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Te Whatu Ao

Technical Recommendations to Guide Woodchip Bioreactor Design for Reduction of Farm Nitrate-N Losses

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Executive summary

Denitrifying woodchip bioreactors (woodchip bioreactors) — here defined as excavated beds filled with woodchips that intercept artificial drainage — are an interceptive, diffuse pollution mitigation tool with possible application in Aotearoa New Zealand. In the right conditions, they can provide a practical means for mitigating nitrate loads to waterways by reducing nitrate losses in agricultural drainage water. Hence, bioreactors are a tool that farmers can use alongside other on-farm nutrient management actions to reduce their environmental impact while maintaining production.

The aim of this report is to provide recommendations tailored to New Zealand conditions for the technical design, implementation, and expected performance of woodchip bioreactors for reduction of nitrate-N losses from artificial drainage to waterways.

This project was prepared for Regional Councils and Unitary Authorities as part of an Envirolink Grant 2617-ESRC509 coordinated by Environment Southland. To ensure the recommendations for application of on-farm woodchip bioreactors are scientifically robust and meet council land management staff needs, the project was guided by an advisory panel. The advisory panel included experienced regional council land management officers and industry representatives, along with scientists and engineers actively working on the application and performance assessment of woodchip bioreactors in New Zealand and the United States of America.

Artificial drainage (subsurface drainage and surface drainage ditches) is crucial for providing suitable soil water conditions that enable agricultural production in soils with low permeability and/or high groundwater levels. However, artificial drainage can act as a significant pathway by which nitrogen lost from land (leached as nitrate) enters surface waterways. To assist with identifying candidate sites for treating hot spots of nitrate, an overview of conventional subsurface drainage infrastructure and a summary of subsurface drainage flows and nitrate-N concentrations and loads are provided.

Clear documentation of nitrate load reductions from well-designed woodchip bioreactors provides users and regulators with realistic expectations of their performance in treating artificial drainage. A focused review of New Zealand field-scale case studies shows that performance is broadly consistent with average overseas results under comparable conditions. However, lower removal rates observed in some New Zealand trials likely reflect experimental designs, nitrate limitation, or preferential flow. As a practical benchmark, a saturated volumetric removal rate of ~3 g N m^{-3} d^{-1} represents a realistic, best-case performance for well-designed woodchip bioreactors in New Zealand, subject to temperature and operating conditions.

A summary of different approaches to characterise drainage flows and nitrate-N loads is provided, since an understanding of these parameters is needed as a basis for sizing the volume of woodchip required for bioreactors. For example, woodchip bioreactors can be sized using information on the median daily winter subsurface drainage flux (g nitrate-N d^{-1}), a realistic seasonal removal rate (g nitrate-N m^{-3} saturated woodchip d^{-1}), and a percentage nitrate-N load reduction target.

The process of designing the size of a bioreactor is illustrated for an example situation. The workflow provided uses the catchment area, modelled estimates of groundwater flows, information from published subsurface drainage studies, plausible drainage nitrate–N concentrations, and nitrate removal performance metrics based on New Zealand bioreactor case studies. It can be adapted to other situations with site–specific information to estimate woodchip bioreactor size (volume) based on specified criteria.

Case studies in this report highlight practical experience and results from the Waikato, Manawatū–Whanganui, Canterbury, and Southland in association with farmers, catchment groups, councils, and industry. Important factors to consider when designing a woodchip bioreactor include: the drainage conditions, desired treatment capacity, available space, climate, and the budget for construction and maintenance. It is also important to consider the bioreactor shape, operational water levels, and plumbing arrangement (i.e., design of the inlet and outlet structures to ensure even distribution and treatment of drainage water through the bioreactor). High flows greater than what can be intercepted by the bioreactor need to be routed via an external bypass to maintain drainage capacity in the field or paddock.

Overall, New Zealand and overseas field–scale case studies of woodchip bioreactors indicate that bioreactors treating artificial drainage are generally a low–risk mitigation option when properly designed and managed. While site–specific factors still need to be considered, experience to date suggests that appropriately designed and well–functioning woodchip bioreactors are unlikely to result in significant adverse environmental effects and can reduce nitrate loads from artificial drainage.

This guidance provides the technical basis for production and dissemination of a more end–user orientated practitioner guide in collaboration with rural industries and regional councils (to be developed subsequently). Correctly designed, constructed, and maintained woodchip bioreactors conforming to this guidance should be able to achieve the nitrate reduction estimates provided. To verify this and provide feedback to refine this guidance in the future, we recommend field testing and performance monitoring of woodchip bioreactors built according to this design guidance.

1 Introduction and project brief

Nitrate leached from agricultural land contributes to excessive nutrient enrichment of watercourses, rivers, wetlands, lakes, estuaries, and coastal marine areas. This degrades water quality, ecosystem health, mahinga kai (cultural food gathering), and recreation values of waterways over substantial areas of New Zealand (Ministry for the Environment & Stats New Zealand, 2026). Interceptive mitigations placed along flow-pathways to attenuate excess nitrogen before it enters receiving waterways are a useful complement to nutrient source control measures and in-field practices to manage nitrate pollution (Goeller et al. 2020a; Tanner et al. 2020).

The mitigation of agricultural diffuse pollution with edge-of-field interceptive tools can be an effective approach in the protection and rehabilitation of aquatic ecosystem values and functions, whilst maintaining agricultural production. Each interceptive mitigation has its niche in terms of the landscape fit, flow-path(s) intercepted, contaminant(s) treated, efficacy, associated co-benefits, relative cost, operation and maintenance requirements, longevity, and consenting requirements (Tanner et al. 2023). Denitrifying woodchip bioreactors (woodchip bioreactors) — here defined as excavated depressions or beds filled with woodchips that intercept artificial drainage — have proven effective in trials both overseas and in New Zealand. Woodchip bioreactors are a proven interceptive, diffuse pollution mitigation specifically designed for the treatment of nitrate-rich drainage waters (Addy et al. 2016). These engineered systems typically consist of excavated depressions or beds filled with woodchip, through which drainage water containing nitrate is routed. The woodchip acts as a reactive porous medium, like a water filter. Carbon in the woodchip fuels microbial activity, notably denitrifying bacteria that function under low-oxygen conditions and convert reactive nitrate to inert dinitrogen gas (Schipper et al. 2010a; Christianson et al. 2012).

Overseas, the design of woodchip bioreactors has been informed by practice standards (e.g., USDA-NRCS Conservation Practice Standard 605, USDA-NRCS CPS 605; United States Department of Agriculture 2020), national or regional guidance (e.g., Danish bioreactor guidelines, Plauborg et al. 2021; Queensland bioreactor guidelines, Wegscheidl et al. 2021), or expert advice. So far, implementation in New Zealand has been guided by researchers (e.g., Massey University Catchment Solutions 2026) and rural professionals (e.g., Constructing a bioreactor on a farm, Praat 2020), resulting in a range of different designs and performance outcomes. Results from field-scale monitoring studies of woodchip bioreactors treating subsurface drainage in New Zealand mirror overseas findings that reductions in annual nitrate loads can vary significantly ranging from ~10% (Goeller et al. 2019), ~50% (Hudson et al. 2019), 72-79% (Singh et al. 2025), to over 99% (Rivas et al. 2020) (see discussion in section 5.1). This wide range in performance reflects the challenges in designing bioreactors of appropriate size, suited for the local drainage setting and climate. It also reflects differences in the rigour of monitoring approaches, as well as yearly and seasonal variability in water temperatures, drainage flows, and nitrate loads at different sites (Addy et al. 2016; Christianson et al. 2021).

Measuring the nutrient load reduction performance of bioreactors receiving highly variable flows is logistically challenging and economically expensive. This is due to the highly dynamic nature of pollutant loading in farm drainage, which requires detailed measurements of drainage flows and nutrients in and out of the bioreactor, ideally over several drainage seasons, to calculate an average and range of nutrient load reduction performance.

Therefore, a major challenge in developing standardized design and performance specifications for bioreactors is deriving realistic estimates of achievable reductions in diffuse pollution given the relatively few field installations that have been monitored in New Zealand.

Developing standardised design specifications that can be tailored to local conditions will enable expected performance ranges to be forecast and guide effective design and implementation of woodchip bioreactors in local environmental conditions. This should improve efficacy and cost–effectiveness and reduce potential environmental risks of woodchip bioreactors. Also, such information enables farmers to assess the potential applicability and cost: benefit of woodchip bioreactors. It demonstrates to regulators and industry the efficacy of appropriately designed mitigations for achieving regulatory limits and addressing water quality and sustainability targets. Examples of technical design and performance specifications for interceptive, diffuse pollution mitigations in New Zealand include guidelines for surface flow constructed wetlands (Tanner et al. 2022), vegetated riparian buffers (McKergow et al. 2022), and detainment bunds (Paterson 2019). Guidance that aids in the design and informs the performance of mitigations is beneficial to multiple stakeholders and end–users:

- Council land management staff, industry, and rural advisors receive practical information to assist famers to achieve regulatory limits, and the council to achieve outcomes proposed in regional and catchment–based plans.
- Farmers are provided with trusted information, helping them fulfil their legal obligations by supporting the development of auditable, well–designed freshwater farm plans with effective mitigations.
- Council planners can access realistic information on the applicability and mitigation performance to inform plans and policies, including reducing regulatory hurdles and encouraging uptake of beneficial mitigations.

The aim of this report is to provide recommendations tailored to New Zealand environmental conditions for the technical design and expected performance of woodchip bioreactors for reducing losses of nitrate–N from agricultural drainage to waterways. To ensure the specifications are practical and meet farmer, regulatory, and industry needs, the project was guided by an advisory panel. The advisory panel consisted of experienced Regional Council land management officers and industry representatives, along with scientists and engineers actively working on the application and performance assessment of woodchip bioreactors in New Zealand and the United States of America.

This guidance will help standardise good practice for woodchip bioreactors, identify limitations for their applicability, and highlight future information needs. Finally, this guidance will inform the subsequent development of user-friendly practitioner guidelines in collaboration with rural industries and regional councils.

2 Purpose and scope of this guidance

The recommendations in the report have been prepared to provide rural land management advisors with sufficient information to incorporate woodchip bioreactors within farm nutrient management plans and for regulatory agencies to assess their expected efficacy for typical New Zealand conditions. This guidance focuses on simple, passive bioreactors for the interception and treatment of artificial drainage water from intensive agricultural land in relatively flatter areas (>0° and < 4° slope). Specifications are provided on their technical design and expected performance for reduction of nitrate–N losses from artificial drainage in New Zealand.

Why use woodchips? Although a wide range of different sources of slow–release organic carbon can be used to stimulate microbial denitrification (e.g., straw, corn cobs, cardboard, Cameron and Schipper 2010; Fan et al. 2022), recommendations are limited to systems using woodchip media. This is because woodchips are widely available in New Zealand, have good hydraulic conductivity, and can provide relatively steady rates of denitrification over long operational lives, approaching a decade.

This guidance draws primarily on field–scale evidence, since laboratory (Moghaddam et al. 2022) and mesocosm studies (Sukias et al. 2005; 2006) cannot fully reproduce the coupled hydraulic–biogeochemical conditions that influence treatment performance under field conditions (Christianson 2025). Although much is known about how woodchip bioreactors function, quantitative data on their treatment performance at field– and catchment–scales over multiple years is sparse. This is particularly so for pastoral land–uses and for relatively wet and mild oceanic climates like New Zealand's. Therefore, these technical guidelines are based on the expert opinion of those researching on woodchip bioreactors in New Zealand together with analysis of the much more extensive international data. Provisional estimates are provided for the reductions of nitrate expected under New Zealand conditions for woodchip bioreactors. However, the generalised guidance provided in this document needs to be adapted to local site conditions and applied judiciously, incorporating local knowledge, and appropriate experience and expertise.

2.1 Where do woodchip bioreactors fit?

The concentration of diffuse flows within artificial drainage networks creates opportunities for interception and treatment with woodchip bioreactors, constructed wetlands, saturated buffers (Jaynes and Isenhart 2014; Maintaining the Edge 2025), and controlled drainage (Frankenberger et al. 2024).

Compared to other interceptive diffuse pollution mitigations, woodchip bioreactors offer several advantages including a small footprint (e.g., 0.1 – 0.5% of the drainage area, equivalent to approximately 10–50 m² of bioreactor footprint per hectare of contributing catchment). They have minimal energy requirements and the ability to operate under variable flow conditions (e.g., flows typically ranging from 0.1 to 10 times the design flow rate) (Schipper et al. 2010a; Christianson et al. 2012).

Woodchip bioreactor beds can be located between the edges of agricultural fields and along the banks of streams, rivers and other waterbodies where they intercept subsurface drainage. Due to the flat topography of many of high nitrate areas, the treatment options mostly relate to subsurface tile drains which are ~900 mm below ground, making large earthworks prohibitively expensive. Therefore, woodchip bioreactors have a particular potential for low-lying, highly drained and modified agricultural land.

Bioreactors are best suited to landscapes and artificial drainage systems where water flows can be intercepted, slowed, and treated, particularly subsurface-drained land and managed surface drains with relatively low sediment loads. Bioreactors are generally less suited to landscapes dominated by organic soils and hump-and-hollow drainage systems, where artificial subsurface drainage is limited, nitrate concentrations may be relatively low, and denitrification within naturally saturated soils can already provide some nitrate attenuation. They fit well in lowland plains (e.g., Canterbury, Hawke's Bay) and in rolling or hilly pastoral areas (e.g., Manawatu, Taranaki, Southland) where subsurface drainage is present. They can also be used on groundwater-fed surface drains or ditches, provided these are not dominated by flashy storm runoff carrying high fine sediment loads. A key caveat is that surface drains typically require upstream treatment, such as a sedimentation pond, wetland, or detainment bund, as part of a treatment train. This reduces suspended sediment (and therefore clogging risk), dampens flow variability, and improves bioreactor performance. Even in subsurface drainage systems, particularly in more responsive or "flashy" catchments, adding simple hydraulic retention (e.g., controlled drainage structures or a small upstream pond) can stabilise inflows, increase contact time, and enhance treatment effectiveness.

Each woodchip bioreactor installation is somewhat unique, operating within a site-specific context defined by local geology, soil properties, climatic regime, and runoff dynamics. Woodchip bioreactors targeting subsurface drainage can be installed on existing or new drainage lines with catchment sizes varying from 1 to >100 ha. Bioreactors are typically situated on low-sloped land with slopes >0° and <4°, which is where most of the subsurface drainage is located (section 4 Artificial subsurface drainage'). Long-term studies indicate that these systems can maintain nitrate removal for around 9 and up to 15 years without further maintenance or carbon supplementation because woodchips degrade sufficiently slowly under anoxic conditions (Schipper and Vojvodic-Vukovic 2001; Christianson et al. 2012).

There are a range of factors that need to be considered before installing a woodchip bioreactor. Generally, bioreactors are unlikely to be an appropriate interceptive mitigation where:

- Artificial drainage is not present, or there are insufficient flows from the drainage (e.g., no flow most years except extremely wet years, or when flowing, flows are very intermittent).

- Median nitrate concentrations in the drainage are below 2 mg L^{-1} nitrate–N, based on at least three to five grab samples taken over winter. Bioreactors in such situations are unlikely to be economical (e.g., high–cost relative to low nitrate load reductions). Also, low influent nitrate concentrations may create nitrate–limiting conditions within the bioreactor, which may have unintended environmental trade–offs (e.g., methane and hydrogen sulphide production).
- Nitrate is not the predominant form of nitrogen in the drainage water: For example, where nitrogen is mainly present as ammonium or organic forms in the drainage from land–treated wastewater.
- Situations where bioreactors might receive high sediment loads (e.g., from flashy surface drains, surface runoff, or infiltration galleries that mix surface runoff with subsurface drainage).
- The immediate downstream receiving environment is sensitive to organic pollution or low oxygen concentrations, especially where bioreactors might discharge a relatively large volume of water into a small, sensitive waterway during summer when flows are low and temperatures are high.
- Ground slopes at the bioreactor's location are too flat to allow for gravity–based drainage flows and/or waterlogging around the immediate vicinity of the bioreactor is a non–negotiable risk (e.g., due to farm management needs, clashes with other infrastructure, or risk to stream bank stability). Pumping of drainage waters into and out of bioreactors may be an option in these cases, but pumping adds to the complexity and cost of the bioreactor design, construction and operation (Díaz–García et al. 2026). Advanced bioreactor design options for flat sites or pumped sites provides examples of advanced design options which do not rely on gravity–based drainage flows.
- Ground slopes at the bioreactor's location are too steep e.g., above 4°. As the elevation between the water inlet and outlet (hydraulic gradient) increases in the bioreactor, more water can be passed through the bioreactor. However, this means that a greater proportion of the woodchip is not saturated (and therefore not directly reactive). Also, excavations on steeper land require greater volumes of earthworks, which adds to overall cost.

2.2 Other bioreactor applications not covered by this guidance

Woodchip bioreactors have significant potential to supplement land treatment for a range of different wastewaters. However, subsurface drainage needs to be very carefully managed at these sites due to the high potential for inadequate treatment due to bypass flow and rapid transport to surface waters (Houlbrooke et al. 2004), with discharges prone to elevated concentrations of ammonium, organic matter, and faecal microbes. Different soil characteristics, levels of prior treatment (e.g., primary or secondary), and application methods and rates markedly affect the contaminant characteristics likely to be discharged to subsurface drains. In particular, substantial nitrification of ammonium–N (microbial conversion to nitrate–N) is required during prior treatment or soil infiltration of organic wastewaters for woodchip bioreactors to be an appropriate treatment or mitigation option for nitrate reduction in drainage from land treatment sites. Therefore, we consider that situations involving land treatment of wastewaters or direct treatment of wastewater (e.g., effluent from homes, glass houses, or livestock facilities (Rambags et al. 2016; Schipper et al. 2010b; Tanner et al. 2012) require specialist assessment and design. They are therefore not explicitly dealt with in the current report and excluded from the treatment performance predictions.

As well as optimising bed hydraulics and residence times, bioreactor performance can be enhanced to boost nitrate removal performance. Enhancing bioreactor performance is discussed further in section 8.5. This is an active area of research, requiring expertise, knowledge, and experience to implement in field conditions.

Finally, other types of bioreactors such as denitrification walls that intercept shallow groundwater (Burbery et al. 2020; Schipper and Vojvodić-Vuković 2001) or in-ditch bioreactors that are placed with streambeds (Burbery et al. 2025a; Burbery et al. 2025b) are out of scope.

3 Bioreactor treatment concepts

Woodchip bioreactors are passive treatment systems designed to remove nitrate from agricultural drainage water. Treatment occurs primarily through microbially mediated denitrification under anaerobic conditions. Bioreactor treatment performance (nitrate load reduction) is governed by the need to balance nitrogen loading against the system's capacity for denitrification. Nitrogen loading is a function of drainage flow and nitrate concentration, whereas denitrification rates depend strongly on temperature and carbon availability. This creates a highly dynamic, multi-variable system that is inherently challenging to optimise through design alone. At short (hourly) timescales, treatment efficacy is controlled by how quickly water and solutes enter and pass through the bioreactor, while at seasonal to interannual scales, variability in rainfall and temperature drives substantial fluctuations in performance, even at a single site. Overall, performance is governed by interactions among nitrate loading, hydraulic conditions, temperature, and carbon availability. The following sections describe these processes and controls.

3.1 Denitrification, nitrate loading, and mass removal dynamics

The primary treatment mechanism involves creating an anaerobic environment where denitrifying bacteria use the woodchips as a carbon source while converting water soluble nitrate to a gaseous form of nitrogen (denitrification). Denitrification performance of a bioreactor is influenced by key environmental and design factors, including hydraulic retention time (HRT), influent nitrate concentration, temperature, and woodchip carbon quality. For example, low nitrate concentrations may limit treatment, while aged woodchips can reduce carbon availability and performance over time. Water temperature is a critical control on biological activity.

Because carbon is non-rate-limiting in a woodchip bioreactor and nitrate is typically plentiful, for design purposes, nitrogen removal is commonly represented as a zero-order process (i.e., constant) for design purposes (Schipper et al. 2010a). Accordingly, nitrate loading — the product of inflow and influent concentrations — influences treatment performance (and can be used to inform bioreactor designs). Bioreactors are most effective when treating drains with consistently elevated nitrate concentrations and flows, as these conditions sustain removal rates and maximise total nitrogen removal. In contrast, when flows are intermittent or seasonally reduced, nitrate loading declines and overall mass removal is lower, even though short-term removal rates may increase due to greater availability of labile carbon.

Bioreactors have different operational states (e.g., start-up, steady-state, reducing conditions – see discussion in section 8 on operation and maintenance). They also operate under fluctuating environmental conditions, which can include drying-rewetting cycles and flashy or episodic nitrate loading. Episodic nitrate loading occurs in New Zealand pastoral subsurface drainage networks (Rivas et al. 2020; Barkle et al. 2021; Singh et al. 2025).

'Flashy' or episodic nitrate loading can be a common occurrence due to seasonally fluctuating water tables and drainage water nitrate concentrations that change rapidly in response to rain events, especially in rolling country. In contrast, drainage flows can be more consistent and almost year–round in areas that are wetter (e.g., Southland, West Coast of the South Island) and/or where drains are groundwater–fed (e.g., coastal Canterbury, Rangitīkei Sand Country, Wairarapa).

So long as the influent nitrate load exceeds the bioreactor's mass removal rate, the mass removal rate of nitrate is relatively constant per volume of wetted woodchip media (g NO_3-N m^{-3} d^{-1}). Therefore, stable flows and concentrations favour highest N mass removals. However, when drainage stops periodically or slows down seasonally, the mass removal rate may be somewhat higher due to higher organic carbon availability, although overall mass removal will be lower due to decreased loading.

3.2 Impact of hydraulic conditions and temperature on nitrate removal

Hydraulic conditions strongly influence treatment. Variability in flow ("hot spots" and "hot moments") can make it difficult to size bioreactors to treat peak flows effectively. Increasing the proportion of flow routed through the bioreactor (i.e., reducing bypass) increases nitrogen mass removal but reduces apparent percentage removal, reflecting the finite treatment capacity of the system (Plauborg et al. 2023; Christianson et al. 2026). Larger or more intensively farmed catchments typically generate higher and more sustained nitrate loads, but bioreactors remain constrained by the long–term carbon supply from the woodchips.

