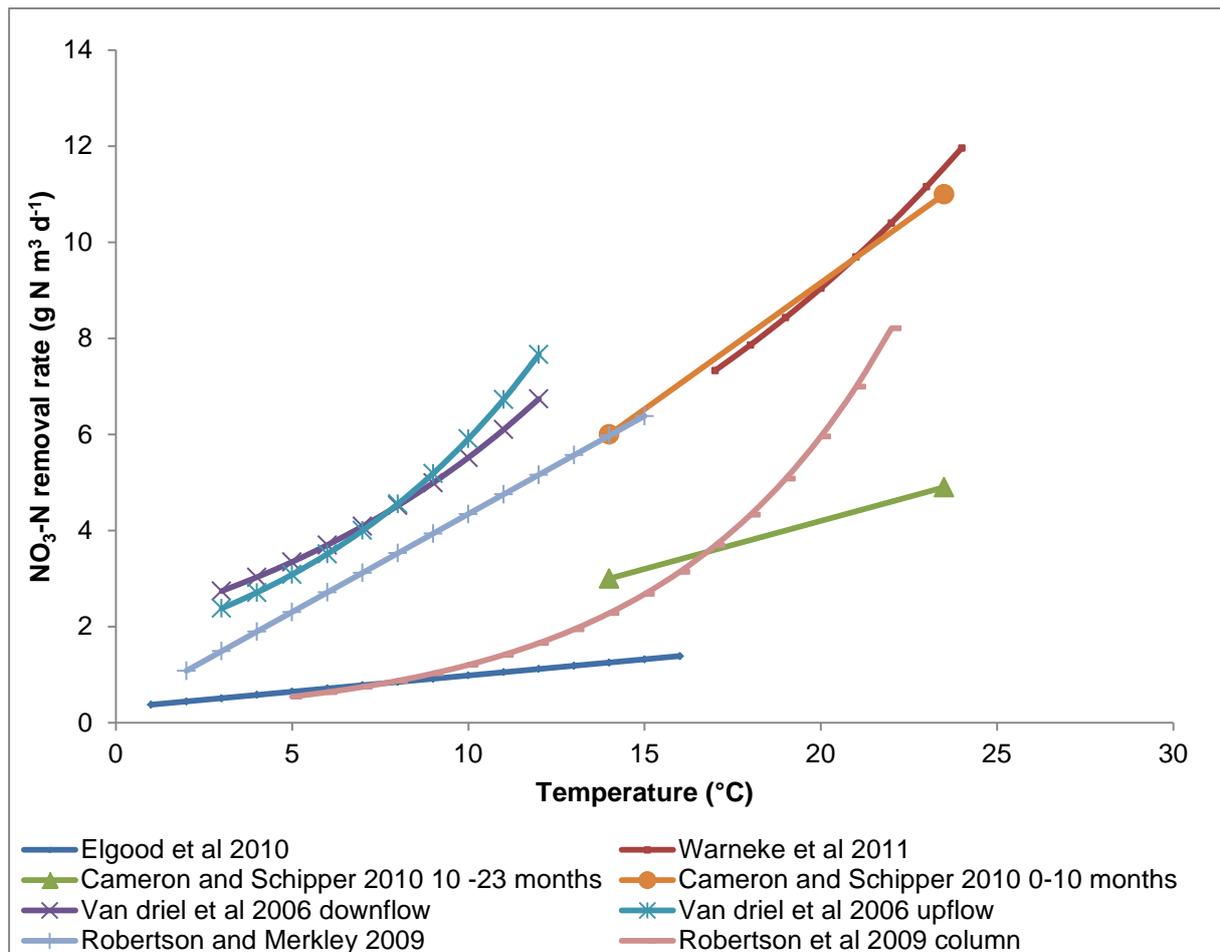


Figure 2.5: Effect of temperature on NO₃-N removal rate. Removal rates in some studies were constrained by NO₃-N concentration (Schipper, 2012).

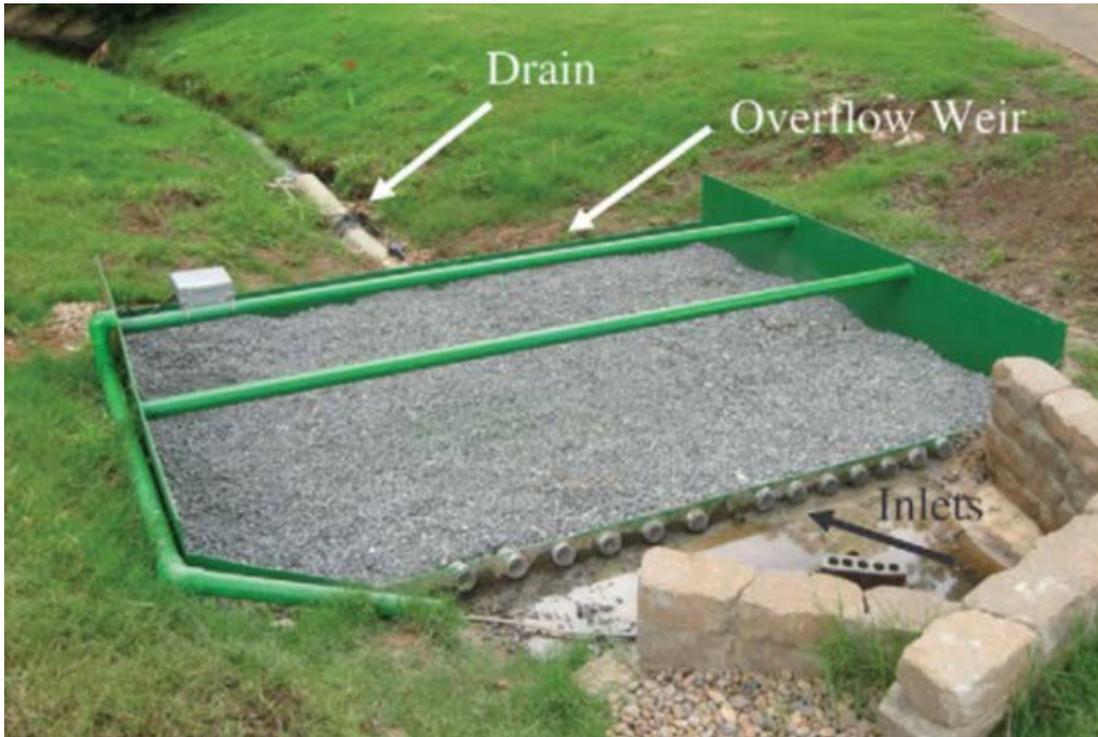


Figure 2.6: Photograph of a phosphorus (P) removal structure with runoff inlets, drain for treated water, and overflow weir. The P sorption material in the structure is 2712 kg of 6.3- to 11-mm-diameter steel slag (Penn et al., 2012).