Changes in flow and loading influence internal biogeochemical conditions. High flows and loading reduce HRT and decrease nitrate removal efficacy (percentage load reduction), whereas lower flows (and loading) can increase percentage nitrate load reduction but shift conditions beyond optimal denitrification. This can lead to release of other redox products due to longer HRT (e.g., hydrogen sulfide, which is toxic, corrosive, and smells like rotten eggs). This phenomenon is an example of 'pollution swapping', which is discussed further in section 5.5.

Lower temperatures slow microbial processes and reduce nitrate removal rates. This relationship is commonly described using Q_{10}, the factor by which a reaction rate increases when temperature rises by 10°C, or the modified Arrhenius coefficient (θ), the multiplicative increase in reaction rate for each 1°C rise in temperature. The expected influence of temperature on nitrate–N removal performance for mature field–scale woodchip bioreactors receiving variable flows of artificial drainage is shown in Figure 3–1. These estimates assume median zero order reaction rates at 20°C (k_{20}, i.e., the reference removal rate at 20 °C) of 3 g N m^{-3} d^{-1} with a potential range (shaded zones) of 1.5 – 4.5 g N m^{-3} d^{-1} and a modified Arrhenius temperature coefficient (θ) of 1.08 (corresponding to a Q_{10} of 2.15 as reported by Addy et al. 2016).

They reflect the generally lower mass removal performance reported so far from the limited studies of field-scale bioreactors carried out in New Zealand (discussed later in section 5.2), than have commonly been reported from other areas of the world (e.g., Addy et al. 2016; Christianson et al. 2021). Across New Zealand, shallow groundwater and subsurface drainage temperatures broadly reflect mean annual air temperatures (approximately 10–16°C from south to north), suggesting that regional differences in treatment performance on the order of up to 40% are plausible and should be factored into performance expectations. However, when assessing system behaviour, it is the temperature during active drainage periods that are most relevant, rather than annual averages.

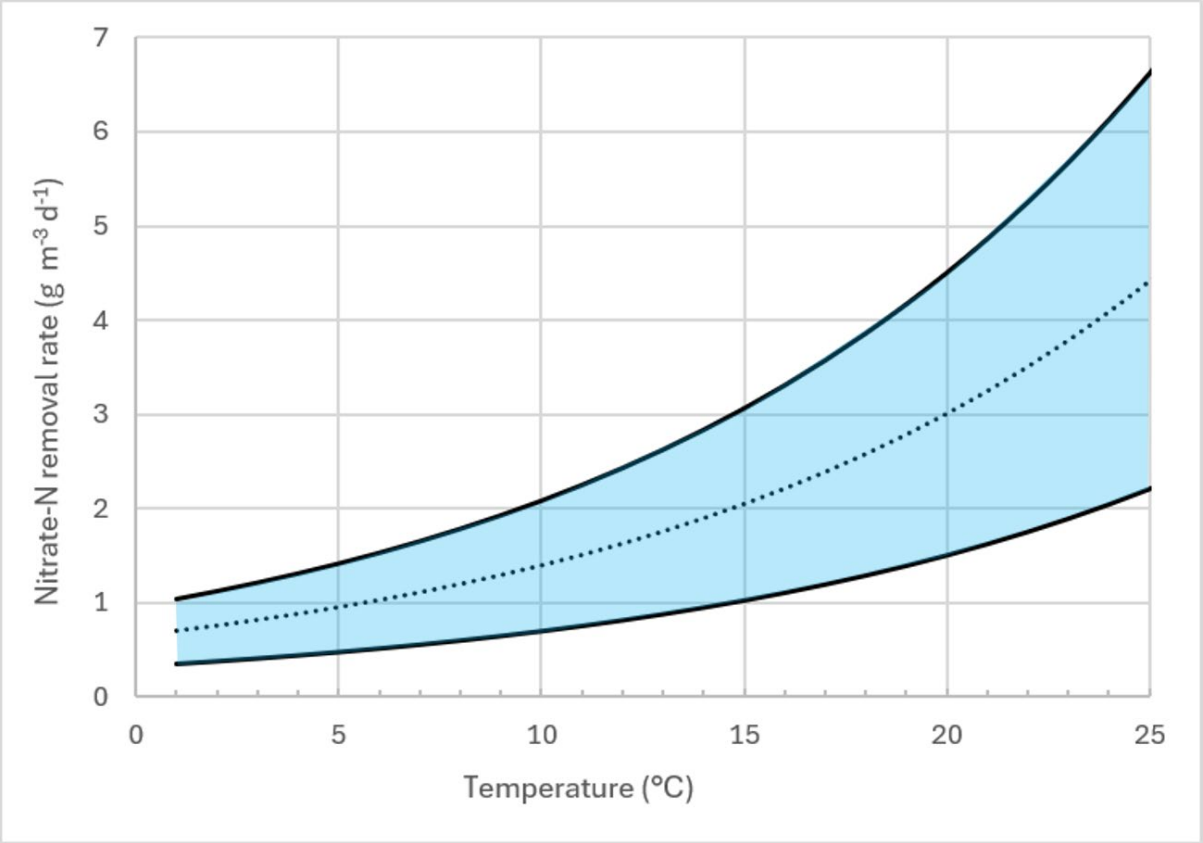


Figure 3-1: Expected nitrate-N removal performance (saturated volumetric removal rate in g N m^{-3} d^{-1}) in relation to water temperature (°C) for mature field-scale woodchip bioreactors receiving variable flows of subsurface drainage under New Zealand conditions. The shaded zone shows the estimated range and the dashed line the estimated median mass removal rate.

4 Artificial subsurface drainage

4.1 What is subsurface drainage?

Subsurface drainage is crucial for providing suitable soil water conditions that enable agricultural production in soils with low permeability and/or high groundwater levels. It is used to remove excess water from within the soil profile to prevent waterlogging, improve soil aeration, and enhance pasture or crop growth. Subsurface drains are also used on farms to intercept and remove water from seeps, springs, and small temporary streams. In pasture systems, this drainage improves soil strength and bearing capacity, improve plant growth, supporting higher stocking rates, and increasing periods for cultivation and movement of vehicles.

Subsurface drainage is primarily achieved by installing a network of perforated plastic pipes (or historically clay tiles) below the soil surface to lower the water table (Figure 4-1). The drainage discharges into surface drainage ditches or other receiving waterways (Figure 4-2). In low permeability, heavy clay soils, subsurface drainpipes are commonly supplemented by mole channels ploughed perpendicular to the pipe drains to channel water into them (Monaghan 2014). Mole ploughs consist of a torpedo shaped foot at the end of a shank that is drawn though the clay soil behind a tractor at the correct soil moisture level, forming a pipe through the clay. The shank of the plough creates cracks in the soil from the surface down into the mole drain which allow water to percolate down into the drains. This method is common in Southland soils like Pukemutu and Waikoikoi silt loams, where dense subsoils restrict water draining naturally (Pearson 2015).

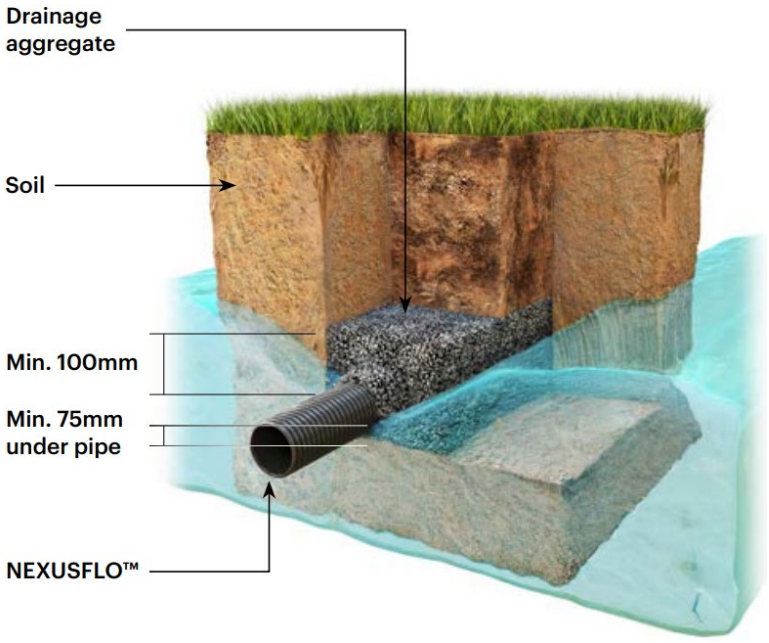


Figure 4-1: Schematic 3D cross-section of an installed subsurface perorated plastic pipe drain, laid in a gravel envelope. If mole drains are laid they will intercept the gravel envelope feeding lateral flows into the pipe drain. The gravel envelope improves hydraulic connectivity to the pipe and helps reduce blockage of perforations, although it is not always used in New Zealand agricultural drainage systems due to cost and installation practicality. Diagram credit: https://www.iplex.co.nz/assets/Uploads/IPL0233-Nexus-Design-Install-Guide.pdf.



Figure 4-2: Subsurface drain discharging to a surface drainage ditch or waterway. Photo credit: Chris Tanner.

Where land is too flat for tile drainage, farmers sometimes use soak holes to stop water pooling on the surface. A soak hole is a deep pit filled with gravel that lets water quickly drain through the restrictive layer below. For example, this approach is common on Southland's flat, slowly-permeable soils, such as those around Edendale (Pearson 2015). In some situations, these soak holes may also form surface inlets that connect to subsurface drainage networks, which may increase sediment loads to the drainage.

On land with a constant slope or no undulations, subsurface drainage is often installed in grids, often in conjunction with open ditches to lower the water table to a uniform depth (Figure 4-3). In contrast, on sloping or undulating land, subsurface drainage is often laid in hollows or swales that follow the natural contour of the landscape. The depth that the subsurface drainage is installed at depends on the depth of the water table and the amount of fall necessary to drain the area. For example, in Southland, subsurface tile drains are typically found at 60-80 cm depth, spaced between 20-100 m apart, whereas mole drains are typically ploughed at 30-45 cm depth and can be as close as 2 m apart (Houlbrooke and Monaghan 2009).

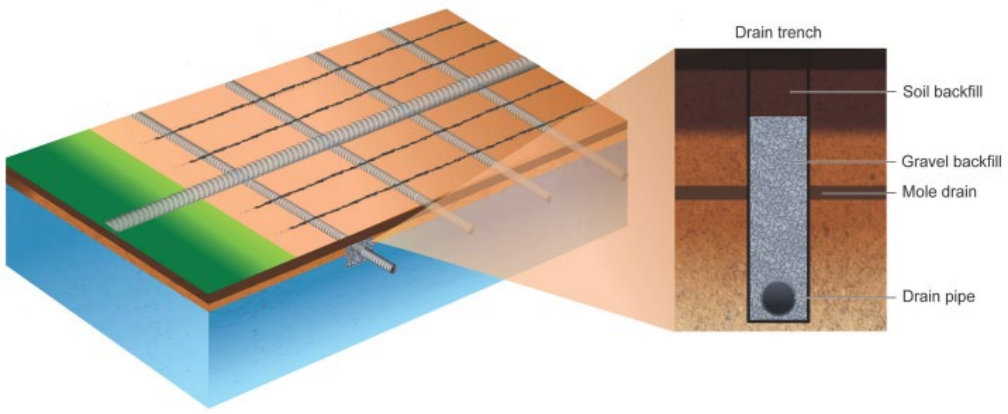


Figure 4-3: Example of a grid layout of mole drains. Mole drains (black) direct flows to subsurface drains (narrow grey) set in gravel filled-trenches, intercepted by a central pipe drain (wide grey), from Ghane (2025).

Subsurface drainage systems are generally designed to achieve a specific drainage coefficient, tailored to the intensity of proposed land-use (Skaggs 2017). Drainage coefficients are usually expressed as the depth of water in mm which must be removed in a 24-hour period. Recommendations for pastoral farming in New Zealand are shown in Table 1.

Table 1: Recommended drainage coefficients for various land uses and associated design flows. Modified from Bowler (1980) and iPLEX (2025).

Land-use	Drainage co-efficient (mm d^{-1})	Drainage capacity (m^3 $ha^{-1}d^{-1}$)	Drainage capacity (L s^{-1} ha^{-1})
Dairy farm with winter grazing	25	250	2.9
Dairy farming without winter grazing	15-20	150-200	1.74-2.31
Intensive sheep and beef grazing and cropping	10-15	100-150	1.57-1.74

The drainage system capacity will vary significantly depending on the network design, drainpipe density, diameter, internal roughness (smooth or corrugated) and slope (see nomographs in Appendix A). In practice, the actual efficiency, yields, and flow rates from a drainage system (and the contaminant concentrations and loads) will also vary markedly. These differences in drainage efficiency are influenced by soil characteristics, rainfall intensity and duration, antecedent soil moisture, land management (e.g., time since grazing, stocking rate, fertiliser and supplementary feed regimes), and the age and state of functionality of the drainage system. Therefore, estimating drainage system flow rates and contaminant loads based on this information is challenging.

4.2 Flow and contaminant losses via subsurface drainage

Subsurface drainage alters hydrologic pathways, creating rapid transport routes for nutrients and other contaminants to agricultural waterways (Kennedy et al. 2012; Tomer et al. 2003; Williams et al. 2015). Although this can decrease the export of sediment and associated contaminants, it often increases losses of mobile dissolved nutrients such as nitrate-N and orthophosphate (Gilliam et al. 1999; King et al. 2015; Monaghan et al. 2016; Norris et al. 2025; Tomer et al. 2003). Subsurface drains can also convey fine sediment, agrichemicals, and faecal pathogens (Blann et al. 2009; Gramlich et al. 2018). By bypassing natural attenuation in soils and riparian zones, subsurface drainage becomes hot-spot and key pathway for nitrate export from artificially drained land (Armstrong and Garwood 1991; Blann et al. 2009; Ritter and Shirmohammadi 2001).

Looking at the combined data from across New Zealand studies of subsurface drainage, excluding irrigated sites, provides useful information to describe 'typical' drainage characteristics and their variability (Tanner et al. in-prep). Differences in rainfall between regions and over different years at individual sites accounts for some of the spread in the data. The overview of compiled subsurface drainage characteristics reported for New Zealand pastoral agriculture (Tanner et al. in prep.) shows that 'typically' (based on the interquartile range from 44-75 site-years) flow occurs for 6-9 months y 1, involves 175-347 mm y 1 of subsurface drainage, representing 19-37% of annual rainfall. Nitrate-N concentrations typically range from 12-16 mg L 1 and nitrate-N yields (area-weighted loads) from 12-32 kg ha y 1 (Table 2).

Table 2: Summary of compiled subsurface drainage characteristics reported for New Zealand pastoral agriculture. (Tanner et al. in prep). Irrigated sites were excluded, with each site-year of data used as a datapoint. Study sites were predominantly located in intensive agricultural regions in Waikato, Manawatu, western Otago, and Southland.

Statistic	Annual subsurface drainage (mm y^{-1})	Annual drainage as a percentage of rainfall (%)	Annual drainage period (months y^{-1})	NO_3–N concentration (mg L^{-1})	Annual NO_3–N yield (kg ha^{-1} y^{-1})
Overall median	248	26	6	9.3	19.1
Overall average	289	29	7.3	9.8	24.7
Overall st dev	170.3	14	2.4	5.8	17.4
Number of site-years	75	68	44	52	67
25%ile	175	19	6	12.0	11.9
75%ile	347	37	9	16.1	32.4
Min	51.9	11	3	1.6	1.8
Max	853	86	12	35	82.4

4.3 Locating subsurface drainage

Many agricultural landscapes are likely to incorporate subsurface drainage, especially those with poorly drained soil on flat to gently sloped land. Manderson (2018) estimated that 5.4 million ha or 20% of the total land area of New Zealand is suitable for artificial drainage (which includes surface and subsurface drainage), with a moderate likelihood that artificial surface and/or subsurface drainage is present on 2.5 million hectares of land (9% of the total land area)[1]. The national-scale map of likely areas with artificial drainage are shown in Appendix B (New Zealand Map from Manderson 2018). Land-uses with artificial drainage present were mostly high intensity pastoral farming, but also included arable cropping, flowers, fruit and vegetable growing, plant nurseries, and viticulture.

Most subsurface drainage in New Zealand is likely to be installed on land with slopes between 0° and 4° (Manderson 2018). To verify the national-scale mapping work, Manderson (2018) evaluated tile drain information from Southland and found that artificial drainage was mostly on flat to undulating slopes (88% of tile lines occurred on slopes 0–7°), noting that in Southland, drains bay be on steeper slopes than typical elsewhere in New Zealand.

[1] At the moderate confidence level (0.55 fuzzy probability value). Suitable areas for artificial drainage were identified based on soil drainage class, soil permeability class, and expert drainage suitability ratings. Likely areas with artificial drainage present were identified from suitable areas based on the Agribase farm dataset (proxy for land-use type and intensity), slope, and the proximity of receiving drain networks (to transfer water away from drained areas).

In-between slopes (rolling, 7–15°) carried the most uncertainty, and strongly rolling country where slopes approached 20° was least likely to have subsurface drainage. However, in hill country areas, artificial subsurface drainage may also be installed to drain seeps and springs (Pearson 2015).

Finding subsurface drains is difficult because they are buried and signs of their presence may not be obvious from the surface. Bidwell (1978) proposed a ~40-m spacing as a starting point for determining drain locations. At the farm- or field-scale, finding subsurface drainage infrastructure is conventionally done using paper maps and the knowledge of local drainage contractors and landowners (Rahmani et al. 2023). The locations of tile drain outlets can be verified by walking the length of waterways, although the outlets are frequently partially buried or obscured by vegetation (Figure 4-4). Listening for running water (if the drainage is flowing) or using a probing rod or shovel to find the outlets are useful methods. In some cases, old subsurface drainage infrastructure may be present. These may still be functioning well or partially or completely blocked, which creates some uncertainty around the intensity and extent of the drainage network that feeds to a particular outlet and its likely discharge characteristics.

Aerial photos or drone-deployed remote sensing techniques can be useful to identify earthworks scars or linear patterns over the drainpipes due to small changes in elevation, soil moisture, soil temperature, or grass or crop growth (Allred et al. 2020; Tilahun et al. 2021; Woo et al. 2019). Other non-invasive, non-destructive methods include the use of ground-penetrating radar (GPR), which has been used to map clay pipe and corrugated plastic subsurface drainage lines in a variety of soil types overseas (Allred et al. 2018, Koganti et al. 2021). In New Zealand, GPR has also been trialled successfully to detect subsurface drainage on silt loam and clay loam soils in the Waikato (Goeller et al. 2020b) and Mottled Fragic Pallic and Mottled Firm Brown soils in Southland (Deuss et al. 2023). Remote sensing techniques offer different advantages, accuracies, and scalability to identify locations of subsurface drainage.

Unfortunately, however, there is no dependable framework available to guide selection of the appropriate remote sensing method to identify the location of subsurface drainage for specific soil types and drainage installations, and implementation of these methods is time-intensive and expensive.



Figure 4-4: Plastic subsurface drain outlet partially covered by drain vegetation. The subsurface drainage runs along the bottom of a swale in a gently undulating paddock in the Toenepi catchment, Waikato. Photo credit: Brandon Goeller.

5 Woodchip bioreactor pollution reduction performance

5.1 Explanation of nitrate treatment performance metrics

There are several common ways to evaluate nitrate removal (attenuation) performance or treatment efficacy of a bioreactor. The term efficacy rather than efficiency is used because it focuses on a bioreactor's ability to achieve the desired change in contaminant loads, rather than the resources or effort required to do so. Treatment efficacy can be positive or negative, indicating whether the bioreactor reduces contaminant mass or, in some cases, results in an increase through the treatment process. Performance estimates based on load reductions or mass removal of nitrate–N are generally preferred, as they account for both drainage volume and influent concentrations, which can vary seasonally and annually. However, some studies report changes in nitrate concentrations instead, as these can be directly related to toxicity and ecological effects in the receiving environment.

Nitrate removal efficacy (NRE) expresses the percentage reduction in nitrate concentration or load across the bioreactor. It is a simple and widely used metric that highlights treatment effectiveness from a water quality perspective. Note that concentration units of mg L^{-1} and g m^{-3} are numerically equivalent.

Concentration removal efficacy

$$\mathrm{NRE_{conc}}(\%) = \frac{C_{\mathrm{in}} - C_{\mathrm{out}}}{C_{\mathrm{in}}} \times 100$$

Where:

- C_{in}= influent nitrate–N concentration (g N m^{-3})
- C_{out}= effluent nitrate–N concentration (g N m^{-3})

NRE based on concentration only should ideally be calculated using flow–proportional concentrations. Flow–proportional concentrations account for the coupling between flow and concentration dynamics, ensuring that high–flow periods, which often contribute a disproportionately large fraction of the total nutrient load, are appropriately represented in performance calculations. Accordingly, this requires simultaneous measurement of flow and nitrate–N concentrations entering and exiting the bioreactor.

Flow–proportional influent and effluent concentrations

$$C_{\mathrm{fw}} = \frac{\sum(C_i Q_i)}{\sum Q_i}$$

Where:

- C_{fw}= flow–weighted concentration (g N m^{-3})
- C_i= discrete concentration measurements at timestep i (g N m^{-3})
- Q_i= flow volume associated with timestep i (m^3)

When concentrations are calculated on a flow–proportional basis, concentration–based nitrate removal efficacy is mathematically equivalent to load–based nitrate removal efficacy. The load is the integrated mass flux over a specified period. Loads can be estimated as:

$$L = C_{\text{fw}} \times Q$$

or, for time–resolved data:

$$L = \sum(C_i\, Q_i)$$

Where:

- L= nitrate–N load over the specified period (g or kg N)
- C_{fw}= flow–weighted concentration (g N m^{-3})
- Q= total discharge over the period (m^3)
- C_i, Q_i= concentration and flow volume at time step i

Changes in loads can be expressed either relative to the load that the bioreactor intercepted (relative load reduction) or considering untreated bypass flow (total or edge–of–field load reduction). Relative nitrate removal efficacy represents the percentage reduction in nitrate load for the portion of flow that is treated by the bioreactor only. It reflects the internal treatment performance of the system, independent of how much water bypasses it.

Relative nitrate load removal efficacy

$$NRE_{relative}(\%) = \frac{L_{in} - L_{out}}{L_{in}} \times 100$$

Where:

- L_{in}= nitrate–N load entering the bioreactor over the assessment period (g or kg N)
- L_{out}= nitrate–N load leaving the bioreactor over the assessment period (g or kg N)

Edge–of–field NRE reflects the actual reduction in nitrate export from the drainage system, which depends strongly on the proportion of flow treated versus bypassed.

Edge–of–field nitrate load removal efficacy

$$NRE_{total}(\%) = \frac{L_{in} - (L_{out} + L_{bypass})}{L_{in}} \times 100$$

Where:

- L_{in}= Edge–of–field nitrate–N load generated by the contributing drainage area before treatment (g or kg N). Note that L_{in} includes both the load directed through the bioreactor and the load carried in bypass flow.
- L_{out}= nitrate–N load discharged from the bioreactor effluent over the assessment period (g or kg N)
- L_{bypass}= nitrate–N load in untreated bypass flow over the assessment period (g or kg N)

Although they are conceptually simpler to communicate, percentage–based performance metrics do not reflect differences in the magnitude of the load intercepted by the bioreactor or how removal scales with catchment area or bioreactor size. Consequently, percentage–based metrics provide limited information about the absolute mass of nitrate removed. In contrast, reporting load reductions as area–weighted loads (i.e., yields, kg N ha^{-1} yr^{-1}) represents the mass of nitrate removed per area of catchment over time. Assessing performance in this way helps to standardise load reductions due to different sized catchments and different nitrate loads across different site–years. Also, since farmers are used to accounting for nutrient inputs and losses as yields, this metric is useful for informing farm management decisions, including implementing bioreactors to reduce nutrient losses.

Area–weighted nitrate load reduction (yield)

$$Y = \frac{L}{A}$$

or, expressed explicitly for removal:

$$Y_{\text{removed}} = \frac{L_{\text{in}} - L_{\text{out}}}{A}$$

Where:

- Y= annual area–weighted nitrate load or yield (kg N ha^{-1} yr^{-1})
- Y_{removed}= annual area–weighted nitrate load removed (kg N ha^{-1} yr^{-1})
- L= annual nitrate–N load (kg N yr^{-1})
- L_{in}, L_{out}= influent and effluent nitrate–N loads during the assessment period (kg N yr^{-1})
- A= contributing catchment area (ha)

To compare performance across bioreactors with different designs, media types, or operating conditions, volumetric mass removal rates are useful. The saturated volumetric removal rate reveals the mass of nitrate–N removed per volume of reactive wood (i.e., performance is normalised for the portion of the bioreactor that is hydraulically active).

Saturated volumetric removal rate

$$R_{\text{sat}} = \frac{L_{\text{in}} - L_{\text{out}}}{V_{\text{sat}}}$$

Where:

- R_{sat}= saturated volumetric removal rate (g N m^{-3} d^{-1})
- $L_{\text{in}} - L_{\text{out}}$= nitrate–N mass removed per unit time (g N d^{-1}), calculated as the difference between influent and effluent loads. Annual loads (kg N yr^{-1}) can be converted to daily rates by multiplying by 1000 and dividing by the number of days per year with flow.
- V_{sat}= saturated (wetted, reactive) volume of woodchip media (m^3)

Expressing nitrate removal per unit of saturated volume provides insight into treatment efficacy of the reactive media, while normalising removal to total installed volume allows evaluation of whole–system performance and cost–benefit. The total volumetric removal rate provides insight into whole–system performance relative to the total volume of media installed.

Total volumetric removal rate

$$R_{\text{total}} = \frac{L_{\text{in}} - L_{\text{out}}}{V_{\text{total}}}$$

Where:

- R_{total}= total volumetric removal rate (g N m^{-3} d^{-1})
- V_{total}= total installed volume of woodchip media (m^3)

5.2 Synthesis of nitrate load reduction performance from New Zealand bioreactor case studies

Nearly a dozen field trials of woodchip bioreactors treating agricultural drainage from pastoral land have been completed in the Waikato, Wairarapa, Manawatū–Whanganui, Canterbury, and Southland regions. Those trials have provided useful information on the appropriate design and performance expectations of bioreactors tailored for New Zealand conditions. To determine a conservative range of realistic nitrate load reduction performance metrics, we evaluated data from a focused subset of monitored, field-scale, New Zealand woodchip bioreactor case studies that treated subsurface drainage. Table 3 provides an overview of New Zealand case studies of woodchip bioreactors treating nitrate from artificial drainage, based on twelve site-years of data from five bioreactors. These bioreactors were monitored after an initial flushing period of several months to a year.

Monitoring was completed for two to three years with varying rigorousness ranging from ~monthly sampling and spot measurements of flows (Goeller et al. 2019), weekly sampling and continuous water level monitoring (Rivas et al. 2020; Singh et al. 2025), and continuous water level and nitrate concentration monitoring (Hudson et al. 2019; Rivas et al. 2023). The 2017 site-year for the Waikato bioreactor (Rivas et al. 2020) was excluded when comparing performance across bioreactors and site-years. In that year, inflow restriction reduced loading and led to over-treatment, an issue that was corrected in subsequent years. The five case studies are described in greater detail in Appendix C.

Summary of percentage load reductions:

- Across the twelve site-years from five bioreactors, the annual relative bioreactor nitrate load reductions ranged widely from 10–81% (Goeller et al. 2019; Hudson et al. 2019; Rivas et al. 2020; Singh et al. 2025).
- After averaging across the site-years for each bioreactor, the New Zealand mean and range of annual relative load reductions from the five bioreactors were 55% and 10–78%, respectively (Table 3).
- When considering the edge-of-field drainage nitrate load (including any untreated bypass flows), the percentage load reductions are smaller, e.g., the range of annual edge-of-field nitrate-N load reductions across all twelve site years was 8% in the Coastal Rangitikei and Upper Manawatū trials (Singh et al. 2025) to 74% in the Waikato trial (Rivas et al. 2020). However, it is worth noting that these were constructed as research trials, i.e., the bioreactors were not designed to optimise nitrate load reductions at the study farms.

- After averaging across the site–years for each bioreactor, the New Zealand mean and range of edge–of–field nitrate load reductions from the five bioreactors were 33% and 10–61%, respectively. This is largely due to interannual differences in the nitrate removal efficacy and the proportion of untreated bypass flow (Table 3).

Summary of area–weighted loads (i.e., yields, kg N ha^{-1} yr^{-1}):

- Subsurface drainage nitrate–N yields were reported for six site years of data, ranging from 2.8 kg N ha^{-1} yr^{-1} (Hudson et al. 2019) to 24.4 kg N ha^{-1} yr^{-1} (Rivas et al. 2023), with an overall average of 10.4 kg N ha^{-1} yr^{-1} (Table 3). The nitrate–N yields in the Southland trial were low due to both low drainage flows and low inflow concentrations with a three year median of 2.2 mg nitrate–N L^{-1} (Hudson et al. 2019). Due to uncertainty in the size of the groundwater catchment feeding the drainage, Goeller et al. (2019a) and Singh et al. (2025) did not report drainage nitrate–N yields from their studies.

- Across ten site years from four bioreactors (excluding Goeller et al. 2019), the annual nitrate–N yields removed by bioreactors were between 1.4 kg N ha^{-1} yr^{-1} in Southland (Hudson et al. 2019) and 9.2 kg N ha^{-1} yr^{-1} in the Waikato (Rivas et al. 2020).

- After averaging across the site years for each bioreactor, the New Zealand mean and range of annual yields of nitrate–N removed from the four bioreactors were 3.6 kg N ha^{-1} yr^{-1} and 1.8 – 7.7 kg N ha^{-1} yr^{-1} (Table 3).

Differences in the size of bioreactor (expressed as either the volume of saturated woodchips or the total volume of woodchips) can also mask underlying differences in performance. For example, differences in bioreactor sizes, changes in the volumes of flow intercepted, and water level management in the bioreactor influence how much of the woodchip is saturated and reactive. In practice, the saturated volume of woodchips is typically much less than the total volume of woodchips available in the bioreactor. For example:

- Across eleven site years from five bioreactors, the percentage of the total volume of woodchips that was saturated ranged from 53 or 54% in the Coastal Rangitikei and Canterbury bioreactors (Goeller et al. 2019; Singh et al. 2015) to 92% in the Upper Manawatū bioreactor (Singh et al. 2015). For one site year of data, saturated volume could not be estimated due lack of monitoring water levels in the bioreactor (Singh et al. 2015).

Generally, measuring water levels accurately over the length of the bioreactor and relating this to the saturated volume under changing flow conditions is technically challenging, with further uncertainties in estimates of saturated volume caused by to the different measurement techniques and frequences across the studies.

However, knowing the saturated woodchip volume allows is key to calculating saturated volumetric mass removal rates to compare performance between bioreactors that differ in design, media type, or operating conditions. For example:

- When nitrate attenuation performance is assessed as a saturated volumetric removal rate (i.e., the amount of nitrate–N removed per cubic meter of saturated woodchip per day of flow), annual removal rates ranged from 0.54 in Southland (Hudson et al. 2019) to 3.69 g N m^{-3} d^{-1} in the Upper Manawatū (Singh et al. 2025) across all twelve site–years.
- After averaging across ten site–years for four bioreactors (not including Goeller et al. 2019), the New Zealand mean and range of saturated volumetric removal rates were 1.5 g N m^{-3} d^{-1} and 0.9–2.9 g N m^{-3} d^{-1}, respectively (Table 3). The extraordinarily high saturated volumetric removal rates >10 g N m^{-3} d^{-1} reported by Goeller et al. (2019a) were an outlier amongst the New Zealand studies, so they were not considered representative and are not included in Table 3.
- When nitrate attenuation performance is expressed as a total volumetric removal rate (i.e., the amount of nitrate–N removed per cubic meter of woodchip per year, not considering the saturated volume or number of days of flow), nitrate–N removal rates were somewhat lower. E.g., across ten site years for four bioreactors (excluding Goeller et al. 2019), the total volumetric removal rate was between 0.14 g N m^{-3} d^{-1} in the Waikato (Rivas et al. 2020) and 2.99 g N m^{-3} d^{-1} in the Upper Manawatū (Singh et al. 2025).
- After averaging across ten site years for four bioreactors (not including Goeller et al. 2019), the New Zealand mean and range of total volumetric removal rates were 1 g N m^{-3} d^{-1} and 0.2–2.3 g N m^{-3} d^{-1}, respectively (Table 3).

As a practical guide, land managers can expect about ~3 g N m^{-3} d^{-1} as a realistic, best–case saturated volumetric removal rate for well–designed woodchip bioreactors in New Zealand (subject to temperature and operating conditions; Figure 3–1).

Other woodchip bioreactor designs in Canterbury intercepting groundwater in a woodchip wall (Burbery et al. 2020) and in an in–ditch woodchip bed (Burbery et al. 2025a) with constant flow indicate woodchip reaction rates of around 2–3 g N m^{-3} of saturated woodchip per day. Those reaction rates align with results from the Upper Manawatū bioreactor (Singh et al. 2025) and international data evaluated by Addy et al. (2016).

5.3 Comparison of New Zealand and international case studies of bioreactor nitrate load reduction performance

When compared to overseas field-scale trials of woodchip bioreactors treating nitrate from subsurface drainage (Addy et al. 2016; Christianson et al. 2021, 2026; Plauborg et al. 2023), the New Zealand case studies performance metrics are generally within the range of average performance, e.g., percentage of total flow treated, total percentage load reductions, and volumetric removal rates. Notably however, the New Zealand bioreactors, tended to have slightly lower volumetric mass nitrate-N removal rates. These lower saturated mass removal rates are likely due to sub-optimal design (i.e., bioreactors were used for testing different treatment conditions), nitrate limitation, or preferential flow that biases results.

The meta-analysis by Addy et al. (2016) which includes results for 27 bioreactor beds from 14 peer-reviewed studies reported an overall mean mass removal performance of 4.7 g N m^{-3} d^{-1} (2.9 – 7.3; 5-95% confidence interval) with a predicted mass removal rate at 20°C (k_{20}) of ~9 g N m^{-3} d^{-1}. However, their study included results from some systems that were immature and some treating drainage waters with much higher nitrate-N concentrations.

Christianson et al. (2021) compiled data from a further 23 bioreactor bed studies, including some with biochar additions and/or receiving nitrified wastewaters and reported a median nitrate load removal rate of 5.1 g N m^{-3} d^{-1}. A recent review of the performance of 10 full size bioreactors in Illinois (36 site-years) all designed according to the USDA-NRCS CPS 605 (United States Department of Agriculture 2020) found a median removal of 4.6 g N m^{-3} d^{-1} at an annual flow weighted temperature of 10.6 ± 1.9 °C, equating to 51% of the influent nitrate-N load (Christianson et al. 2026). This appears to be well above what we have measured for New Zealand systems. This may be due in part to the relatively high levels of bypass noted for many of these systems, leading to them receiving a less variable range of flows than New Zealand systems.

The New Zealand studies summarised in Table 3 were all monitored after an initial flushing period when higher organic loads are released by the woodchip, and they often received comparatively low concentrations or loads of nitrate-N for much of the year. Addy et al. (2016) noted significantly reduced mass removal performance for bioreactor beds in operation for more than 13 months (e.g., 13-14 months, mean 2.8 g N m^{-3} d^{-1}, 5-95% confidence interval 1.4-5.4) compared to bioreactors operating for less than 13 months (e.g., mean 9.1 g N m^{-3} d^{-1}, 5-95% confidence interval 4.7-16). Also, bioreactors receiving <10 mg L^{-1} of influent nitrate-N had lower mass removal performance (mean 2.4 g N m^{-3} d^{-1}, 5-95% confidence interval 1.0-4.9) compared to situations with higher influent nitrate-N concentrations 10-30 mg L^{-1} (mean 4.9 g N m^{-3} d^{-1}, 5-95% confidence interval 2.9-7.9).

Table 3: Overview of New Zealand case studies of woodchip bioreactors treating nitrate from artificial drainage. Bioreactors were monitored after an initial flushing period and for 2–3 years with varying sampling intensity ranging from approximately monthly sampling and spot measurements of flows (Goeller et al. 2019), weekly sampling and continuous water level monitoring (Singh et al. 2025), and continuous water level and nitrate concentration monitoring (Hudson et al. 2019; Rivas et al. 2023). Case studies are described in Appendix C.

	Bioreactor case studies						
		New Zealand mean (range)	Tatuanui / Waikato (Rivas et al. 2020 & 2023)	Coastal Rangitīkei / Manawatū–Whanganui (Singh et al. 2025)*	Dannevirke / Manawatū–Whanganui (Singh et al. 2025)	Lowcliffe / Canterbury (Goeller et al. 2019)**	Waituna / Southland (Hudson et al. 2019)
Catchment details	Years monitored (years)		2 (2018–2020)	3 (2021 – 2024)	2 (2022 – 2024)	2 (2015 – 2017)	3 (2016 – 2019)
	Mean annual days of flow (n)	298 (102 – 365)	102	351	309	365	365
	Catchment size (ha)	22 (0.65 – 65)	0.65	13	65	>20	9.6
	Subsurface drainage nitrate–N yield (kg ha^{-1} yr^{-1})	10.4 (2.8 – 24.6)	15.5				5.2
Bioreactor sizing	Woodchip volume (m^3)	95 (28 – 180)	78	90	180	28	100
	Average saturated woodchip volume (m^3)	71 (15 – 165)	56	51	165	15	70
	Saturated volume: total woodchip volume (%)	69 (54 – 92)	72	57	92	54	70
	Saturated volume: catchment area (m^3 ha^{-1})	25 (3 – 87)	87	4	3	<2	7
Nitrate load reduction	Mean percentage of annual drainage treated (%)	72 (29 – 100)	99	33	29	100	100
	Mean nitrate–N load removed (kg N ha^{-1} yr^{-1})	3.6 (1.8 – 7.7)	7.7	1.8	2.2		2.6
	Mean relative nitrate–N load removal (%)	55 (10 – 78)	61	74	78	10	51
	Mean edge–of–field nitrate–N load removal (%)	33 (10 – 61)	61	22	22	10	51

	Bioreactor case studies						
		New Zealand mean (range)	Tatuanui / Waikato (Rivas et al. 2020 & 2023)	Coastal Rangitīkei / Manawatū-Whanganui (Singh et al. 2025)*	Dannevirke / Manawatū-Whanganui (Singh et al. 2025)	Lowcliffe / Canterbury (Goeller et al. 2019)**	Waituna / Southland (Hudson et al. 2019)
	Mean removal rate nitrate-N per saturated volume (g N m^{-3} d^{-1})	1.5 (0.9 – 2.9)	0.9	1.1	2.9		1.0
	Mean removal rate nitrate-N per total volume (g N m^{-3} d^{-1})	1 (0.2 – 2.3)	0.2	0.7	2.3		0.7

*The Coastal Rangitīkei bioreactor received flows from a controlled drainage scheme that intercepted mainly shallow groundwater conveyed in open ditches.

**The Lowcliffe bioreactor catchment area was indeterminable due to having a much larger groundwater-fed catchment

5.4 Treatment of other contaminants (sediment, phosphorus, and microorganisms)

There is a scarcity of data on field-scale bioreactors treating subsurface drainage in New Zealand to demonstrate the removal efficacy for other contaminants. However, some insights can be gleaned from bioreactors treating other drainage types (e.g., groundwater and surface drainage in Canterbury: Burbery et al. 2023a; Burbery et al. 2023b) and field-scale trials conducted overseas:

- Sediment: Subsurface drainage does not normally carry high sediment or suspended solid loads, so there are little field-scale data on sediment removal in woodchip bioreactors treating subsurface drainage (Christianson et al. 2021). Woodchip bioreactors can filter sediment, but the accumulated sediment will eventually hinder N removal performance by clogging and siltation. This restricts flows through the media and encourages development of preferential flow paths, thereby shortening the design life for denitrification (Christianson et al. 2020; Christianson et al. 2021). The extent and timeframe of this process would vary depending on factors such as the sediment characteristics, loading rates, and bioreactor media (notably woodchip particle size). Therefore, where high sediment is present, it is recommended to pair bioreactors downstream of detention basins or constructed wetlands to pre-settle out suspended particulates (Choudhury et al. 2016; Tanner et al. 2012).
- Phosphorus: Mechanisms of bioreactor P removal include physical filtration. Chemical adsorption, and microbial immobilization. However, the leaching of dissolved reactive phosphorus (DRP) is a potential performance trade-off. In New Zealand, bioreactors may release some DRP during the first drainage season but typically become a net sink or show negligible release thereafter at bioreactors in the Waikato, Upper Manawatū, and Coastal Rangitikei (Perera et al. 2024a; Singh et al. 2025). For example, an in-ditch woodchip bioreactor in South Canterbury was found to be a net sink for DRP overall (Burbery et al. 2025b). However, short-term DRP release can occur under specific redox conditions—particularly during low flows, high temperatures, and iron- or sulfate-reducing environments—highlighting that P dynamics are strongly influenced by site conditions and hydrology.
    - To determine whether full-size woodchip bioreactors were a net source or sink of DRP, Sanchez-Bustamante et al. (2025) evaluated 23 site-years of data from 10 full-scale Illinois bioreactors (1,904 samples). Overall, DRP decreased across bioreactors in 15 of the 23 site-years. Median annual performance showed a 12% DRP reduction. However, all metrics ranged across zero, indicating there were times when the bioreactors released phosphorus, particularly when the bioreactor was new and under very low flow conditions in the summer. The highest daily removal occurred when inflow DRP concentrations were high and

hydraulic retention times were low, indicating performance improved under higher loading conditions.

- Microorganisms and other agricultural pollutants: Most studies on bioreactor removal of other agricultural pollutants have been done either at the lab-scale or on water that was not subsurface drainage (Christianson et al. 2021). Removal of microorganisms, including bacteria and viruses, has been documented with field-scale bioreactors in New Zealand. Bioreactors have some, albeit limited, capacity to filter microbial contaminants like *Escherichia coli* and protozoa (Tanner et al. 2012; Rambags et al. 2016; Burbery et al. 2025b). Overseas studies of field-scale bioreactors treating other effluent sources have showed removal of herbicides and veterinary antibiotic compounds, but the fate of these is unknown (Gottschall et al. 2016; Hassanpour et al. 2019; Ranaivoson et al. 2019).

5.5 Pollution swapping potential

Field-scale case studies in New Zealand and overseas show that woodchip bioreactors treating artificial drainage are generally a low-risk mitigation when well designed and managed. However, there is an inherent need to balance nitrate removal performance with the risk of "pollution swapping." This reflects the fact that interceptive mitigation tools operate under strong redox gradients, which can drive other biogeochemical processes, particularly when nitrate becomes limiting (Christianson et al. 2013; Goeller et al. 2016; Weigelhofer and Hein, 2015). While these effects are not typically prohibitive, they highlight the importance of good design, site selection, and management, supported by an understanding of catchment priorities and receiving environment sensitivity.

Potential pollution swapping effects that should be considered in design and operation include:

- Temporary leaching of excess bioavailable carbon, ammonia, and phosphorus during bioreactor start-up (Burbery et al. 2022; Healy et al. 2012; Sharrer et al.2016; Christianson et al. 2017; Hartfiel et al. 2022).
- Gaseous emissions of carbon dioxide (CO_2) and nitrous oxide (N_2O), associated with high dissolved organic carbon (DOC) and variable redox conditions; N_2O production is more likely when nitrate reduction is incomplete (e.g., low pH or elevated dissolved oxygen) (Moorman et al. 2010; Warneke et al. 2011).
- Methane (CH_4) production under strongly reducing conditions.
- Release of tannins, chemical oxygen demand, methyl mercury, hydrogen sulphide, and bioaccumulated metals are also potential risks (Fenton et al. 2016; Wickramarathne et al. 2020; Lepine et al. 2021). In the New Zealand context, mercury risk is considered low due to low background levels and

limited evidence of mobilisation in agricultural drainage, although local geothermal settings should be considered where relevant.