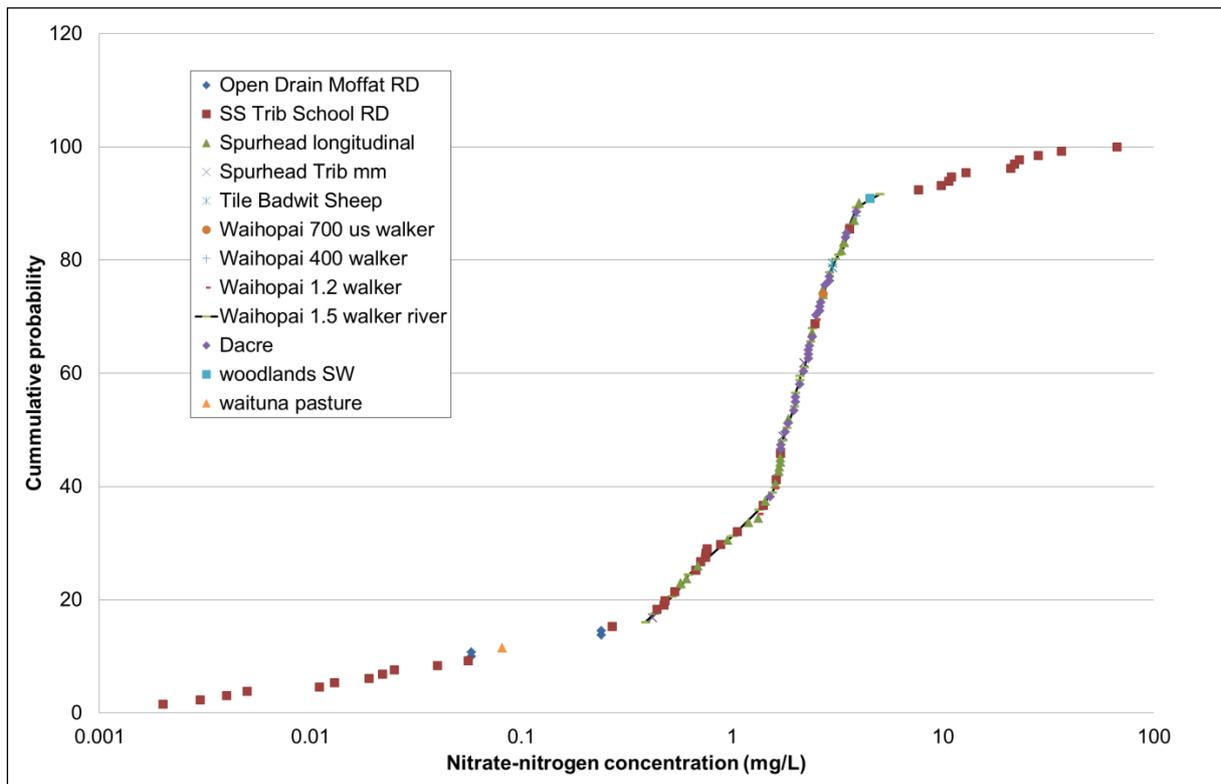


Figure 3.1: Cumulative frequency graph of tile drainage and open drain $\text{NO}_3\text{-N}$ concentrations in the Upper Waihopai River catchment (Hodson 2012).

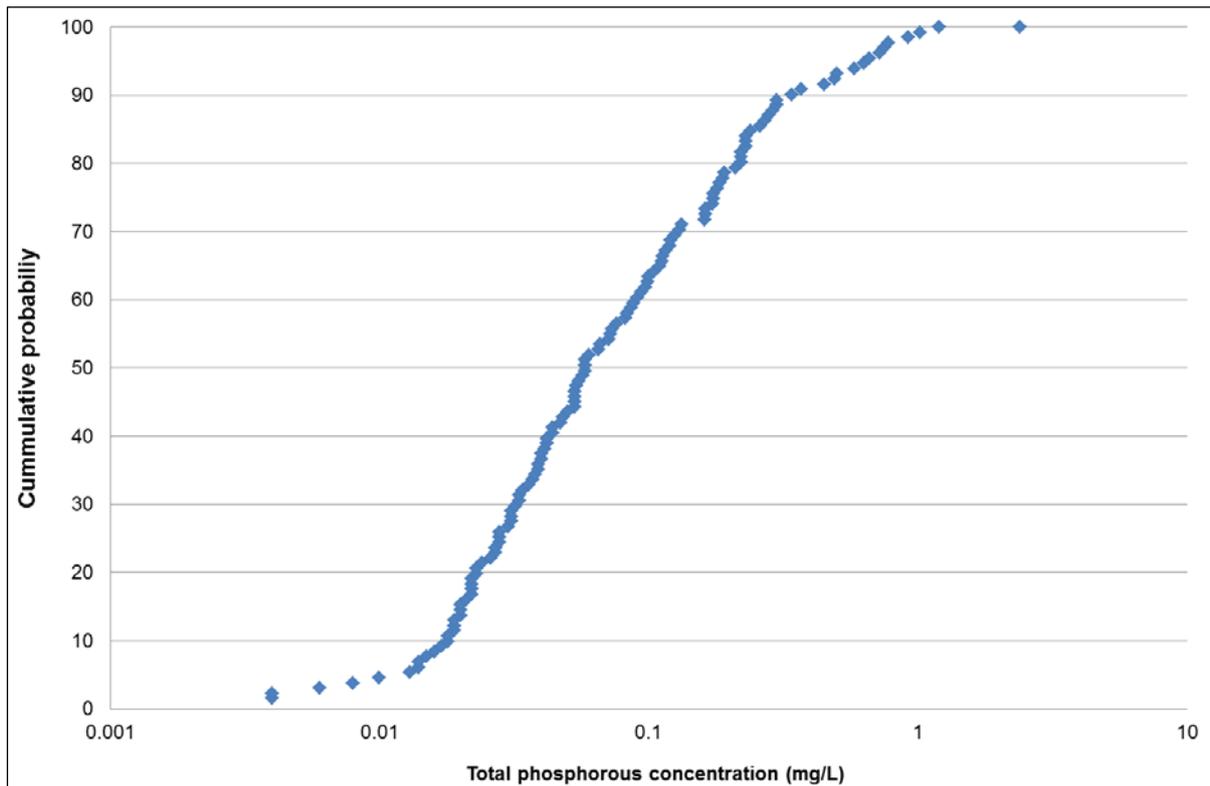


Figure 3.2: Cumulative frequency graph of tile drainage and open drain TP concentrations in the Upper Waihopai River catchment (Hodson 2012).

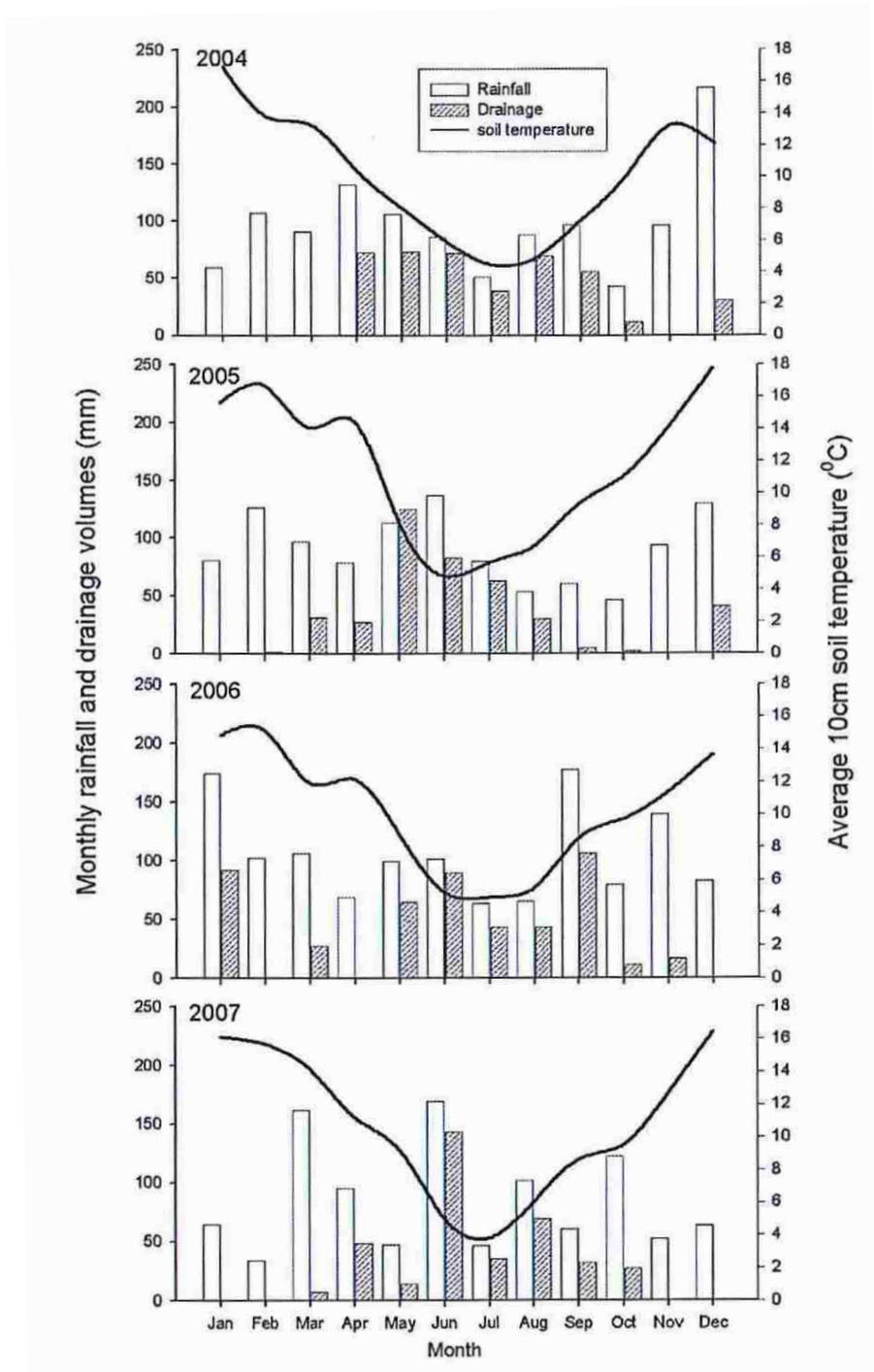


Figure 3.3: Monthly drainage volumes, rainfall and evapotranspiration (ET) totals from 2004 to 2007 (Monaghan et al., 2009).