However, if designed and operated appropriately, these issues can be managed. Pollution swapping can be effectively minimised through:

- Using aged (not fresh) woodchip to reduce leaching of organic compounds.
- Managing start-up flows, including diverting or recirculating first flush water onto land.
- Avoiding commissioning or rewetting during low-flow, high-temperature periods when receiving environments are most sensitive to organic pollution.
- Maintaining steady hydraulic conditions (e.g., controlled drainage or upstream retention) to limit extreme redox shifts.
- Managing bioreactors to optimise nitrate removal (supporting complete denitrification) and reduce greenhouse gas emissions. Methane emissions and under certain conditions nitrous oxide emissions can be further reduced by installing a soil cap over the woodchip (Brunton et al. 2024; Carstensen et al. 2019; Manca et al. 2021).
- Considering receiving environment sensitivity and catchment priorities in siting and design.

Evidence of ecological impacts from bioreactor effluent remains limited compared to other interceptive diffuse pollution mitigation tools (Goeller and Woodward 2021). Laboratory ecotoxicology studies in New Zealand indicate little to no acute or chronic effects on algae, zooplankton, or īnanga under tested conditions (Thompson and Goeller 2023), but these results are based on controlled eco-toxicity scenarios and a limited range of operating conditions. Field-scale assessments of ecological effects are scarce due to cost and complexity, and further work—particularly under "worst-case" operating conditions—would improve confidence in risk assessments. Finally, accounting for the pollution swapping potential of nutrient attenuation tools like bioreactors as compared to agricultural sources is important to evaluate their overall environmental impacts. For example, GHG emissions from woodchip bioreactors are likely to be no greater than emissions from other agricultural sources or nitrate-polluted waterways (Elgood et al. 2010; Groh et al. 2015; Goeller et al. 2019; Ratcliffe 2023).

6 Woodchip bioreactor design

6.1 Rules and regulations

Keep it legal: If you want to construct a woodchip bioreactor and it involves excavation near to watercourses or wetlands, disturbance to waterways, or water diversion, you may need resource consent. Requirements vary between regions, as councils regulate activities such as earthworks, stream disturbance, and discharges under regional plans developed through the Resource Management Act. In many cases, these activities can be permitted, but only if specific conditions are met; otherwise, consent will be required (Milne 2022).

Regulatory requirements can influence how and where diffuse pollution mitigations (including bioreactors) are installed, and may affect project timelines, costs, and design choices. Variability in council rules and consent conditions is a recognised barrier to wider uptake of these systems, particularly where works occur in or near waterways (Milne and Luttrell 2020).

Always contact your local regional council early for advice on applicable rules and consent requirements before any earthworks commence. Councils can also help identify funding opportunities and ensure your design is compliant, which can avoid delays and improve the likelihood of successful implementation.

6.2 Design fundamentals

A typical bioreactor is constructed as an excavated depression or bed filled with woodchips (Figure 6-1). A bypass feature is included to prevent any drainage problems in the field. Subsurface drainage water is routed into the bioreactor through an inlet manifold. The inlet manifold consists of a distribution pipe or pipes that disperse water laterally along the cross-sectional area of the woodchip bed. Flow through the system needs to be well dispersed to prevent short-circuiting or dead zones. As water passes through the woodchip bed, microbial activity converts nitrate in the drainage water into harmless nitrogen gas. The treated water is then collected by an outlet manifold and discharged back to the main subsurface drain or a designated outlet.

The operating depth and therefore the active volume of saturated woodchip is regulated by an outlet control structure. Keeping the base of the bioreactor parallel to the bottom of the drainage line and with a gentle slope is preferable to allow for gravity-led drainage of the bed outside of the active flow season (Figure 6-1). Consider providing a small notch or orifice in the bottom of the outlet control structure to allow the bioreactor to fully drain after inflows have stopped; this may be relatively more important at sites with lower flow rates or flashier flow trends. This avoids excessively long HRT and stagnant water effects that influence pollution swapping and can result in production of sulphurous odours (Maintaining the Edge 2025).

Bioreactors may be open, where woodchips extend to the ground surface, or covered with a shallow soil cap (which helps reduce GHG emissions).

They commonly include a synthetic liner to prevent water from entering or leaving the bioreactor through the surrounding soil, especially where bioreactors are built for research purposes. Practically, however, a fully water–light liner will not likely be necessary where soils are clayey enough to retain water. Where this is not the case, a low–cost liner can be used to keep the wood chip and surrounding soil separate.

Woodchip bioreactor beds tend to subside unevenly as they age (Christiansen et al. 2025a) creating a potential hazard to livestock, people and farm machinery trying to traverse them. It is therefore recommended to fence bioreactors to exclude livestock and in cropped areas to clearly mark flow control structures and corners of the bioreactor to deter vehicle access. Keeping livestock and vehicles off bioreactors also avoids compaction of the woodchips and possible damage to liners and pipework (e.g., from hooves or tyre tracks).

However, there may be situations where fencing is not practical or is undesirable due to loss of productive land. In such cases, unfenced bioreactors may still be used, provided land managers are aware of the trade–offs. Grazing or trafficking over a bioreactor can lead to compaction, reduced hydraulic performance, and a shorter operational lifespan, and there is an increased risk of surface irregularities affecting stock safety over time. On the other hand, allowing continued land use over the bioreactor footprint may improve farm integration and uptake where the alternative is that no mitigation is installed.

Where bioreactors are not fenced, managers should consider lower stocking pressure, avoiding heavy machinery, and periodic inspection to identify subsidence or damage early. Overall, this reflects a practical trade–off between maintaining full design life and performance versus enabling implementation with minimal impact on productive land—an area where further field evidence in New Zealand would help better quantify long–term outcomes.

Bioreactor designs should aim for a targeted load reduction and maintaining conditions suitable for denitrification. Generally, effective denitrification in bioreactors is most likely to happen under the following conditions:

- Nitrate–N concentration of the inflow: nitrate–N concentrations should be on average > 2 mg L^{-1} (to avoid nitrate–limiting conditions) and ideally >5 mg L^{-1} during the main flow season. In comparison, Danish bioreactor guidelines recommend that inflowing winter nitrate–N concentrations should be on average 4 mg L^{-1}, based on three to four samples (Plauborg et al. 2021). Queensland guidelines for woodchip bioreactor beds recommend that inflowing nitrate–N concentrations should be at least 3 mg L^{-1} for most of the time (Wegscheidl et al. 2021).
- Use carbon sources that are bioavailable. Woodchip bioreactors should use coarse, untreated woodchips (e.g., nominal chip size 20–50 mm) that maintain their hydraulic properties better than finer woodchips (Johnson et al. 2022).

- Good hydraulic efficiency: water can move freely into and evenly through the bioreactor along its length without preferential flow or dead zones.
- Consistent inflows: flows are consistent during the drainage season and the operating depth within the bioreactor can be maintained to saturate at least 50% (ideally 60–80%) of the total available woodchip volume.
- Hydraulic retention time (h): The HRT promotes full denitrification whilst not over–treating N. HRT should be at least 4 h to avoid undertreatment and not longer than 24 h to avoid overtreatment. These general recommendations also depend on temperature and water chemistry.
- Dissolved oxygen levels in the bioreactor are low (<2 mg L^{-1}) to promote denitrification.
- Suitable pH (between 5.5–8.0; Wegscheidl et al. 2021). New Zealand groundwater generally has a circumneutral pH.
- Warmer temperatures. Performance is likely to be reduced in colder regions and seasons. In colder regions, bioreactor size needs to be increased to compensate for lower temperatures, since nitrate removal is temperature dependent (Figure 3–1).

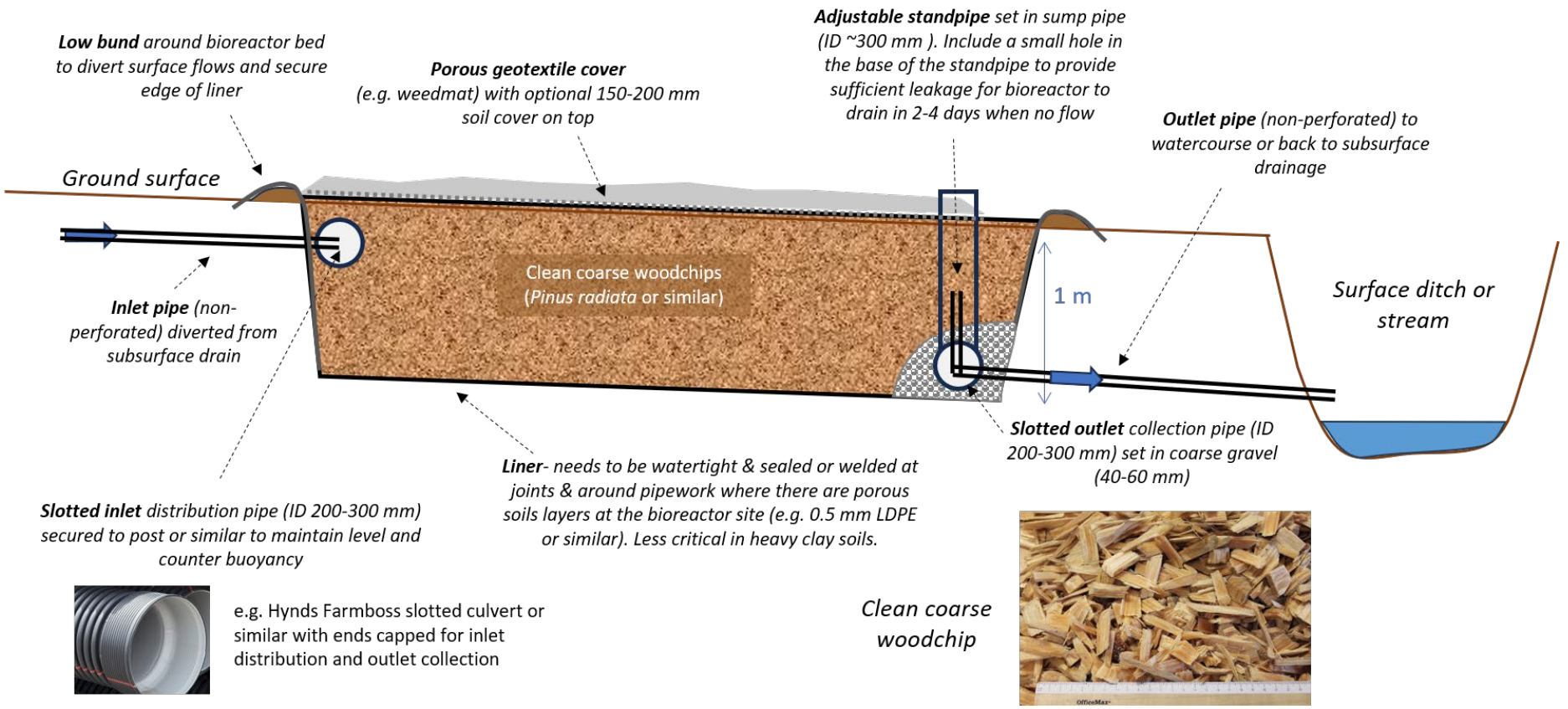


Figure 6-1: Overview of the design features of a woodchip bioreactor intercepting subsurface drainage. The inlet pipe is installed at a higher elevation than the outlet pipe to provide sufficient hydraulic head for flow through the bioreactor. The water level within the bioreactor is controlled by an adjustable standpipe at the outlet; however, adequate elevation difference between the inlet and outlet is still required to create the hydraulic gradient needed to drive water through the woodchip media. The outlet pipe is positioned at a lower elevation and aligned with the existing tile drainage system to allow treated water to discharge back into the main drain or receiving waterway. A slight longitudinal slope is incorporated into the bioreactor bed to promote complete gravity drainage when flow ceases, reducing the potential for stagnant water within the system.

6.3 Location and configuration

Bioreactors are best positioned near the outlets of subsurface drains or can intercept controlled drainage from groundwater–fed surface drains at the edges of fields or paddocks, near where they discharge to a surface ditch or watercourse. To mitigate interfering with the up–stream drainage function, they are best installed off–line, adjacent to the drainage line. The goal is to treat nitrate in drainage water without compromising the primary function of the drainage system—efficiently removing excess water from the land. Figure 6–2 shows a range of possible configurations for off–line bioreactors. A portion of the subsurface drainage is diverted via a weir system or short standpipe into the bioreactor. Appendix D shows examples of bioreactor water level control structures used in the United States of America (Maintaining the Edge, 2025). Excess flows bypass the bioreactor by overflowing the weir and continuing down the original drainage line. In–line configurations are also possible, with excess flows diverted around the bioreactor.

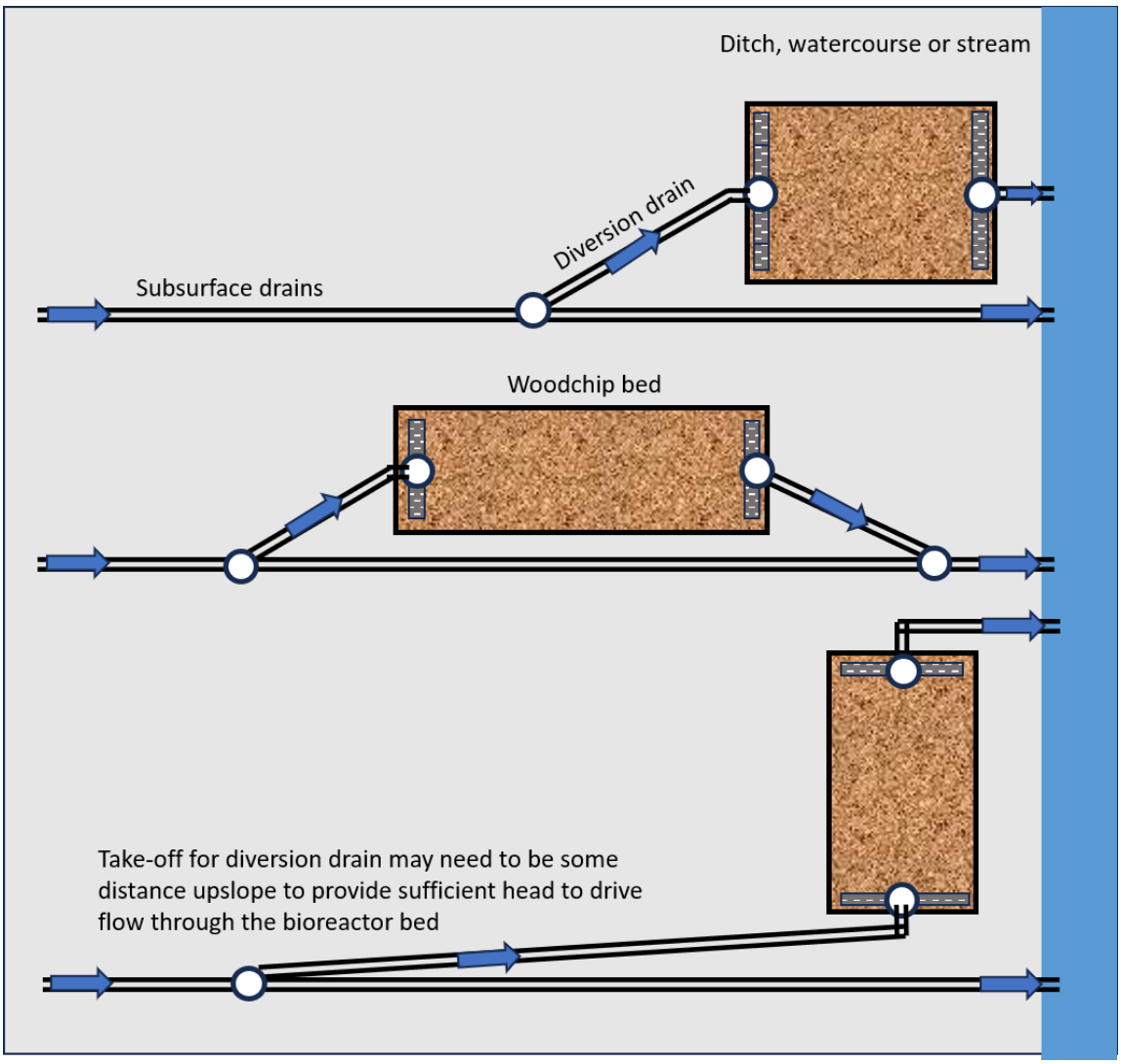


Figure 6–2: Plan view of three typical configurations for off–line bioreactors discharging to a water course. During high flows, the bioreactor will receive the first flush of drainage. Flows exceeding the hydraulic capacity of the bioreactor discharge will pass through the subsurface drain directly to the watercourse.

There are several factors to consider when choosing a site for a bioreactor, especially not interfering with the drainage functions of the existing drainage infrastructure. Careful consideration of these factors can help ensure that the bioreactor is able to effectively reduce nitrate levels in agricultural drainage water without adversely affecting the function of the upslope drains or interfering with farming activities.

Considerations for suitable sites:

- A suitable site needs to be able to intercept artificial drainage flows from the upslope catchment and have sufficient slope to maintain proper functioning of the drains. Allowance also needs to be made for the additional head loss caused by passage of water through the bioreactor media (range ~ 0.2–0.8 m depending on configuration) and diversion of drain flow preferentially to the bioreactor generally requires use of a low weir or standpipe (~0.1–0.2 m). Where an existing drain is diverted through a bioreactor, sufficient fall must remain between the upstream drainage system and final outlet to accommodate these head losses without impairing drainage performance. This can be achieved by locating the diversion point further upslope to utilise the natural fall available in the drain network (Figure 6–2), or by installing a diversion drain at a shallower grade than the existing drain while maintaining the original drainage outlet elevation (Figure 6–3).
- Sites with a gentle gradient between the inlet and the final discharge point will help ensure that subsurface drainage flows through the bioreactor without adversely impeding the drainage above or outflow from the bioreactor (Figure 6–3). Ideally, final discharge outlets to watercourses should be above high flow water levels so discharge from the bioreactor outlet drain is not restricted. This also helps to aerate the outflow.
- Surface drainage ditches that are mainly groundwater–fed with clear water and low fine sediment loads can be intercepted by off–line bioreactors (Figure 6–4). These sites incorporate controlled drainage by diverting a portion of the drain flow into a side channel with a settling pond upstream of the inlet. The settling pond helps to provide more consistent inflows and allows for any sediment to settle so that it does not clog the bioreactor. The efficacy of sedimentation ponds will depend on the sediment concentration, particle settling characteristics, and the flow rate.
- Very flat sites with minimal fall in subsurface drains are generally not suitable for achieving gravity inflow into and through a bioreactor. Example diagrams showing possible bioreactor designs for flat sites or sites with pumping are provided as advanced design options in Appendix E.

Site features and situations to avoid:

- Drainage systems transporting high sediment loads are generally unsuitable, as sediment can clog the woodchips, restrict flow through the bioreactor, and reduce both performance and operational lifespan.

- Avoid situations that may impede drainage. Temporarily backing-up drainage upgradient of the bioreactor inlet can potentially cause sedimentation in the drain-line. If drainage above the bioreactor is seriously impeded, high water tables over extended periods may also cause collapse of mole-drains where present. On gently undulating, sloped land (e.g., > 1° slope), the distance back-up will occur is generally relatively small and there is negligible risk of flooding mole drains because they are located well above the pipe drains (Figure 4-3).
- It is also important to avoid areas that are prone to flooding by surface run-off or storm-flows, and to avoid disturbing any natural wetlands or wetland vegetation.
- Bioreactors should not be sited in an area where it will interfere with other farm activities or infrastructure.

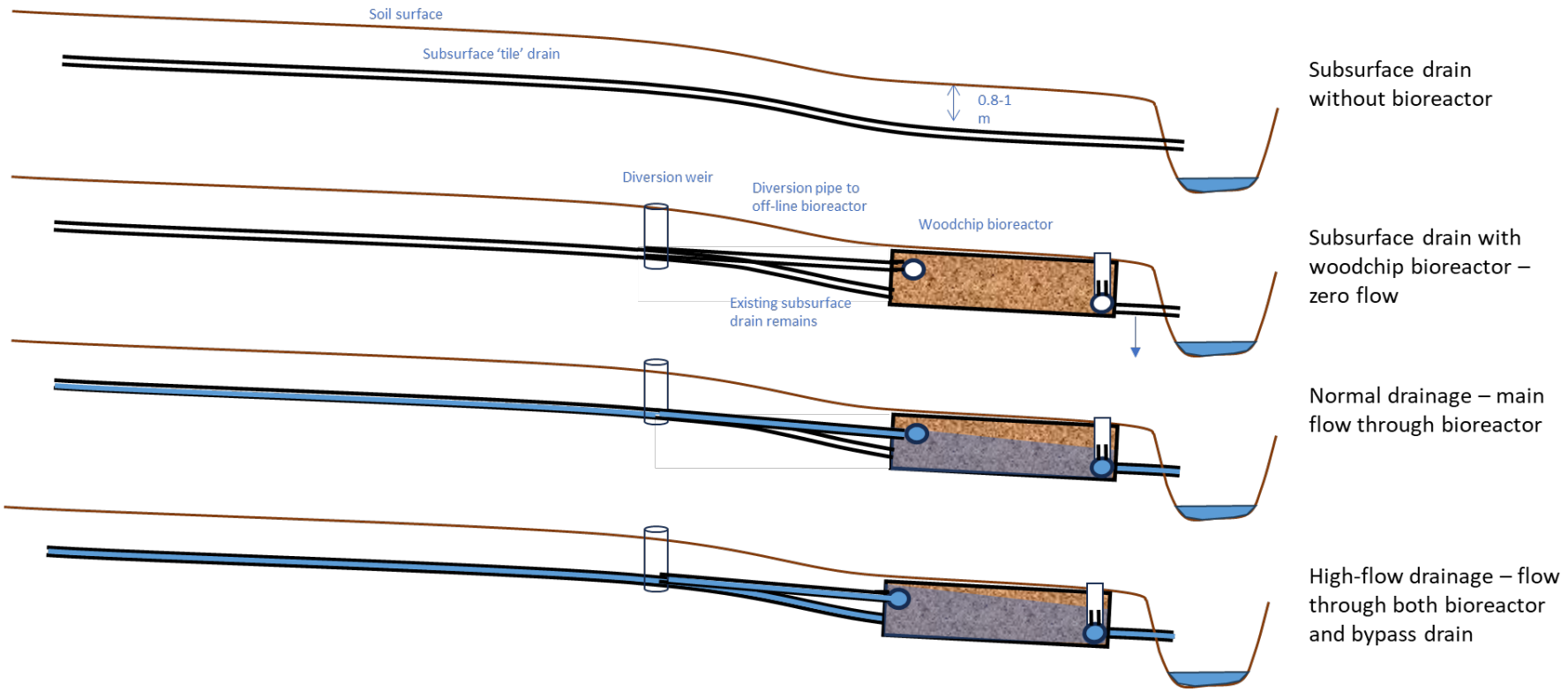


Figure 6-3: Typical longitudinal cross-section of a subsurface drain before and after installation of a woodchip bioreactor on a gently sloping site, showing no-flow, normal-flow, and high-flow condition. Bypass flows are conveyed through the existing subsurface drainage system rather than the bioreactor bed. The inlet pipe is set higher than the outlet pipe to provide the hydraulic gradient required for flow through the woodchip media. Water levels within the bioreactor are controlled by an adjustable outlet standpipe, while the lower outlet elevation allows discharge back to the existing drainage system or receiving waterway. A slight bed slope promotes complete gravity drainage when flow ceases.