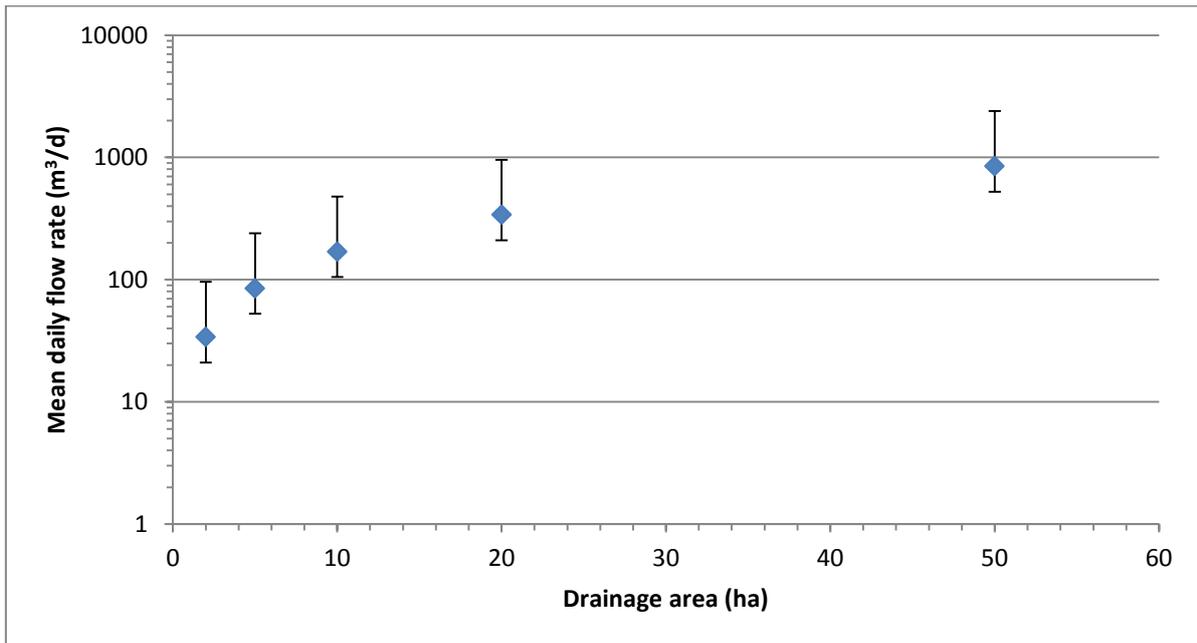
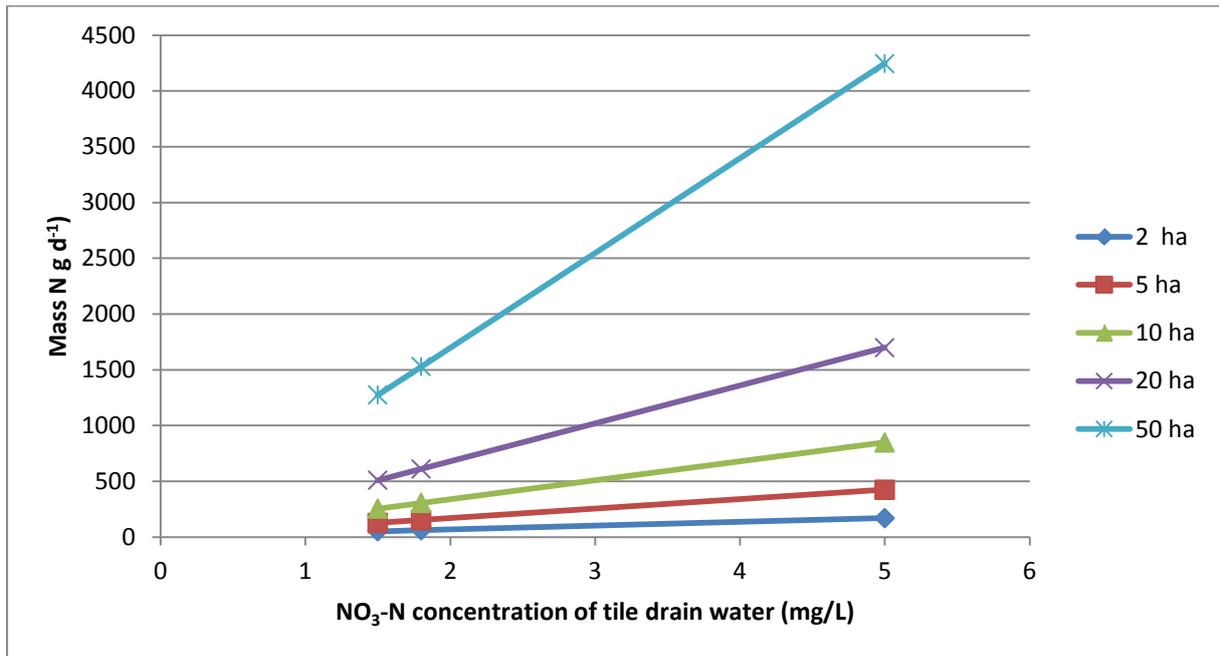
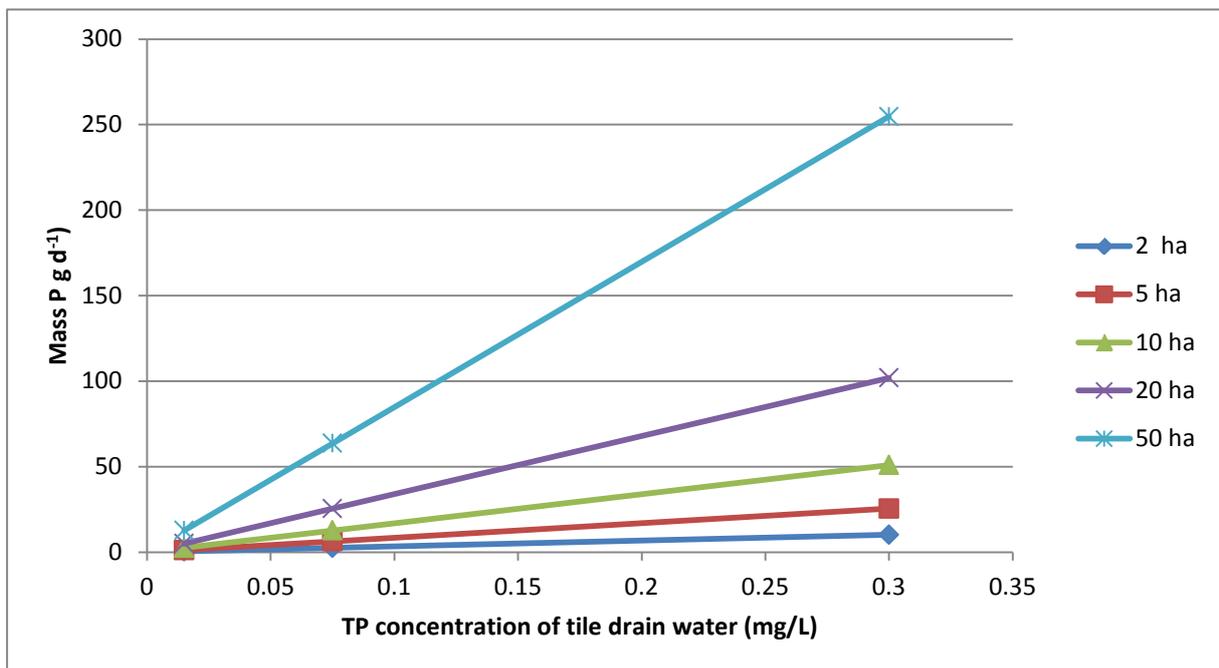