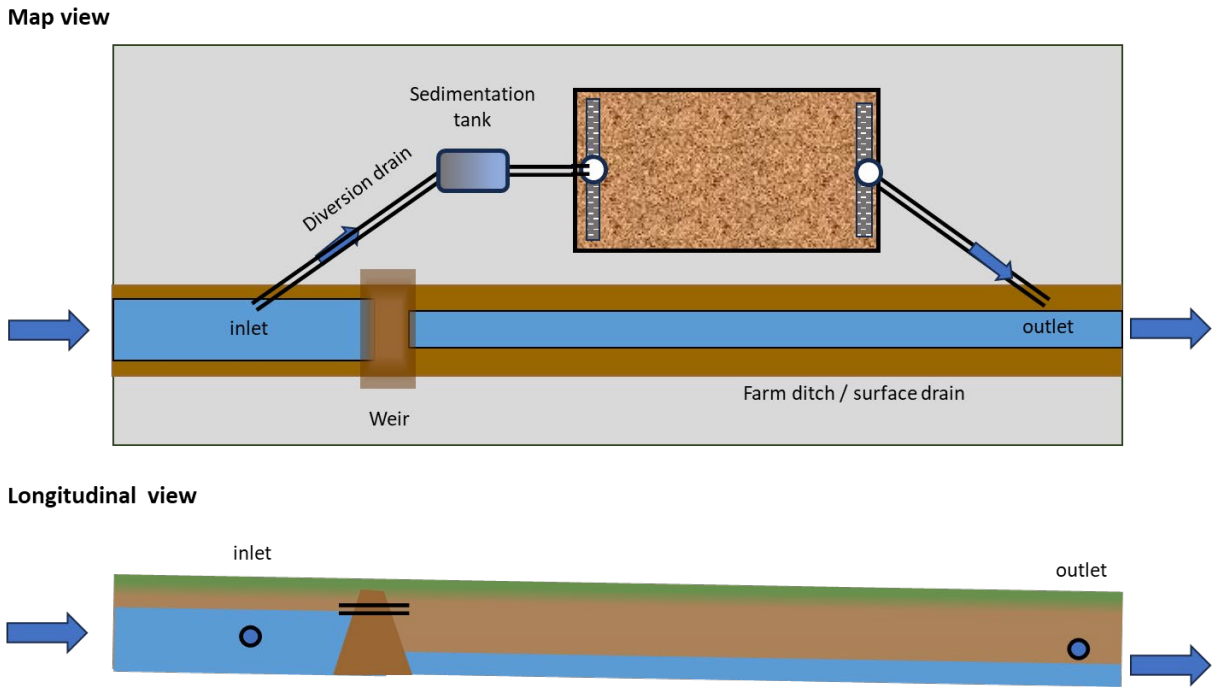


Figure 6-4: Off-line bioreactor treating low-sediment flow from a groundwater-fed surface drain. A weir provides head to divert water into the bioreactor, with a high-level overflow to bypass excess flow. These systems are most suitable for water with relatively low sediment loads; upstream settling is recommended to minimise clogging. Incorporating controlled drainage stabilises inflows, increases hydraulic retention time, and improves treatment performance. Ensure access for routine maintenance, including cleaning distribution pipes and replacing inlet media.

As discussed in section 4.3, the location of subsurface drains on New Zealand farms is generally poorly documented. Manderson (2018) provides estimates of likely tile drained areas across New Zealand (see Appendix B), and Pearson (2015) provides specific estimates for Southland. Our assumption for these recommendations is that tile drains will generally only be implemented by farmers where they are likely to provide practical benefit, and that farmers will either know where drains have been installed on their farms or be able to identify the drain outlets visually or by the sound of water running when they are flowing. The catchment areas of the drainage systems should then be able to be determined using aerial imagery, light detection and ranging (LiDAR) elevation data, land contours, and knowledge where the drainage network extends to.

Considering catchment size, good potential bioreactor sites have increased flow volumes and increased N loading (relative to catchment area). Generally, larger catchment areas have greater loading. Danish guidelines recommend a minimum drained catchment area of 15 ha (Plauborg et al. 2021), whereas most Illinois bioreactors are on catchments < 19 ha (Christianson et al. 2026). In New Zealand, trials so far have been on catchments ranging from <1 ha (Rivas et al. 2019) to > 9 ha (Hudson et al. 2019; Singh et al. 2025) (Table 3). Several New Zealand bioreactors have intercepted drainage from much larger groundwater-fed catchments with indeterminate catchment areas (Goeller et al. 2019; Singh et al. 2025) (Table 3).

6.4 Bioreactor sizing and performance

The key is to size individual bioreactors with an adequate volume of woodchips to optimise effectiveness at the site-scale, while maximising load reduction to help meet environmental targets at broader scales. If reductions in nitrate-N concentration and load are required, for example to meet a specific downstream water quality target to protect a drinking water source or sensitive fish species, then larger woodchip bioreactors (or smaller source catchments) will be required. These will, however, achieve a lower load reduction per unit of volume.

Internationally, the prevailing bioreactor sizing paradigm is to estimate the volume of woodchips required based on achieving a percentage reduction of the load treated by the bioreactor and ensuring a target HRT is achieved at a designated flow rate (often expressed as a percentage of peak flow). However, from a catchment management perspective, it is less worthwhile to treat a smaller volume of water very well (i.e., a high percentage load reduction but a low mass load reduction), as compared to treating a larger volume of water less well but removing a higher total load (Christianson et al. 2025b; Christianson et al. 2026).

The USDA-NRCS CPS 605 (United States Department of Agriculture 2020), Danish bioreactor guidelines (Plauborg et al. 2021) and Queensland bioreactor guidelines (Wegscheidl et al. 2021) follow a sizing approach that aims for a minimum HRT that is related to a targeted percentage load reduction. This is influenced by precedent of the USDA-NRCS CPS 605, as well as the reality that bioreactors are implemented to help achieve percentage load reductions in the receiving environment (e.g., Gulf of Mexico, Baltic Sea, or Great Barrier Reef catchment).

Moving from an HRT-centric design approach is an emerging topic of discussion in the science community (Christianson et al. 2026). HRT-centric bioreactor design approaches are in some ways less ideal because they do not guarantee an effective design with regards to maximising the active volume of the bioreactor or hydraulic efficiency. Also, HRT alone does not correlate well with other important nitrate load reduction performance metrics. Therefore, we support the recommendations of Christianson et al. (2026) that HRT should be viewed in tandem with other factors during the design process.

For New Zealand bioreactor applications, calculating the volume of woodchip required based on a predicted annual average mass load removed enables farmers to account for their nutrient loss reductions within their farm environmental plans. Farmers and rural advisors are well familiar with nutrient inputs and losses expressed as area-weighted load (yield) standardized per area (e.g., kg nitrate-N ha^{-1} y^{-1}). Therefore, these recommendations support sizing a bioreactor based on a reduction in the nutrient load itself (rather than a desired percentage load reduction).

Table 4 provides a summary of different approaches to characterise drainage flows and nitrate-N loads as a basis for sizing of bioreactors. Cross-validation of the most promising options in Table 4 would help to shed light on whether the different approaches result in similar estimates of bioreactor woodchip volume.

In the meantime, the following approaches are likely to be most useful for designing bioreactors based on the inflow load:

- The optimum approach to determine sizing based on the inflow load is to regularly measure the flow and nitrate concentration in the drainage discharge across the characteristic range of drainage conditions they are likely to experience at the site, ideally over several years (Option 1).
- Where more intensive monitoring is out of scope, spot measurements of discharge flow rates and nitrate concentrations, observations of the period that drainage flows occur, and how full the tile drain is flowing can provide useful information (Option 2).
- In lieu of either monitoring–based approaches, using modelled estimates of peak drainage flows (Option 4) is recommended.
- Alternatively, Option 7 provides a solution that leverages combining modelled winter flows and empirical drainage data.

Table 4: Different approaches to characterise drainage flows and nitrate–N loads as a basis for sizing of bioreactors. Summary and comparison.

Approach	Description	Comment
1. High frequency monitoring of drainage flow and concentration.	A scientifically robust approach would be to monitor over several years to get an indication of year–to–year variability. This could be done with in–situ sensors to continuously measure drainage flows and nitrate–N concentrations, e.g., hourly measurements, or by manual or automated sampling of flows and water quality over the drainage season and though drain flow events.	Can provide information on drainage flow and nutrient load variability and enable tailoring bioreactor size to optimise performance. However, it is technically challenging, time consuming, and expensive so it is usually reserved for research purposes or high–profile case studies. There is still then the question of how to use these high frequency measured drainage flows and loads to size bioreactors. This requires technical expertise, which could be time consuming and expensive for practical applications.
2. Targeted monitoring of drainage flow and nitrate–N concentration.	A practical approach would be to approximate peak and typical drainage flows and their duration by measuring after at least two heavy rainfall events during the main drainage season. Average nitrate–N can be approximated from at least three grab samples taken over the main drainage period. Massey University provide instructional videos on measuring drainage flows in this way: https://massey–flowmeters.shinyapps.io/Bioreactor_designer/	Likely to be only practical monitoring option for farm–scale implementation on existing drainage systems and does provide real–world design information. Needs to be done at the right time of year and soon after large rainfall events. Relies heavily on the judgement of the person doing the sampling and whether the conditions being monitored are representative of longer–term behaviour. Moderate costs for sampling and nitrate analysis. There is still then the question of how to use these measured flows and loads to size bioreactors.
3. Theoretical hydraulic capacity of the drainage system.	Standard drainage coefficient values (mm d^{-1}) applied in the design of drainage systems for different land–uses are summarised in Table 1. The drainage co–efficient describes the maximum expected flow through drainage pipes and is calculated using the Manning's equation (or on–line calculators and nomographs that employ it, e.g., Appendix A) using information on the roughness of the pipe (resistance), slope	In practice, standard drainage coefficients are generally presumed based on the land–use. Given the general lack of knowledge of the location and design details for many drainage systems in New Zealand, and the ad hoc way that they have often been implemented, it is likely this approach will only provide

Approach	Description	Comment
	(driving force), and cross-sectional area of the pipe (capacity). Drainage.org provides an on-line tool for measuring the Drainage Coefficient. https://transformingdrainage.org/tools/drainage-rate/ The peak drainage flow rate ($m^3 d^{-1}$) is then calculated by multiplying the peak Drainage Coefficient ($m d^{-1}$) by the drained area (m^2). For sizing bioreactors in the United States of America, the most used approach (Christianson et al. 2021) is to design for an HRT of 3 h at a design flow rate that is at least 15% of the peak estimated drainage flow rate from the drainage system (USDA, 2020). This has been shown under US conditions to facilitate treatment of, on average, 61% of drain-flow (Christianson et al. 2026). However, the appropriateness of these criteria, as well as the Drainage.org tools, may not apply under New Zealand soil, climate and drainage system designs.	a very approximate estimate of peak drainage flow rates. Also, a suitable percentage of the peak estimated flow needs to be selected for local conditions. For New Zealand drainage with flashy hydrology, sizing based on a percentage of a theoretical peak flow may not be appropriate; e.g., this could result in bioreactors that are sized too large for the majority of the flow season, with potential over-treatment during low flows and high cost per kg of N removed.
4. Modelled estimates of peak drainage flows.	There are two other common ways of characterising subsurface drainage rates (Skaggs, 2017). 1) The Kirkham Coefficient is the steady subsurface drainage rate corresponding to a saturated soil profile with a shallow ponded surface. 2) Drainage Intensity is calculated by the Hooghoudt equation and is dependent on the effective saturated hydraulic conductivity of the soil profile, drain depth and spacing, effective radius of the drain, and equivalent depth to the restrictive layer. The question then still arises as to what proportion of this theoretical maximum flow and design residence time should be used for bioreactor sizing and how this differs from using the drainage coefficient (Option 3).	These options, which take account of the characteristics of the soil and the drainage system, are likely to provide more accurate approximations of the likely flows from different drainage systems. They do require more technical information, but much of this is available from on-line sources. However, the ad-hoc nature of many subsurface drainage systems in New Zealand means that as-installed drain depths and spacings may deviate from what is expected. Massey University's Catchment Solutions Programme provides a practical user guide to accessing and applying this information. E.g., this model uses the Manning formula combined with a daily soil water

Approach	Description	Comment
		balance calculation to estimate drainage flows (Avendaño et al. 2026). https://massey-flowmeters.shinyapps.io/Bioreactor_designer/ Transforming Drainage.org provides an on-line tool for measuring both the Kirkham Coefficient and the Drainage Intensity: https://transformingdrainage.org/tools/drainage-rate/
5. Modelled estimates of long-term drainage yields.	Catchment-scale hydrological and water quality models such as HYPE, SWAT, or numerical groundwater models such as MODFLOW can provide generalised estimates of daily subsurface drainage flows and nitrate-N loads back-cast over previous years (e.g., Srinivasin et al. 2020). These can be used to predict drainage flows and yields in tile drained areas around the country. In areas where subsurface drainage is installed, this information can be used to estimate drainage yields, assuming it constitutes a specific proportion of groundwater flow. If this approach is used to determine a median long-term seasonal flow, then it could be applied to determine an appropriate sizing to achieve a specified nominal HRT for the median flow or some proportion of it.	At present, except for some specific catchments, we do not have ready access to catchment-scale predictions specifically focussed on subsurface drainage flows and loads. However, the physically-based, national hydrological model TOPNET can be combined with empirically based modelling approaches to estimate groundwater yields across the country based on long-term data records over the last 26 years (See Appendix B; Clark et al 2008, Booker and Woods 2014, Cattoën et al. 2022). There is still then the question of how to use these estimates of median flow and load to size bioreactors.

Approach	Description	Comment
6. Empirical data on subsurface drainage flows on New Zealand farms.	Areas of New Zealand that support intensive pastoral grazing and have subsurface drainage installed have several commonalities in terms of farm management and climate. Data collated from subsurface drainage studies can provide generalised information on drainage characteristics, such as periods and rates of flow, and contaminant concentrations and loads as shown in Table 2. These data show that subsurface drainage occurs predominantly in the winter. Where monitoring has continued over multiple years, it is obvious that flows and nitrate–N loads vary markedly at the same site in different years but fit within a common range.	There is still the question of how to use these estimates of median flow and load to size bioreactors.
7. Combining modelled winter flows and empirical drainage data.	Hybrid of Options 5 and 6 using long–term median seasonal flow data for the dominant winter period of drainage with median values for nitrate–N concentration can potentially provide a useful basis for determining median seasonal loads which can be used to sizing bioreactors.	There is still the question of how to use these estimates of median flow and load to size bioreactors. An example of a design sheet using this approach is show in Figure 12. Such an approach would be improved if modelled subsurface drainage estimates were available rather than assuming they constitute a proportion of modelled groundwater flows.

Once the expected drainage flows and nitrate–N loads have been established, the volume of the woodchip bioreactor can be sized appropriately for the predicted nitrate load. If the estimated woodchip volume is too small for the predicted nitrate–N load, the bioreactor's removal performance will be compromised. If the estimated woodchip volume is oversized for the predicted nitrate–N load, bioreactors become less cost–effective and the risk of pollution swapping increases.

An example workflow of the woodchip volume sizing process based on a range of winter seasonal groundwater flows is shown in Table 5. This example is similar to Option 7 in Table 4 that uses modelled estimates of groundwater flows, information from published subsurface drainage studies, measured drainage nitrate–N concentrations, and optimal nitrate removal performance metrics based on New Zealand bioreactor case studies. The steps and caveats of this approach are summarised as follows:

1. Estimate median winter groundwater flux. Use a groundwater flux estimate for your location from the map in Appendix B to determine the median winter groundwater flux range (mm season^{-1}) and calculate the midpoint to represent typical conditions.
    - Note: Winter is the main drainage period across New Zealand, although the actual drainage season will be longer in most cases. The midpoint provides a representative design value. However, inter–annual variability in climate means actual recharge may deviate substantially from this estimate in any given year.
2. Convert the winter median groundwater flux to a median daily flux (mm d^{-1}) by dividing by the winter season duration (92 days).
3. Convert median daily groundwater flux to a volumetric daily groundwater flow (m^3 d^{-1}) considering the size of the catchment area.
    - Note: Uses median winter flow only, so does not account for peak flows, storm events, or short–term variability, meaning actual performance during high flow periods may be lower.
4. Assume a proportion of groundwater contributes to subsurface drainage (%).
    - Note: This example assumes a fixed proportion of groundwater (60%) becomes subsurface drainage. However, the fraction can vary significantly with soil type, drainage design, and antecedent moisture conditions.
5. Estimate median daily nitrate–N load (g d^{-1}) by multiplying drainage flow by the example average nitrate concentration (g m^{-3}).
    - Note: Applies a constant nitrate concentration (in this example 5 g m^{-3}), whereas concentrations typically vary over time (e.g., hour–by–hour during storm events or weekly to monthly over the drainage season).

6. Apply the target nitrate removal efficacy (%) to the nitrate load to be removed.
    - Note: This example uses a 50% reduction. This is relative to the load that the bioreactor receives, not taking into account any untreated bypass flows. If the bioreactor bypasses a substantial portion of the annual tile drain N load, then the overall N load percentage reduction will be lower than 50%.
7. Calculate the required woodchip bioreactor volume (m^3) using the assumed saturated volumetric nitrate–N removal rate (g N m^{-3} saturated woodchip d^{-1}) to achieve the targeted load removal.
    - Note: The information in Figure 3–1 can be used to provide a saturated, volumetric nitrate removal rate that is temperature dependent and reflective of New Zealand drainage with high flow variability. This example uses a value of 2 g N m^{-3} saturated woodchip d^{-1}, which is the estimated mass removal rate at 15°C for New Zealand pastoral drainage from Figure 3–1.
8. Express the result as bioreactor volume per hectare (m^3 ha^{-1}) based on the size of the treated catchment (ha).
    - Note: 1 ha is used in this example for simplicity. For catchments >1ha, remember to multiply by the size of the catchment to ensure that the volume of woodchip is scaled correctly. Expressing results as bioreactor volume per hectare provides a useful benchmark for comparing designs between sites. However, it should not be interpreted as a rule of thumb for sizing bioreactors based solely on catchment area, because drainage volumes and nitrate loads vary substantially with climate, soils, drainage infrastructure, and land use.
9. Convert volume required to footprint area expressed as bioreactor size relative to catchment area (%) assuming a saturated media depth of ~1 m, giving plan area relative to 10,000 m^2 ha^{-1}.
    - Note: This percentage is useful for referencing the relative area required for the bioreactor, and it is not intended for use as a sizing rule of thumb. The actual land requirements may be greater due to side slopes, inlet/outlet structures, sediment forebays, and fencing.
10. Check hydraulic retention time. Estimate HRT (h) using effective porosity (void volume, unitless) and median volumetric daily flow (m^3 h^{-1}):

$$\text{HRT} = \frac{V_{void}}{Q_d}$$

Where:

V_{void} is effective porosity, 0.5 in this example

Q_d is the median daily groundwater flow (m^3 h^{-1}) from step 3 converted from days to hours

- Note about porosity: In this example, the porosity value of 0.5 is realistic for coarse woodchips, but woodchip porosity changes over time due to compaction, clogging, and biofilm growth.
- Note about HRT: The system achieves the design target of ~15 hours HRT. HRT should be at least 4 h to avoid undertreatment and not great than 24 h to avoid overtreatment. Actual HRT will decrease during higher flows, and the treatment performance will vary over time. HRT provides no allowance for short–circuiting, dead zones, or uneven flow distribution. Ideal hydraulic behaviour (plug flow) needs to be considered e.g., by designing a bioreactor with an appropriate aspect ratio and hydraulic gradient (discussed in the following section).

11. Estimate the seasonal (winter) nitrate–N load removed (kg ha^{-1} $season^{-1}$) by multiplying the targeted load reduction (50%) by the daily nitrate–N flux (g d^{-1}), the number of days in the winter season (92), the size of the catchment (ha), and 0.001 to convert from g to kg.

Overall, this approach assumes full treatment of drainage flow with no bypass, whereas practical systems often include bypass structures to handle peak flows that exceed the treatment capacity of the bioreactor. The calculated performance represents a steady–state, simplified design approach, best suited for screening–level sizing of bioreactor woodchip volumes and comparative assessments. It is recommended that these are refined for detailed, site–specific designs. For example, the actual volume of woodchip required might be +20% of the estimated saturated volume, which accounts for fluctuating (higher) water levels associated with peak flows and breakdown and settling of the woodchip over time. Moreover, site–specific details may limit the maximum size that a bioreactor can be based on logistical constraints (e.g., the maximum size based on the total volume of woodchips that can be transported in a lorry; the maximum reach of a digger arm, etc).

Table 5: Example spreadsheet model for sizing estimating the volume of reactive woodchip for a bioreactor based on median winter season groundwater flux. Default parameter values used in the model are shown below the table and can be adjusted to reflect site-specific conditions. The method integrates published groundwater recharge estimates, drainage assumptions, and bioreactor performance metrics derived from New Zealand case studies.

Design sheet based on median daily winter flow										
Median winter season ground-water flux (mm season^{-1})	Midpoint (mm season^{-1})	Median daily ground water flux (mm d^{-1})	Median daily groundwater flow (m^3 d^{-1})	Median winter subsurface drainage flow (m^3 d^{-1})	Median winter subsurface drainage nitrate-N load (g d^{-1})	Woodchip volume (m^3) to remove Y% nitrate-N yield @ X g m^{-3} d^{-1} average removal	WB size relative to catchment (m^3 ha^{-1})	Bioreactor size as % of drained catchment area	Average design HRT achieved (h)	Drainage nitrate-N load removed over winter season (kg ha^{-1})
101-150	125	1.36	13.59	8.15	41	10.2	10.2	0.10%	15	1.89
151-200	175	1.90	19.02	11.41	57	14.3	14.3	0.14%	15	2.62
201-250	225	2.45	24.46	14.67	73	18.3	18.3	0.18%	15	3.36
251-300	275	2.99	29.89	17.93	90	22.4	22.4	0.22%	15	4.14
301-350	325	3.53	35.33	21.20	106	26.5	26.5	0.26%	15	4.88
351-400	375	4.08	40.76	24.46	122	30.6	30.6	0.31%	15	5.61
401-450	425	4.62	46.20	27.72	139	34.6	34.6	0.35%	15	6.39

Parameters to adjust

92	days/season
1	Subsurface drain catchment area to be treated (ha)
60%	% groundwater that generates subsurface drainage
5	Average drainage nitrate concentration (g m^{-3})
2	X = seasonal saturated volumetric N reduction rate (g m^{-3} d^{-1}) Estimated mass removal rate at 15˚C for New Zealand pastoral drainage from Figure 1.
50%	Y = percent nitrate-N reduction targeted
50%	% woodchip porosity

6.5 Bioreactor shape and woodchip media

Bioreactors are typically rectangular shaped with elongated length: width ratios of at least 2:1 to 4:1. The cross-sectional geometry is typically rectangular or trapezoidal. In the case of trapezoidal bioreactors, creating sloped sides with a gradient of 1:1 helps to prevent the sides caving in during the construction or woodchip recharging process. The length of the bioreactor influences the hydraulic retention time, whereas the cross-sectional area influences the volume of flow that can pass through the bioreactor. In practical terms, wider bioreactors can more easily pass greater volumes of flow, but they require good distribution of flows at the inlet and outlet to maintain hydraulic efficiency. In contrast, larger volumes of water cannot be pushed through a long, narrow bed.

Within a bioreactor, drainable (effective) porosity—the proportion of the woodchip volume that can freely transmit water—is a key control on hydraulic performance and therefore nitrate removal. While bulk density (mass per unit volume) influences porosity, it is the available pore space and connectivity that directly determine flow paths, residence time, and treatment efficiency (Christianson et al. 2023; Johnson et al. 2022).

Drainable porosity is affected by woodchip characteristics such as species, age, particle size and shape, as well as handling and installation factors including settling, compaction, and moisture content. Coarse woodchips (e.g., nominal chip size 20–50 mm) are generally preferred because they maintain higher drainable porosity, are less prone to compaction, and provide more stable hydraulic performance than finer materials. Field measurements have shown that variability in bulk density within installed bioreactors can translate to differences in porosity and permeability, reinforcing the importance of media selection and careful installation (Christianson et al. 2023; Johnson et al. 2022).

Coarse woodchips are also more resistant to clogging than fine woodchip particles. This is important because where subsurface drainage transports sediment into the bioreactor, clogging can occur within a few years. Otherwise, bioreactors over five years old may be more vulnerable to clogging. Importantly, woodchips should be from untreated wood and not green or fresh wood, as these can be sources of nutrients.

7 Construction

Constructing a bioreactor may require various permits and regulatory approvals. Therefore, before beginning any work, consult your local council to determine the specific obligations. Before excavation, identify any existing underground services by contacting electricity, water, and telecommunications providers. It is also advisable to engage with local Iwi to ensure areas or objects of cultural significance are not affected. Seek expert guidance prior to construction to confirm that the bioreactor's location and dimensions are appropriate, considering soil characteristics, groundwater conditions, and the subsurface drainage hydrology and contaminant loading. Massey University's Catchment Solutions project have a video of constructing a woodchip bioreactor: https://www.youtube.com/watch?v=qS263WZFhHQ.

7.1 Timing and construction sequence

The best time to construct a bioreactor is in dry conditions during summer outside of the active drainage season. A bioreactor can generally be completed within a few days, depending on its size and any required piping or associated structures. Denitrification begins within hours of nitrate entering the system. Because denitrifying microorganisms occur naturally, no additional inoculation or 'seeding' is necessary.

7.2 Excavation and lining

- Determine the depth of the subsurface drains. The depth of the bed will need to be determined based on the location and configuration of the bioreactor relative to the subsurface drain line.
- Mark out the site and excavate the trench. For bioreactors with a trapezoidal cross-section, the sides should be battered to a 1:1 slope.
- Use a laser level to ensure the base of the bioreactor is even and at the correct slope. Maintaining a slight slope along the base of the bioreactor helps to promote complete drainage of the bioreactor outside of active flow periods. For example, Queensland bioreactor guidance recommends the base of the bioreactor should have slopes 0.4–0.5% to facilitate complete draining (Wegscheidl et al. 2021). This also helps avoid preferential flow paths or pockets of standing water.
- For sites without heavy, clayey soils: installing a woven polyethylene liner or heavy-duty plastic liner like LDPE will help to create a barrier between the water in the bioreactor and the surrounding water table. This helps with treatment by reducing mixing of treated and untreated water as the inflow moves to the outlet. Silage wrap should not be used as a liner.
- Sites with clayey soils may not require a waterproof liner, so long as the clay can be compacted adequately to maintain an adequate seal. A lower specification geotextile liner would then be sufficient to keep the soil and woodchips separate and reduce the potential for soils to be dispersed into the woodchip bed.

7.3 Diversion of flow from artificial drains to off-line bioreactor

Two different options for redirecting drainage flows into an offline bioreactor bed are illustrated in Figure 7-1. These allow for excess drainage flows exceeding the hydraulic capacity of the bioreactor to continue down the original drainpipe. Water level control boxes commonly used to achieve this in the United States of America are shown in Appendix D.

Where bioreactors intercept surface drains, a sedimentation or settling pond upstream of the bioreactor inlet is strongly recommended to remove suspended sediment and reduce the risk of clogging (Figure 6-4). For flashy subsurface drainage systems, particularly in catchments with strongly storm-driven, episodic flows, incorporating a small storage or detention pond upstream of the bioreactor can improve performance by buffering peak flows, smoothing hydraulic variability, and increasing effective contact time within the bioreactor. These upstream storage elements function as part of a simple treatment train, helping to stabilise inflows, protect bioreactor media, and increase the proportion of drainage water that can be effectively treated.

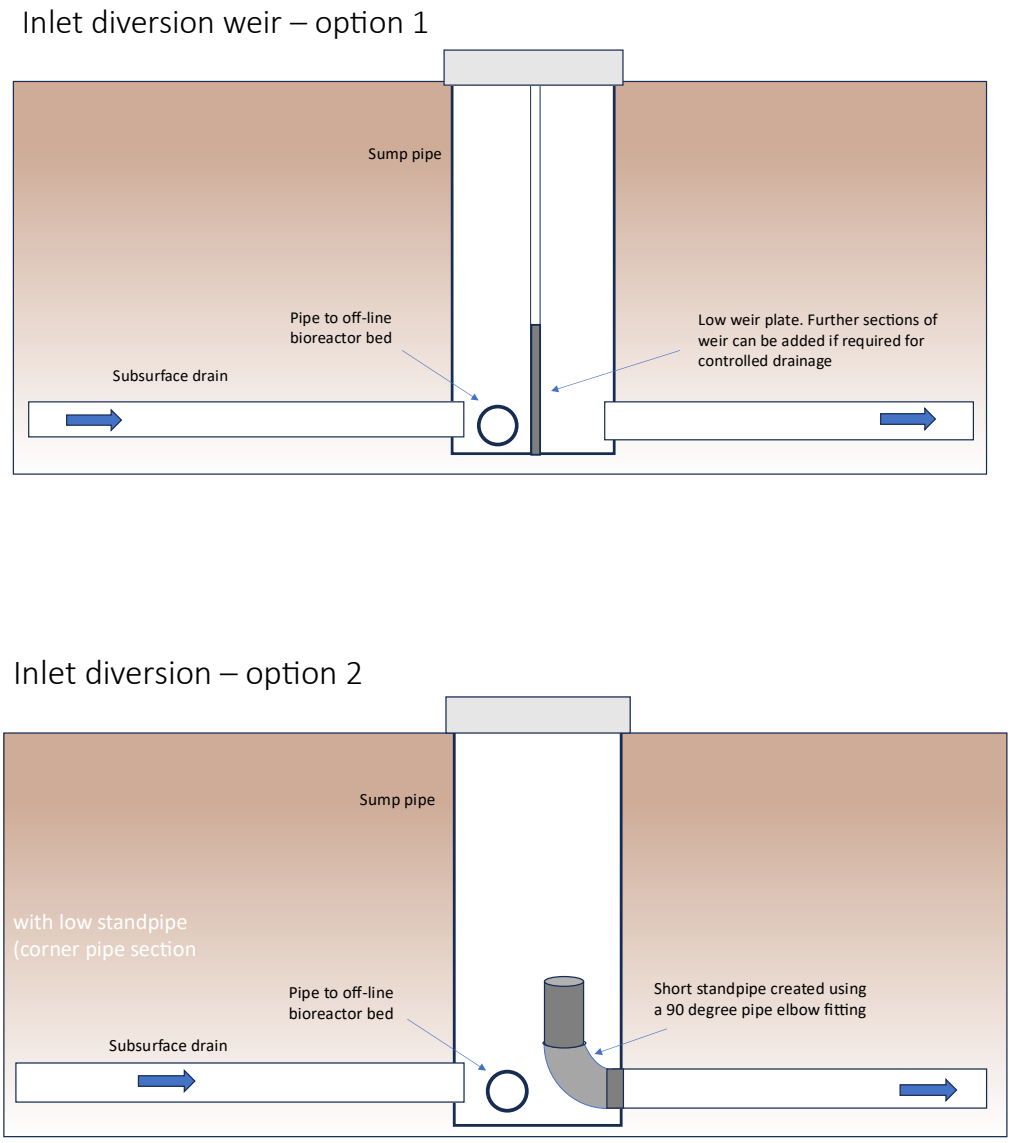


Figure 7-1: Recommended inlet diversion weirs to direct subsurface drainage into an off-line bioreactor. In option 2, the standpipe can be extended to accommodate access for cleaning out with a water blaster.

7.4 Inlet and outlet flow manifolds and water level control structures

- Install the inlet and outlet flow manifolds to the subsurface drainage. To ensure good lateral dispersion of flows across the width of the bioreactor bed and recollection at the outflow end, we recommend using relatively wide bore perforated pipes (200–300 mm) as shown in Figure 6-1. This minimises resistance to flow and good dispersion with additional capacity for sediment accumulation. Also, using relatively wide bore pipes offers the potential to clean these out if required, with minimal clogging of the woodchips. It is recommended that when the bed width is greater than 10 m, the inflow pipe is divided into two via a Y connector or weir box, which should be repeated every 10 m of bed width, so the lateral flow required each side of an inflow is only ~5 m.

- The configuration of the outflow structure is shown in Figure 7-2. This is ideally set in a bed of coarse gravel to anchor the pipework and avoid potential buoyancy. It also provides a final physical filter that protects the outlet pipe from blocking up. The operational water level in the bed is controlled by the height of outflow water level control, i.e., the top of the standpipe. It needs to be set at an appropriate height to keep the maximum volume of woodchips saturated during operation, whilst providing a hydraulic gradient to drive flow through the bed accounting for the resistance to flow through the woodchip media.

- Where there is insufficient throughflow (or inadequate provision to slowly drain between flow events) bioreactors will be prone to generating sulphurous odours and toxic, deoxygenated, highly reduced (anaerobic) discharges that can potentially impact on the health of the receiving water. Therefore, it is advisable to either (1) closely monitor the bioreactor during low flow periods so the outlet level can be reduced as needed or (2) follow recent United States of America practice to allow for gradual drawdown of water in the bioreactor when inflows cease. This can be achieved by creating a small orifice in the standpipe or base of the lowest weir plate that will allow for gradual drawdown of the water level over a period of ~3 days when there is no inflow. This orifice will need to be checked periodically (minimum annually) to remove any blockages.

- If the bioreactor is located adjacent to a sensitive waterway, a structure to reoxygenate the outflow — such as a rock pile at the outlet or rock riffle downstream— should be constructed. For post-treatment re-aeration of bioreactor outflows, Danish guidelines recommend maintaining a 0.6 m elevation drop from the bioreactor outflow to the receiving waterway (Plauborg et al. 2021). Queensland guidelines recommend installing rock riffles downstream of the outlet if the bed is located immediately upstream of an environmentally sensitive waterway (Wegscheidl et la., 2021).

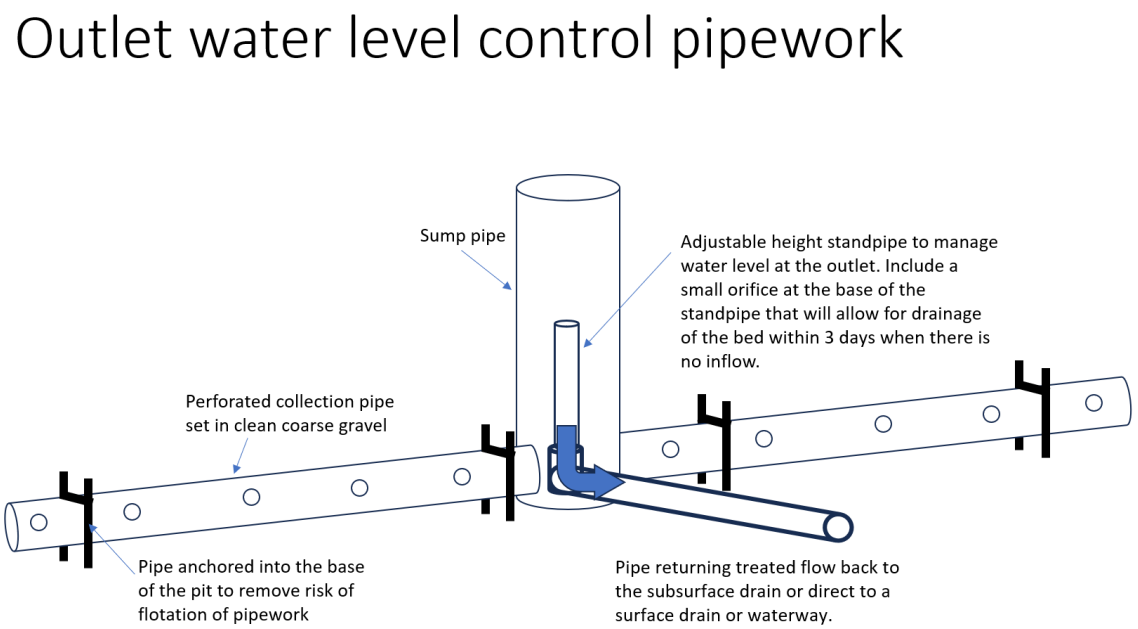


Figure 7-2: Example configuration of a bioreactor outlet water level control structure and pipework.

7.5 High flow bypass

- With the recommended inlet diversion structures, high-flow bypass occurs when water backs up enough to overtop the weir or inlet standpipe. Set the diversion weir or standpipe high enough to maximise flow through the bioreactor while maintaining the performance of the upstream drainage system.

7.6 Woodchip media

- Add the woodchips into the bioreactor trench with a front loader or backing up a tip trailer, or by using a woodchip blower. Avoid compacting the woodchip media. Try to spread the woodchips evenly to mitigate risk of heterogeneous packing. This also helps maintain flow uniformity and reduces the risk of inducing preferential flow.
- *Pinus radiata* has been used and is recommended for New Zealand bioreactors. Generally, woodchips from softwood are preferable and have equivalent nitrate removal performance to woodchips from hardwood, with lesser risk of DOC and GHG emissions (Cameron and Schipper, 2010; Manca et al. 2020).
- Coarse woodchips with a nominal chip size 20-50 mm are recommended for improved hydraulic conductivity (Johnson et al. 2022). Either chipped or hogged wood is acceptable, so long as it is free from dirt, twigs, and leaves.

7.7 Covering

- Putting a geofabric cover over the woodchip layer will help prevent sediment buildup and reduce the risk of clogging in the media. This also allows the woodchip layer to be easily uncovered later in the project if needed.
- Adding a topsoil cap of at least 0.1 m depth and no deeper than 0.5 m will prevent degradation of the geotextile cover and to enable grass growth on the surface. Soil on top of the bioreactor may also minimise the risk of nitrous oxide and methane emissions from the bioreactor. Danish guidelines specify having a layer 0.3–0.5 m of unsaturated woodchip on top of the saturated zone to support methane oxidation and reduce GHG emissions (Plauborg et al. 2021; Plauborg et al. 2023).

7.8 Indicative cost

Indicative ranges for costs to construct a woodchip bioreactor based on 2026 (pre-oil crisis) estimates are shown in Table 6 based on a nominal 100 m^2, 1 m-deep, off-line bed. Costs in the range of \$350–\$750 m^{-2} are broadly similar to the range of \$289–\$535 m^{-2} estimated by Matthews et al. (2024) for bioreactor construction, excluding any allowance for consenting.

Table 6: Estimate of costs for building a 100 m^2 woodchip bioreactor containing 100 m^3 woodchip. (rates 2026 NZD excluding GST).

Item	Indicative cost	Cost range \$ per 100 m^2 (2026)
Site survey, planning, and design.	Lump sum.	\$1,000–\$5,000
Liaison and consenting.	Lump sum.	\$500–\$5,000
Project/construction management.	Lump sum.	\$1,000–\$2,000
Site establishment.	Lump sum.	\$500–\$1,000
Earthworks including clearing site, excavation for connection to subsurface drain and placement of woodchip.	\$6.50 m^{-2} for initial site clearance; \$16 m^{-3} for excavation; \$12 m^{-2} for placement.	\$3,500–\$6,000
Liner and installation (only needed where soils do not have sufficient water holding capacity due to low clay content).	LDPE @ \$ 12–16 m^{-2} assuming 1.6x larger surface area needed to cover the sides and anchoring. Woven polyethylene or polyltarp is a cheaper alternative @ \$6–9 m^{-2}.	\$2,000–\$2,500 for LDPE OR \$1,000–\$1,500 for polytarp
Weirs and pipework (control structures).	Lump sum.	\$2,000–\$4,000

Item	Indicative cost	Cost range $ per 100 m^2 (2026)
Woodchips including transport to site.	Coarse pine woodchips with 20–50 mm nominal chip size at $60–$100 m^{-3} delivered.	$6,000–$10,000
Maintenance.	Not allowed for.	
Contingency (10%).		$18,000–$40,000
Total Cost per 100 m^2		**$35,000–75,000**
Total cost per m^2		**$350–$750**

8 Operation and maintenance

Bioreactors generally require little maintenance once they are installed. Because bioreactors operate passively, the operational requirements are minimal. However, responsible persons will need to periodically check-in and perform maintenance on the bioreactor at key periods to ensure optimal performance.

8.1 Start-up

The start-up phase refers to when a bioreactor is first operational after its installation.

- If the surrounding area will be grazed, fence off the bioreactor to exclude livestock. This will also help to mark its location and prevent unintended damage to equipment.
- If the bioreactor is unfenced, there is a potential reduction in the operational lifespan due to compaction by livestock, although further research is needed to measure this. For unfenced bioreactors, ensure that the flow control structures and corners of the bioreactor are marked to prevent damage from farm equipment.
- Inspect the control structures to make sure that water can flow freely.
- Once the drainage is flowing, make sure that water is flowing freely through the bioreactor (check that it is flowing out). Under normal flow conditions, the amount of water flowing out should be roughly the same as the amount flowing into the bioreactor.

8.2 Drawdown during no flow periods and re-wetting after the no flow period

During the drainage season, which is typically May through October in New Zealand:

- Ensure all inlets remain clean and protected to prevent sediment, algae, or debris from entering the bioreactor. Remove any obstructions impeding flow. Carry out repairs whenever problems are identified.
- During high flow conditions (during or immediately after heavy rain events), check that the high flow bypass is operating as intended. Look for signs of blowouts or other damage in the drainage system.
- If drainage backs up following heavy rain, the outlet standpipe can be lowered to reduce retention time in the bioreactor (at the cost of sacrificing performance during high flow).

Drawdown at the end of the drainage season:

- In late spring when drainage has stopped, drain the water from the bioreactor. This can be done manually or by using a small orifice at the base of the standpipe in the adjustable outlet water level control structure.

In some situations, pumping may be required if the base of the bed is below the lowest point with the orifice at the base of the outlet standpipe.

- Check that the bioreactor is not holding water (e.g., the inflow structure should be dry, and there should not be any water flowing from the outlet).
- If you consistently smell a rotten egg smell, that is a sign that sulphur reduction is occurring and there is stagnant water in the bioreactor. Make sure that the bioreactor is completely drained.