Figure 3.4: Estimated tile drain mean daily flow rate for drainage areas of 2, 5, 10, 20 and 50 ha. The blue diamond is for 55% recovery of rainfall and lower and upper error bars are 21% and 100% recovery, respectively.



(a)



(b)

Figure 3.5: Estimate of mean daily mass of NO₃-N (a) and TP (b) for the 55% recovery of rainfall and lower, upper and 50 percentile concentrations.

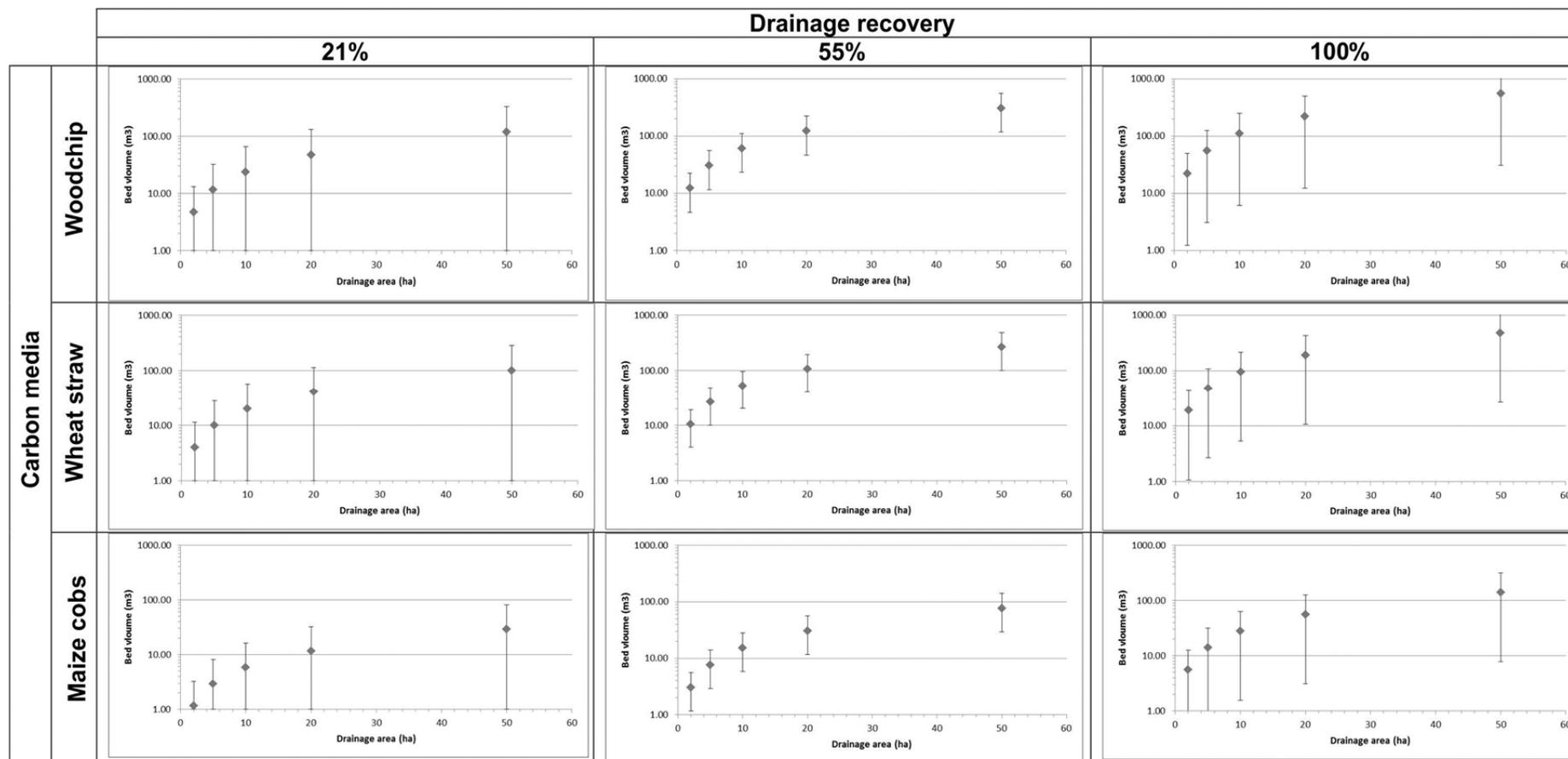


Figure 4.1: Bioreactor volume required to treat estimated $\text{NO}_3\text{-N}$ loading rate for range of carbon media; drainage recovery; drainage area and influent $\text{NO}_3\text{-N}$ concentration. The blue diamond represents reactor volumes based on the influent mean $\text{NO}_3\text{-N}$ concentration; and the lower and upper error bars represent reactor volumes based on the lower- and upper-percentile influent $\text{NO}_3\text{-N}$ concentrations, respectively.