Preparing the bioreactor for re-wetting at the start of the drainage season:

- Prepare the bioreactor for re-wetting ahead of the autumn drainage season.
- Inspect the control structures for damage. Repair or replace components as needed.
- Inspect the area around the control structure, buried pipes, and the outlet for any signs of erosion or ground settling. Restore any affected areas by refilling and regrading as necessary.
- Remove any debris that has accumulated on or around the control structures.
- Check the outlet to make sure that water can flow freely. Look for sediment or debris in the structure that could impact flow and clean out if needed.
- When water starts flowing into the inlet structure and has had sufficient time to fill the bioreactor, check that it is also flowing out. The amount of water flowing out should be roughly the same as the amount flowing into the bioreactor. Note also if there is excessive diversion flow due to water backing up at the inflow weir. If the inflow/outflow to the bioreactor is noticeably below design guidelines with bypass flow evident, this might indicate that it is time to replace the woodchips.

8.3 Replacing the woodchips

- Breakdown of the woodchips is a normal process, but in some cases, woodchips may need to be replaced due to sediment clogging.
- The woodchips will break down, so they will need to be replaced approximately every 10 years.
- Subsidence of woodchips and a consistent reduction in the amount of outflow relative to the inflow are good indicators that the woodchips may need to be replaced (Schaefer et al. 2021; Christianson et al. 2025b).
- Monitor the bioreactor for sinking or subsidence. If water ponding is observed over the surface of the bioreactor, additional material can be added.

8.4 Unclogging the inlet zone

Woodchips at the bioreactor inlet may need to be replaced due to clogging with sediment. This should be done before the start of the drainage season, to ensure good dispersion of flow and internal mixing of flows through the bioreactor.

- Where a wide lateral pipe has been used to disperse the inflow, much of the sediment may accumulate in the inlet pipe and be readily cleaned out e.g., by using a water blaster inserted into the standpipe.
- If the bioreactor is covered with a soil cap, excavate the area over the inlet zone to expose the inlet pipe and woodchips.
- If a section of the woodchips is clogged, excavate them with a digger. Be careful not to damage the inlet manifold or the liner.
- Inspect the inlet zone and inlet manifold for any signs of excessive wear or clogging. Replace any damaged pipes and infrastructure as needed.
- Re-fill the bioreactor with woodchips. If the bioreactor has a soil cap, replace the soil over the bioreactor. Be careful not to compact the underlying woodchips, as this will impede drainage through the bioreactor.

8.5 Enhancing treatment performance

The nitrate removal performance of woodchip bioreactors can decline over time as the labile carbon supply from woodchips diminishes, particularly under episodic flow conditions (Rivas et al. 2020). Several enhancement strategies have been investigated to address these limitations. Moghaddam and Christianson (2025) provide guidance for selecting bioreactor enhancement strategies based on site-specific requirements, performance targets, and economic constraints while addressing potential environmental impacts. Examples of bioreactor enhancement strategies include:

- Biostimulation by providing additional readily available soluble carbon sources (e.g., methanol or acetate dosing: Moghaddam et al. 2023; Feyereisen et al. 2023) or electro-chemical energy via embedded electrodes (Yang et al. 2021). Carbon dosing is the most extensively studied enhancement approach. However, carbon dosing requires careful management to avoid undesirable side effects, including increased sulphate reduction under nitrate-limited conditions and potential clogging of the woodchip media (Moghaddam et al. 2022; Moghaddam et al. 2023).
  - In a pilot-scale field trial on a dairy farm in the Waikato, constant methanol dosing increased seasonal nitrate removal rates from 0.1–1.6 g N m^{-3} d^{-1} (un-dosed, 2018) to 8.6 g N m^{-3} d^{-1} (dosed, 2020) and 5.1 g N m^{-3} d^{-1} (dosed at a halved rate, 2021) (Moghaddam et al. 2023). Under controlled mesocosm conditions, methanol dosing increased removal rates from 7 to 28 g N m^{-3} d^{-1}, with an optimal carbon-to-nitrogen (C:N) ratio of 0.7 identified for maximising nitrate removal while minimising excess carbon losses (Moghaddam et al. 2022).

Methanol removal efficiency exceeded 90–99%, indicating low risk of methanol discharge to receiving waters (Moghaddam et al. 2023).

- Using additional and/or alternatively more "biologically degradable" carbon sources to promote increased nitrate removal. Where woodchips are substituted or supplemented with other carbon-rich materials (e.g., biochar, corn cobs, or other lignocellulosic materials), bioreactors have achieved removal rates of 12.8–15.2 g N m^{-3} d^{-1} in some studies, exceeding the rates reported for carbon dosing alone. E.g., Law et al. (2023) trialled woodchip bioreactors amended with corn cobs; however, these materials may have different longevity, hydraulic properties, and pollution swapping profiles compared to woodchips.
- Bioaugmentation by inoculating with specially adapted or functionalised strains of microbes. (e.g., cold-adapted bacteria or wood-rot fungi; Feyereisen et al. 2023; Wang et al. 2025).
- Chemically (e.g., iron or aluminium composites) or thermally (e.g., biochar) modifying woodchips to promote denitrification and retention of phosphorus (e.g., Perera et al. 2024b) and other contaminants (e.g., pesticides and pharmaceuticals; Zhou et al. 2024).
- Other enhancement strategies include the addition of phosphorus-sorbing media (e.g., to counteract potential DRP leaching), biochar amendments, controlled drainage to manage hydraulic loading, and treatment train configurations that combine woodchip bioreactors with constructed wetlands or other interceptive mitigations.

Enhancement strategies remain an active area of research. Multi-year field validation (>5 years), improved understanding of microbial dynamics during carbon dosing, and systematic scale-up from laboratory to field conditions are identified as critical research priorities (Moghaddam & Christianson, 2025).

9 Key indicators of success

A well-functioning woodchip bioreactor will have:

- Lower nitrate levels in the outflow than the inflow.
- Low sediment accumulation rates in the inlet.
- Evenly distributed flow without short-circuiting or dead zones.
- Appropriate water levels to maintain woodchip saturation and treatment function during operational conditions.
- Little or no water ponding above the bioreactor (except during extremely high flow events that produce an excessive water volume in the bioreactor).
- Little or no waterlogging of soils in the immediate vicinity of the bioreactor, except following extremely high flow events that temporarily pond water.
- Gradual settling and subsidence of the surface of the bed due to aerobic decomposition of the woodchip. Subsidence may be worse at the inlet (and sometimes the outlet) due to aerobic degradation of the woodchips.
- Consistent outflows relative to the inflows. The outflow will consistently decrease over time and the woodchip will subside with age, indicating the need to replace the woodchip.
- Outflow water is generally clear, with low odour, and does not smell like rotten eggs due to hydrogen sulphide release.
- Outflow water will have low dissolved oxygen levels <2 mg L^{-1}.

9.1 Measuring successful treatment performance

Measuring the nitrate concentrations flowing into and out of the bioreactor will help inform whether the bioreactor is effectively removing nitrate as it has been designed for. A comprehensive performance assessment requires a continuous monitoring of inflows and outflows of drainage water and its nutrient concentrations through a bioreactor. This is technically possible but requires significant technical expertise, instrumentation, and resources to implement. However, a more basic monitoring can be indicative of performance of a bioreactor, as follows:

- Use a grab sampling approach to record drainage inflow and outflow rates, along with nutrient concentrations, on a weekly or fortnightly basis. For low to moderate flow rates (e.g., less than 5 L s^{-1}), simple methods such as volumetric (bucket) measurements can be used to estimate flow. Collect samples from both the bioreactor inflow and outflow and analyse them either using a reliable field sensor or by sending them to a certified laboratory. These data can then be used to estimate average drainage flow rates, approximate drainage volumes, and changes in nutrient loads or reductions over time.

As a minimum approach, measure drainage flow rates and nutrient concentrations 3 to 5 times during each drainage season to provide a basic assessment of bioreactor performance.

10 Next steps for New Zealand woodchip bioreactor guideline preparation

Overall, this technical guidance provides a basis for implementation of woodchip bioreactors and development of associated policies and regulations based on current knowledge. It will also provide the technical basis for production and dissemination of a more end-user orientated practitioner guide in collaboration with rural industries and councils (to be developed subsequently). Further work will help to communicate and refine this guidance, for example:

- Evaluating available New Zealand subsurface drainage flow rate data to determine the optimal proportion of drainage flow to intercept for bioreactor cost-effectiveness.
- Assessing whether outputs from HYPE modelling can be used to generate an improved map of expected subsurface drainage flows, and whether seasonal values can be extracted to estimate subsurface drainage loads for the main periods of interest.
- Cross-validating drainage load estimation methods: Compare several of the recommended approaches for estimating subsurface drainage flows and nitrate-N loads using examples from New Zealand bioreactor case studies with good data on flow and nitrate-N concentrations (e.g., Waituna bioreactor). Compare these differently derived estimates against the drainage flows and nitrate-N loads that were measured at the inflows. This will help to narrow down the most suitable methods to recommend for estimating subsurface drainage loads to help size bioreactors.
- Working through different woodchip volume sizing options versus the percentage of average drainage flow intercepted, the N load removed, and the cost:benefit for an example bioreactor application.
- Developing a risk assessment framework and recommendations for minimizing unwanted pollution swapping effects during different bioreactor operational phases (start-up and drying and re-wetting cycles). This will assist practitioners with improving treatment efficacy and minimising unwanted side effects.
- Communicating these technical recommendations for woodchip bioreactor design and performance and seeking feedback from other end-user groups (e.g., rural industries, agricultural engineers, council planning staff, etc). These steps are required to produce a more practitioner-focused set of guidelines.

- Enhancing the scientific credibility of the design and performance recommendations by publishing them in the international, peer–reviewed literature. Scientific endorsement will help to boost the credibility of bioreactors in regulatory contexts and support their use in farm plans and catchment nutrient limit setting.

Finally, and perhaps most importantly, over the long term, this guidance needs to be field tested. This includes designing, implementing, and monitoring the field–scale performance of woodchip bioreactors built according to this design guidance. Evaluating the performance information, ideally from multiple years and multiple sites, is critical to refining and improving the guidance and recommendations.

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12 Glossary of abbreviations and terms

Arrhenius coefficient (θ)	Empirical temperature correction factor used to adjust reaction rate constants for temperatures other than the reference temperature (typically 20°C). Unit: dimensionless.
Bulk density	Dry mass of woodchip media per unit bulk volume, including pore spaces; used to characterise woodchip compaction and storage properties. Unit: kg m^{-3}.
Bypass flows	Water flows that circumvent the intended treatment pathway, reducing overall treatment performance. Units: m^3 d^{-1} or L s^{-1}.
Denitrification	Microbial process that converts nitrate (NO_3^-) to gaseous nitrogen species (N_2O and N_2) under low-oxygen conditions, permanently removing nitrogen from water. Typically expressed as a removal rate in g N m^{-3} d^{-1}.
Drainage capacity	Design flow rate that a drainage system can convey or remove from a specified land area. Units: m^3 ha^{-1} d^{-1} or L s^{-1} ha^{-1}.
Drainage coefficient	Design rate at which excess water is removed from a drained area over a defined period. Unit: mm d^{-1} (equivalent to m^3 ha^{-1} d^{-1}).
Effective porosity	Fraction of total soil or media volume occupied by interconnected pores that actively transmit water and contribute to storage. Units: m^3 m^{-3} or %.
Flow-proportional concentration	Concentration weighted according to the flow volume associated with each sample period, representing the average contaminant concentration transported by the flow. Units: mg L^{-1} or g m^{-3}.
Flux (instantaneous load)	Mass of a contaminant transported past a point per unit time at a specific moment; commonly calculated as concentration multiplied by flow rate. Units: g d^{-1} or kg y^{-1}.
HDPE (High-Density Polyethylene)	Thermoplastic polymer widely used in drainage pipes, geomembranes, liners, and water-management infrastructure due to its high strength, chemical resistance, durability, and low permeability.
Hydraulic efficiency	Measure of how effectively flow is distributed through a treatment system relative to ideal plug-flow conditions, accounting for short-circuiting and dead zones. Unit: dimensionless or %.
Hydraulic head	Potential energy of water expressed as an equivalent water column height, which drives groundwater and surface-water movement. Unit: m.
Hydraulic retention time (HRT)	Average time water remains within a treatment system, calculated as effective storage volume divided by flow rate. Units: h or d.

Hooghoudt equation	Analytical equation used to estimate subsurface drainage intensity and drain spacing by relating hydraulic conductivity, drain depth, drain spacing, and groundwater level between drains. Units depend on application; drainage intensity is commonly expressed as mm d^{-1}, while hydraulic conductivity is expressed as m d^{-1}.
HYPE model	Hydrological Predictions for the Environment (HYPE) model; a semi-distributed process-based model used to simulate water flow, nutrient transport, sediment transport, and water quality across catchments.
K_{20}	First-order reaction rate coefficient standardised to a reference temperature of 20°C and commonly used in nutrient removal modelling. Unit: d^{-1}.
Kirkham coefficient	Drainage design parameter used in Kirkham drainage equations to account for seepage conditions and drainage system geometry when estimating subsurface drainage performance.
LDPE (Low-Density Polyethylene)	Thermoplastic polymer with lower density and greater flexibility than HDPE, commonly used in geomembranes, agricultural films, pipe fittings, and water-management applications requiring pliability.
Load	Total mass of a contaminant transported, received, exported, removed, or retained over a specified time period. Units: kg
Manning's equation	Empirical equation used to estimate open-channel flow velocity or discharge as a function of hydraulic radius, channel slope, and surface roughness. Manning's roughness coefficient (n) is dimensionless; discharge is expressed as m^3 s^{-1} and velocity as m s^{-1}.
MODFLOW model	Modular finite-difference groundwater flow model used to simulate groundwater levels, flow directions, hydraulic heads, recharge, discharge, and groundwater–surface water interactions.
Nitrate-N (NO_3-N)	Concentration of nitrogen present in the nitrate form, expressed as nitrogen mass rather than nitrate mass. Unit: mg N L^{-1}.
Nomograph	Graphical calculation tool consisting of aligned scales used to estimate engineering, hydraulic, or drainage-design variables without numerical computation.
Pollution-swapping	Circumstance in which mitigation of one contaminant or environmental pressure unintentionally increases another contaminant or adverse environmental effect.
Q_{10}	Temperature coefficient describing the factor by which a biological or chemical reaction rate changes for each 10°C increase in temperature. Unit: dimensionless.
Saturated volumetric removal rate	Nitrogen removal rate normalised to the effective saturated treatment volume where denitrification occurs. Unit: g N m^{-3} d^{-1}.

Slope	Gradient of a land surface, channel, or structure expressed either as an angle from horizontal or as vertical rise divided by horizontal distance. Units: degrees (°) or percent (%).
Subsurface drainage	Removal of excess soil water through drains installed below the ground surface, typically perforated pipes or tile drains. Flow units: L s^{-1} or m^3 d^{-1}.
SWAT model	Soil and Water Assessment Tool (SWAT); a semi-distributed catchment model used to simulate hydrology, nutrient cycling, sediment transport, land-use effects, and agricultural management scenarios.
TOPNET model	Topography-based rainfall-runoff model that simulates catchment hydrology using terrain controls on water movement, runoff generation, and streamflow.
Total volumetric removal rate	Nitrogen removal rate normalised to the total treatment-system volume, including hydraulically active and inactive zones. Unit: g N m^{-3} d^{-1}.
Treatment efficacy	Degree to which a treatment system reduces contaminant concentrations or loads relative to influent conditions. Typically expressed as % reduction.
Yield	Area-normalised contaminant load exported from a catchment, field, or drainage area over a specified period. Unit: kg ha^{-1} y^{-1}.

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Appendix A Nomographs for estimating peak flows from subsurface drainage pipes

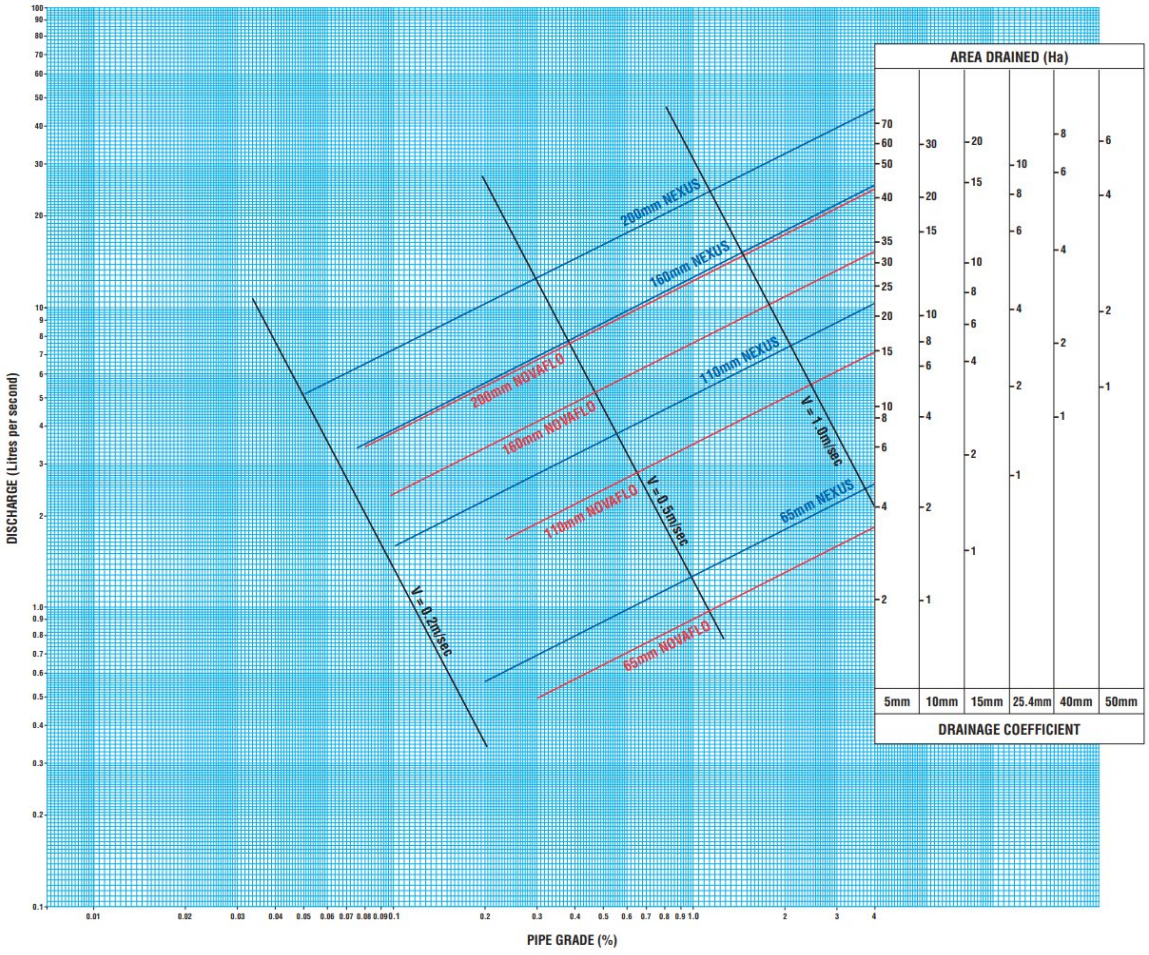


Figure A-1: Modern Nexus and Novaflo plastic drainage pipe nomograph..

The nomograph above provides an estimate of the maximum discharge from single and double-walled plastic drainage pipe of different internal diameters set at different gradients. Novaflow is single wall corrugated plastic drainage pipe and Nexus is double-walled (i.e., with a smooth interior) (iPLEX, 2025). Reproduced from: https://www.iplex.co.nz/assets/Uploads/Flow-Chart-Iplex-Nexus-Novaflo.pdf

Bowler (1980) suggests a minimum recommended grade of ~0.25% for 100 mm pipe and 0.2 for 150 mm pipe, so assuming these for 110 and 160 mm respectively the maximum flow expected would be ~1.8 and 0.35 L s^{-1} for single-walled pipe and 2.5 and 5.5 L s^{-1} for double-walled. Using the table to the right, shows what areas of land could be drained. Assuming a required drainage co-efficient of 25.4 mm, suggests these pipes could drain <1, 2, 1.5, and 2 ha, respectively.

The drainage co-efficient is the depth of drainage in mm required to be drained in 24 hrs. Bowler (1980) suggests 15-25 mm for intensive dairy farms and 10-15 mm for lower intensity farms.