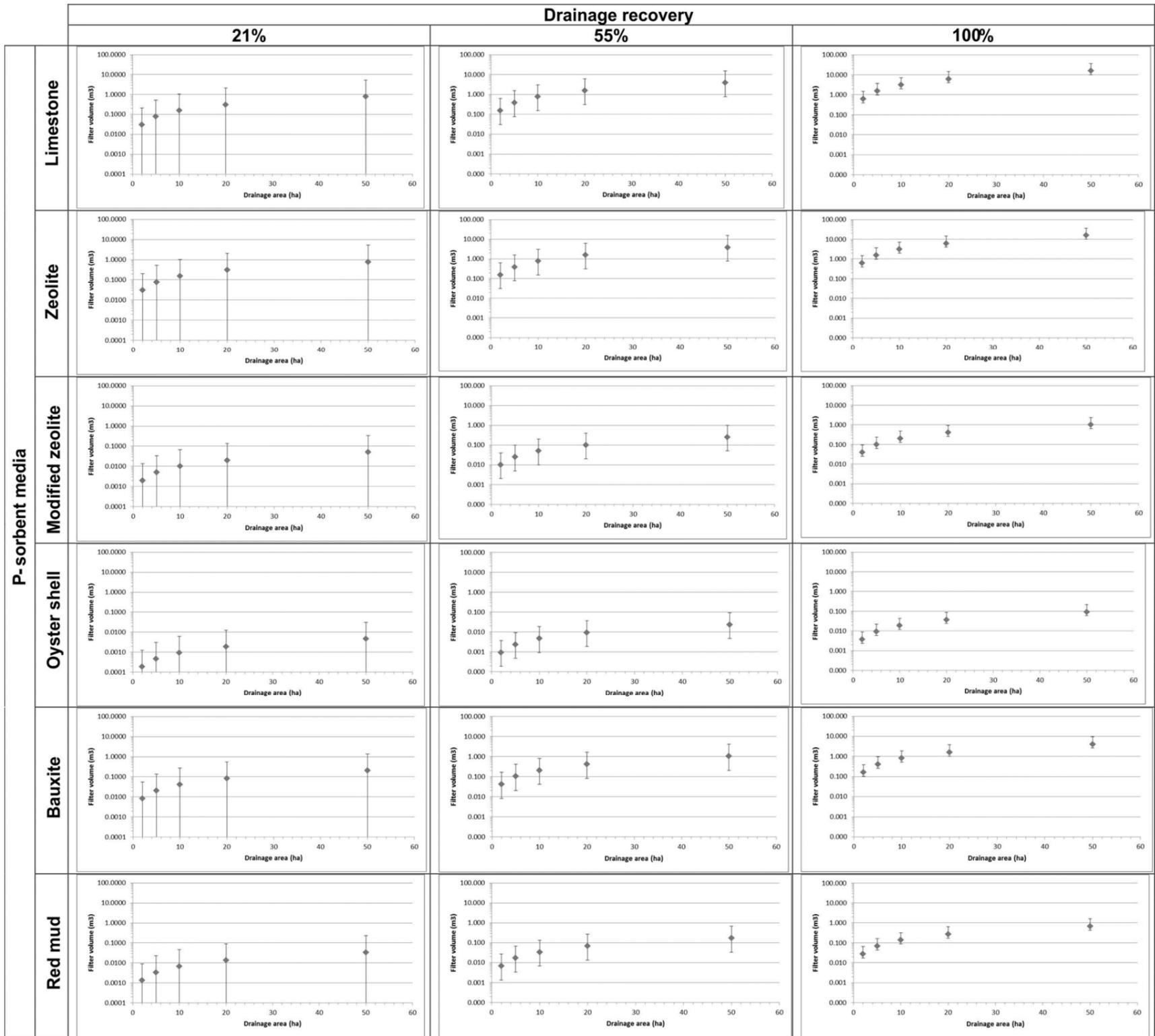


Figure 4.2: Filter volume required to treat estimated P loading rate for variety of P-sorbent material and lower, upper and 50 percentile drainage recovery. The blue diamond represents filter volumes based on the influent mean TP concentration; and the lower and upper error bars represent filter volumes based on the lower- and upper-percentile influent TP concentrations, respectively.



Figure 4.3: Photograph of denitrification bed with small cross sectional area installed for treatment of tile drain water (Photo courtesy of Mark Dittrich, Minnesota Department of Agriculture; from Christianson and Helmers 2011).

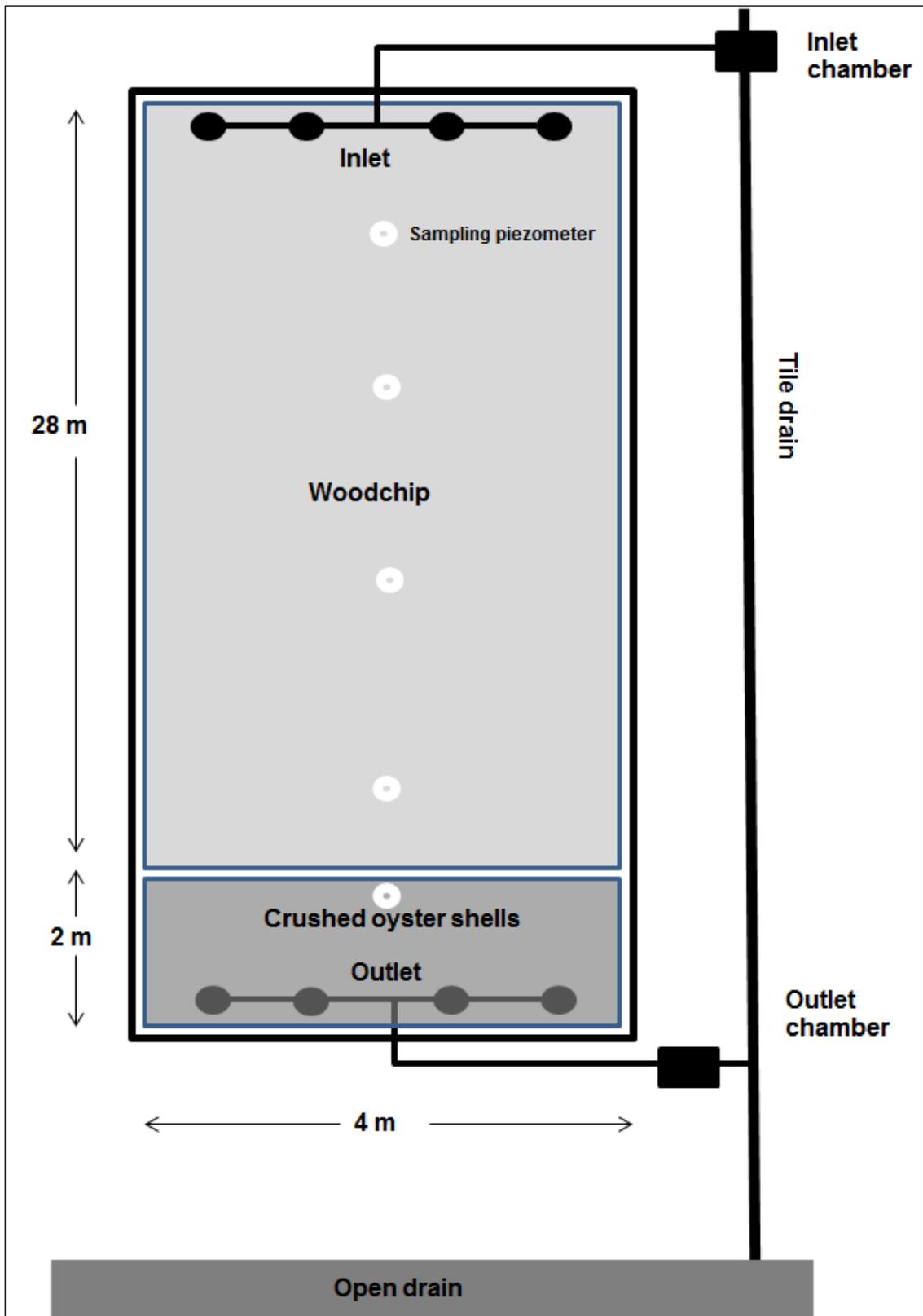


Figure 5.1: Plan view schematic of the pilot trial bioreactor and filter to treat tile drain discharge from a 5 ha drainage (assuming 55% drainage recovery) with $\text{NO}_3\text{-N}$ and TP concentration of 5 mg L^{-1} and 0.3 mg L^{-1} , respectively.

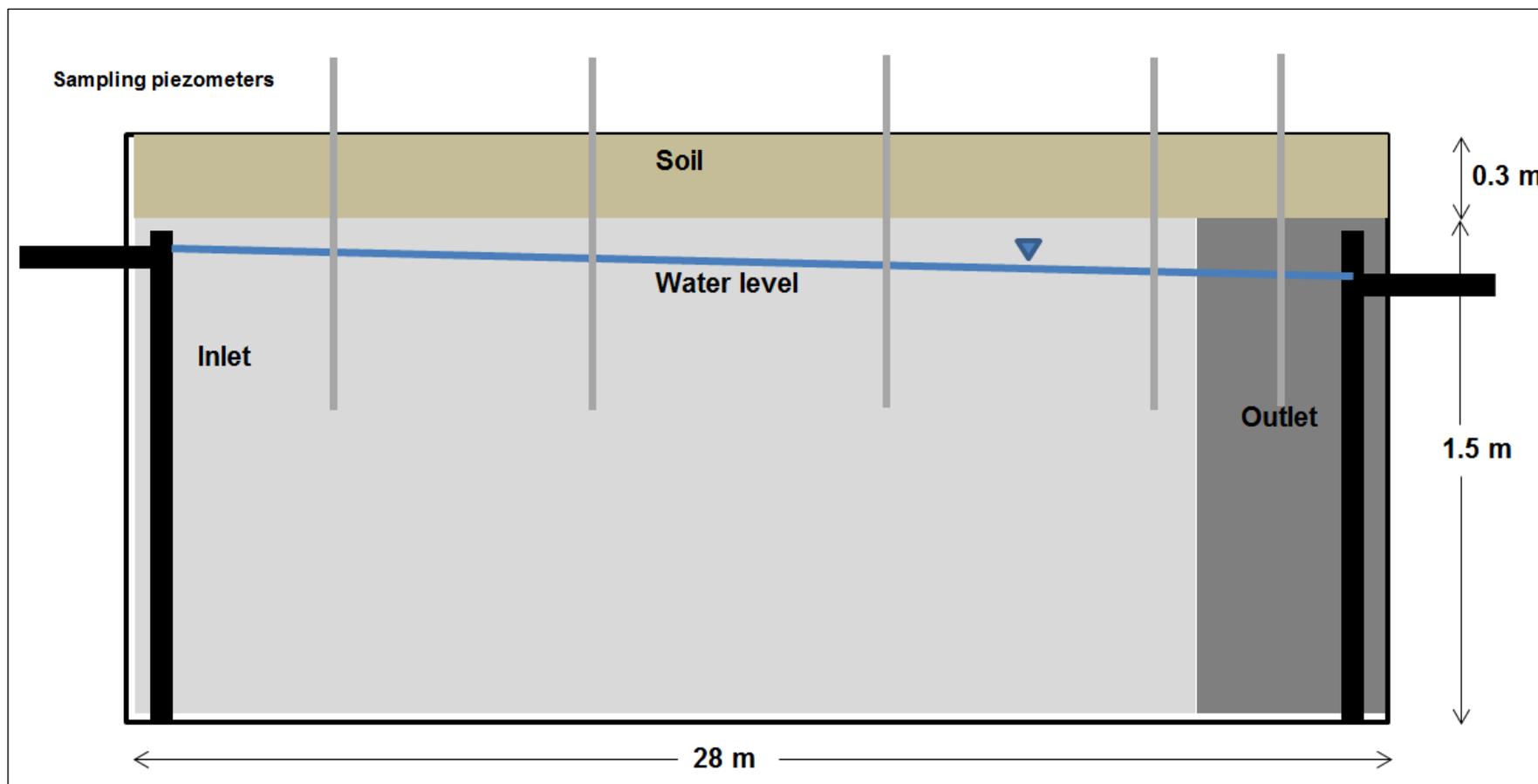


Figure 5.2: Side view schematic of the pilot trial bioreactor and filter to treat tile drain discharge from a 5 ha drainage (assuming 55% drainage recovery) with $\text{NO}_3\text{-N}$ and TP concentration of 5 mg L^{-1} and 0.3 mg L^{-1} , respectively.

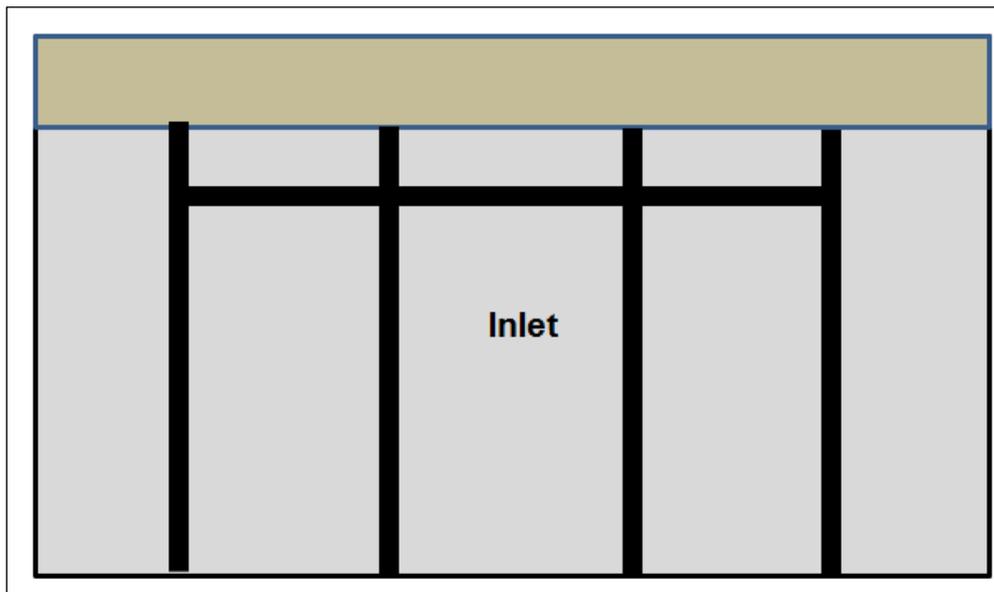


Figure 5.3: End view schematic of the pilot trial bioreactor.

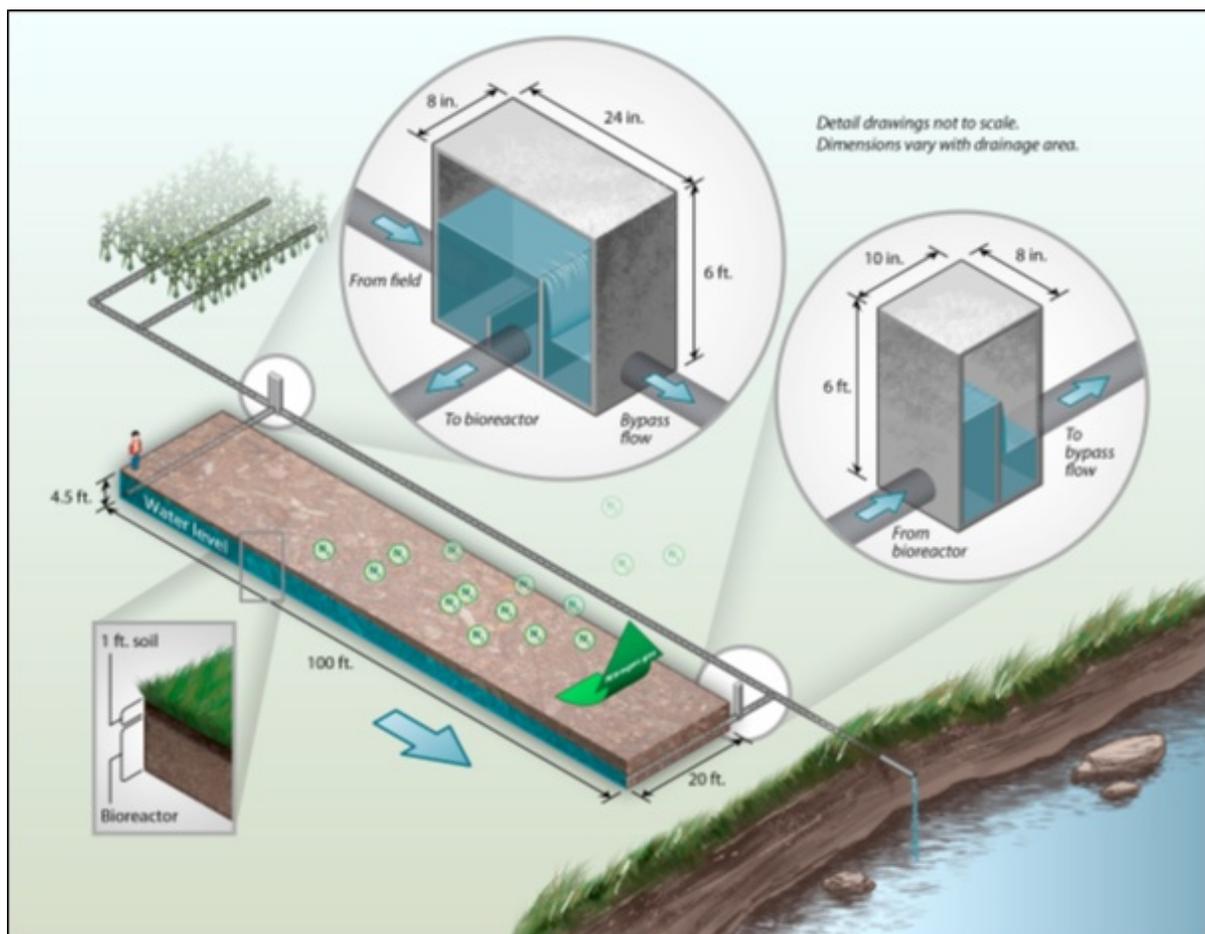


Figure 5.4: Schematic drawing of the general layout of the pilot trial system (Christianson and Helmers, 2011).



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