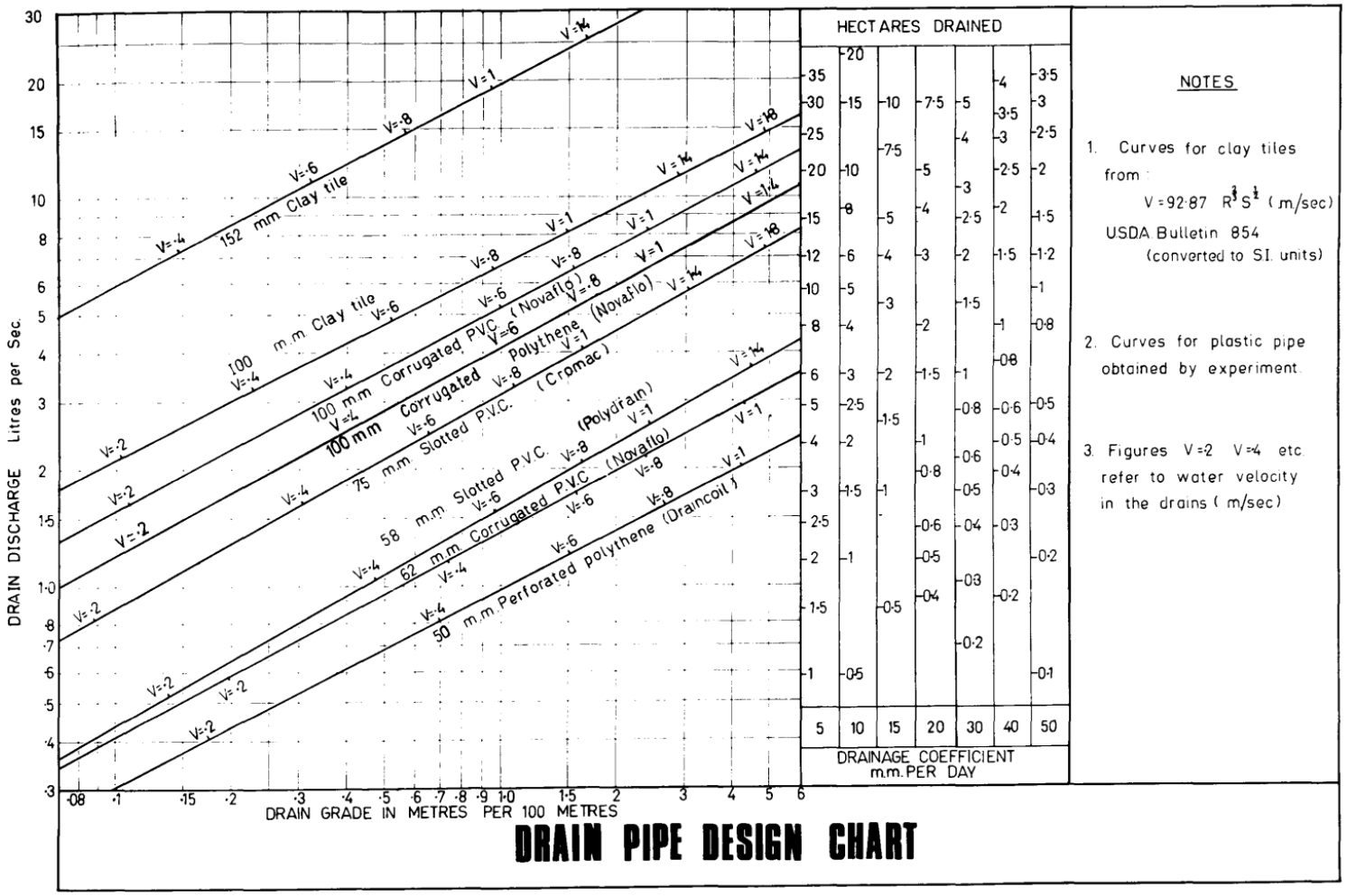


Figure A-2: Old clay tiles versus PVC drainage pipes From Bidwell (1978) Field drainage guide. Reproduced from: https://researcharchive.lincoln.ac.nz/server/api/core/bitstreams/73a38653-dc92-4f53-b555-6015724acd40/content..

Appendix B Spatial planning resources

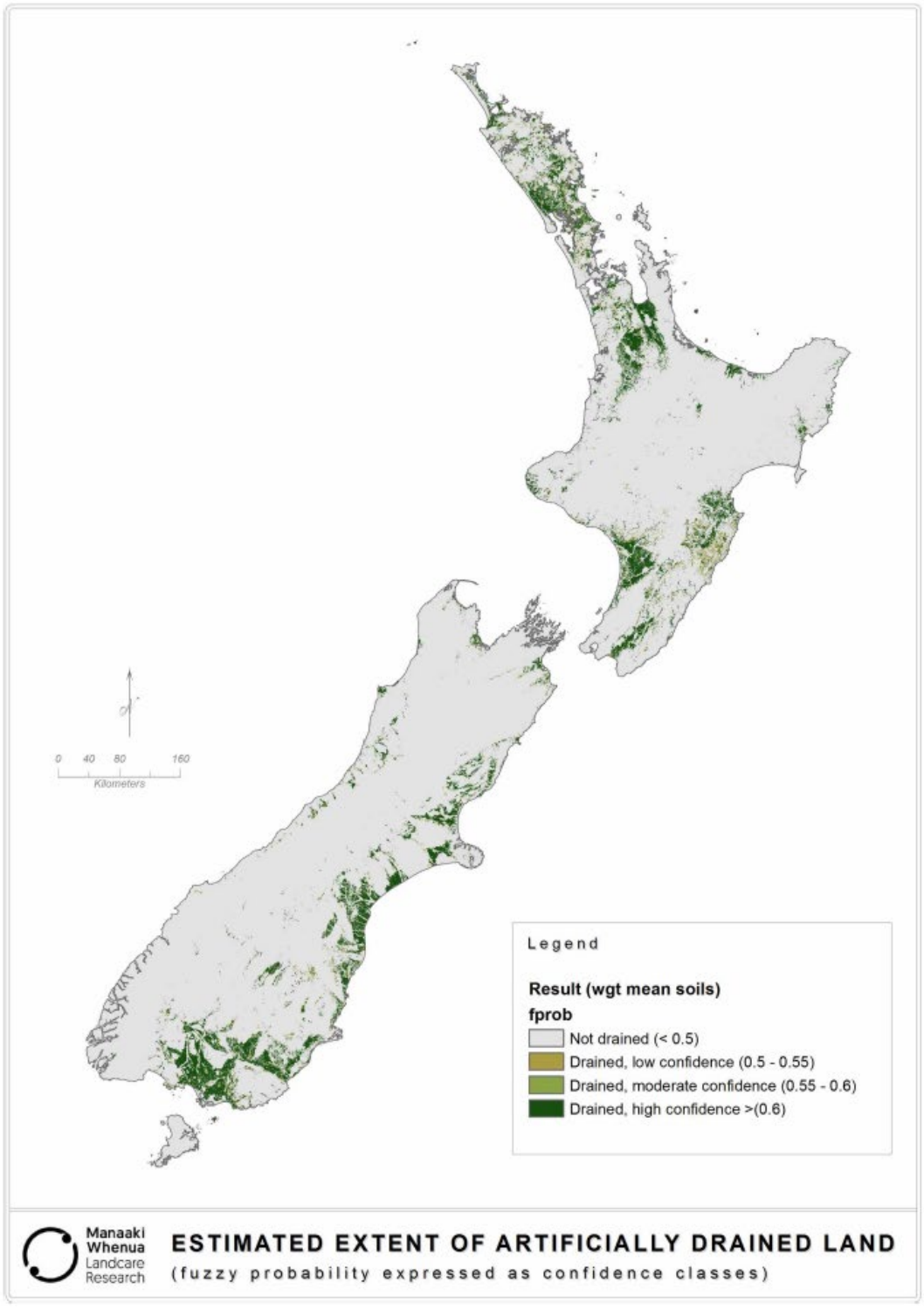


Figure B-1: Manderson (2028) artificial drainage map..

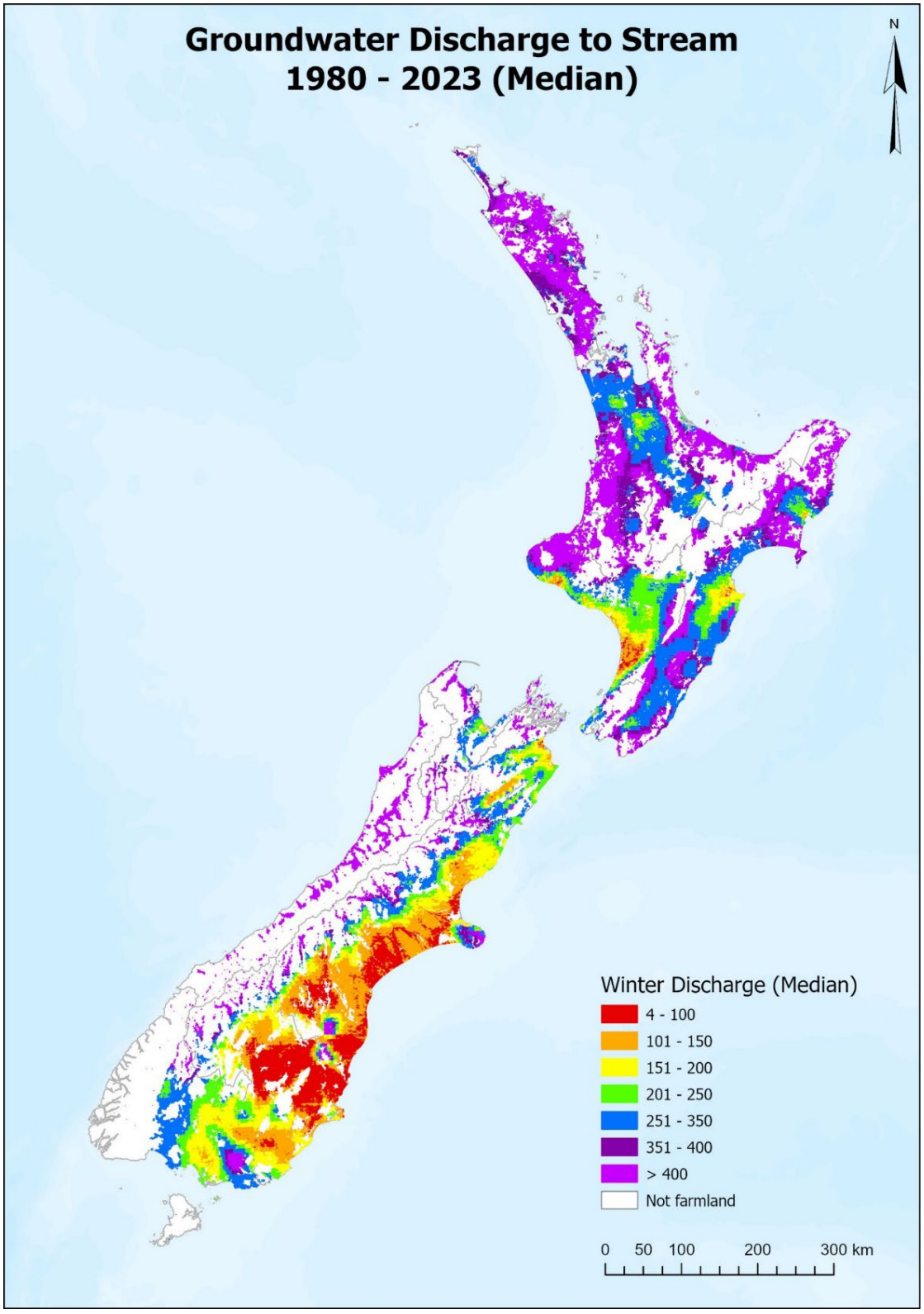


Figure B–2: Map of Aotearoa New Zealand daily median winter groundwater flux (mm d–1) for the period 1980–2023. The map generated by Christian Zammit (ESNZ) based on Top Net predictions from Clark et al. (2008), Booker & Woods (2014), and Cattoën et al. (2022)..

Appendix C Case studies

13.1.1 Tatuanui, Waikato

Location:	Tatuanui (near Morrinsville), North Island
River catchment:	Waitoa River catchment (drains to the Waihou River and the Firth of Thames)
Year constructed:	Installed March–July 2017; first operational drainage season 2017
Bioreactor configuration:	▪ Off-line, located at the edge of a tile-drained paddock. Subsurface tile drain feeds the bioreactor inlet via a Y-junction. ▪ A high-flow bypass returns excess flow to the original drainage line. Treated outflow discharges to a farm drain. ▪ Used as a research platform across two phases: Phase 1 (2017–2019) conventional, un-dosed woodchip bioreactor (Rivas et al.; 2020, 2023); Phase 2 (2020–2021) constant-rate methanol dosing as an enhancement strategy (Moghaddam et al. 2023).
Bioreactor catchment area:	▪ 0.65 ha catchment of intensively grazed dairy pasture. ▪ Drainage type: single subsurface tile drain (perforated PVC), ~0.7–0.9 m below ground surface. ▪ Bioreactor footprint represents ~18.5% of the contributing catchment area (at the upper end of New Zealand examples). Sized at 87 m^3 ha^{-1} catchment in Phase 1, reducing to 38.5 m^3 ha^{-1} in Phase 2 with a lower water-level setting.
Bioreactor size and woodchips:	▪ Trapezoidal shape with the sides battered ~1:1. ▪ Surface area of 120 m^2 and total bioreactor volume of 78 m^3. ▪ Average active (saturated) woodchip volume: 56.3 m^3 in Phase 1 (2017–2019), 25 m^3 in Phase 2 (2020–2021, lower water-level setting). ▪ Drainable porosity ~48% (measured at end-of-season drainage). ▪ *Pinus radiata* woodchips, coarse (typically 0.8–13.1 mm diameter)
Project scope:	▪ The Tatuanui bioreactor was installed as part of the SSIF-funded 'Enhanced Mitigation of Nitrate in Groundwater' programme led by ESR in collaboration with Lincoln Agritech Ltd., Aqualinc Research Ltd., Southern Geophysical, and University of Canterbury to evaluate woodchip bioreactor performance under New Zealand pastoral subsurface drainage conditions. ▪ Site selection criteria: a tile-drained catchment with measurable drainage, sufficient slope for gravity-fed flow, and a willing host farmer. ▪ Performance was monitored each drainage season via flow-proportional or composite sampling at the inlet and outlet, with continuous flow gauging. Spot sampling encompassed nitrate-N, ammonium-N, dissolved organic carbon, sulphate, temperature, and dissolved oxygen.
Approximate cost*:	• $$$ ($40–80K): excavation, plastic liner, woodchips (*Pinus radiata*), inlet/outlet manifolds, water-level control structures, high-flow bypass, fencing. Cost excludes research-grade monitoring equipment. Phase 2 added ~$5–10K for chemical-resistant pump, dosing tank, flow-proportional control, methanol container and weather shelter.

Bioreactor performance:	▪ Phase 1 — Woodchip-only (no carbon dosing). 2017 (Rivas et al. 2020): Inflows were compromised by a geotextile that was put on the inlet manifold. This resulted in very low inflow nitrate yield (3.0 g N m^{-3} d^{-1} yr^{-1}), very low mass removal, but high (over-treatment) of the influent water with a 99% nitrate-N relative load reduction over 106 days of flow, with ~17% of the inflow yield bypassed the bioreactor). The nitrate removal rate was not reported (but outflow nitrate-N was at or below the detection limit). 2018 (Rivas et al. 2020) the inflow plumbing issues were resolved, resulting in greater loading and mass removal: 19.2 kg N ha^{-1} yr^{-1} inflow yield (~1% of the inflow yield bypassed the bioreactor), removal rate 0.71 ± 0.09 g N m^{-3} d^{-1}, and a 48.2% relative load reduction over 124 days of flow. 2019 (Rivas et al. 2023): 73% relative load reduction over 79 days of flow, 24.4 kg N ha^{-1} yr^{-1} inflow yield, removal rate 0.91 ± 0.48 g N m^{-3} d^{-1}. ▪ Phase 2 — Methanol-dosed (enhanced). 2020 (Moghaddam et al. 2023): 85% load reduction over 44 days of flow, 3.4 kg N ha^{-1} yr^{-1} inflow yield, removal rate 8.6 g N m^{-3} d^{-1} (14.4 L d^{-1} of 8% v/v methanol solution). 2021 (Moghaddam et al. 2023): 73% load reduction over 95 days of flow, 3.4 kg N ha^{-1} yr^{-1} inflow yield, removal rate 5.1 g N m^{-3} d^{-1} (dosing rate halved). ▪ Summary ranges: woodchip-only 0.71–0.91 g N m^{-3} d^{-1} (48–99% load reduction); methanol-dosed 5.1–8.6 g N m^{-3} d^{-1} (73–85% load reduction). Methanol dosing increased volumetric removal rate ~5–10× relative to un-dosed operation. ▪ Other contaminants: Phase 1 average DRP load reduction ~44.5%. Methanol consumption efficiency >90–99% in dosed seasons (low risk of methanol breakthrough). No measurable adverse impacts on aquatic test organisms (algae, daphnia, inanga whitebait) detected from post-dosing seasonal effluent (Thompson & Goeller, 2023).
Fencing, liner, and covering:	▪ Fencing: standard farm fencing. ▪ Liner: heavy-duty plastic liner (HDPE). ▪ Covering: geotextile fabric over the woodchips, with a 0.1–0.3 m soil cap; pasture established on top.
Operating depth and active woodchip volume:	▪ Inlet and outlet manifolds with adjustable water-level control structures. ▪ Designated saturated operating depth ~0.65 m in Phase 1 (woodchip-only); ~0.21 m in Phase 2 (methanol-dosed, lower water-level setting). ▪ Average saturated woodchip volume: 56.3 m^3 in Phase 1; 25 m^3 in Phase 2. ▪ Water level monitored continuously at inlet and outlet manifolds during operational seasons.
Operational flow rates and bypass flow:	▪ Drainage line flows highly episodic and event-driven, typical of New Zealand pastoral subsurface drainage. Flow rates ranged from ~0.1 L s^{-1} during baseflow to peaks >5 L s^{-1} during storm events. ▪ Bioreactor inflow range ~0.05–0.5 L s^{-1} (design flow); excess was bypassed. ▪ Bypass occurred during peak storm events; the proportion of total annual drainage bypassing the bioreactor varied with the magnitude and frequency of storm events each season. Across the 2017 and

	2018 drainage seasons, <10% of the annual tile drainage load bypassed the bioreactor. ▪ Bioreactor was drained at the end of each spring drainage season to minimise stagnation and sulphate reduction during the dry summer period.
Additional information:	▪ The Tatuanui bioreactor is one of the longest-running New Zealand field-scale woodchip bioreactors and the only New Zealand field-scale carbon-dosed bioreactor trial. It has provided foundational data on removal rate variability under episodic New Zealand drainage flows, the benefits and trade-offs of methanol dosing as an enhancement, the optimal C:N ratio for nitrate removal (~0.7), determined from companion mesocosm work (Moghaddam et al. 2022), and evidence of negligible ecotoxicology impacts of post-season effluent on freshwater aquatic organisms (Thompson & Goeller 2023). ▪ The site has hosted multiple field days and stakeholder visits and has informed subsequent New Zealand bioreactor installations. ▪ Key references: Moghaddam et al. (2023), Rivas et al. (2020), Rivas et al. (2023), Thompson & Goeller (2023), Perera et al. (2024a)
* $ < 20K, $$ 20-40K, $$$40-80K, $$$$>100K	



Figure C-1: The top photo shows the trapezoidal shape of the Tatuanui bioreactor, with the HDPE liner and inlet and outlet manifold pipes.The geotextile fabric around the inlet manifold was removed after the first year of operation since it restricted inflows and promoted biofilm growth and clogging. The bottom photo shows the completed installation of the Tatuanui woodchip bioreactor showing the main components: (1) inlet control structure with auto sampler and stilling well, (2) outlet control structure with auto sampler and stilling well, (3) rain gauge, solar panel, and control panels for the instruments. Photo from Rivas et al. (2020)

13.1.2 Coastal Rangitīkei Sand Country, Bulls

Location:	Mixed pastoral farm, Bulls, North Island
River catchment:	Rangitīkei River Catchment
Year constructed:	August 2020
Bioreactor configuration:	▪ Offline, located on the Paddock corner, along a surface drain ▪ This was part of a controlled drainage trial conducted by Massey University (Singh et al. 2025). The inflows were held in a separate drain that helped to buffer flow variability and encourage deposition of fine sediment, before being diverted into the bioreactor.
Bioreactor catchment area:	▪ 13 ha catchment (calculated based on the surface drain length and spacing in the paddock) ▪ Surface drains, approximately 1.5 m deep and 1.5 m wide – trapezoidal shape.
Bioreactor size and woodchips:	▪ Trapezoidal shape with a 131 m^2 surface area ▪ Surface dimensions 17.5 m long x 7.5 m wide x 1.3 m deep with 1 m woodchip depth, and 30 cm topsoil cover. Aspect ratio L:W of 2.3 ▪ 90 m^3 of coarse pine woodchips with 20–60 mm chip size were sourced from the forestry slash at the farm.
Project scope:	▪ Develop and demonstrate novel drainage management technologies ▪ A field monitoring campaign was conducted for a full year (2018–2019) to establish a baseline of drain flows and water quality (nitrate and dissolved phosphorus concentrations). The collected field info was used to select the site and design the pilot woodchip bioreactor demonstration. ▪ The bioreactor performance was monitored weekly between September 2020 and July 2024.
Approximate cost*:	▪ *< $15k (includes construction, pipes, woodchips, liner, cover, and fencing). Cost excludes research-grade monitoring equipment.
Bioreactor performance:	▪ Approximately 23% reduction in the total surface drain nitrate–N load, and a 71% reduction in the nitrate–N fluxes diverted through the bioreactor. ▪ Estimated nitrate–N removal rates at 2.18 g N m^{-3} d^{-1}. Outlet nitrate–N concentrations ranged <0.001 to 3.76 mg L^{-1}, with an average of 0.42 mg L^{-1} versus inlet concentrations of <0.001 (during summer season) to 8.71 mg L^{-1}, with an average of 1.66 mg L^{-1}. ▪ The bioreactor was 1 to 4 years old when it was monitored. ▪ The field observations suggested an increase in outlet dissolved reactive and total phosphorus concentrations, especially during warm, low-flow periods. Long residence times at high temperatures may enhance internal phosphorus release. Therefore, operation of bioreactors under low flow summer conditions should be avoided in the study area.
Fencing, liner, and covering:	▪ It was fenced off mainly to protect the research monitoring equipment. Otherwise, no need to fence off from the farming operation perspective. ▪ A plastic liner (silage cover material) was used. ▪ The bioreactor was capped with a 30 cm topsoil layer and pasture grass.

Operating depth and active woodchip volume:	▪ The designated saturated operating depth was 1 m. But in practice, the operating depth was 50 cm (48 m^3 of saturated woodchip. ▪ The water level was monitored weekly from March 2022 to July 2023. ▪ The inlet depth was set at 35 – 40 cm from the soil surface ▪ The outlet height was adjustable, and it was generally set at 70 – 80 cm above the bioreactor bottom. ▪ Inflow dispersion was achieved with a set of three perforated, plastic drainpipes laid across the bioreactor width. ▪ Outflow collection was by a set of three perforated, plastic pipes laid across the bioreactor width. The outflow collection pipes drained to a sump, which was connected to the height–adjustable outflow to the adjacent surface drain.
Operational flow rates and bypass flow:	▪ The overall surface drain flow ranged from 0 (no flow) to 7.22 L s^{-1}, with an average of 1.36 L s^{-1}. ▪ The bioreactor inflow ranged from 0 (no inflow) to 2.79 L s^{-1}, with an average of 0.50 L s^{-1}. ▪ Overall, the bioreactor captured about 36% of total surface drain flow measured over the three years (Oct 2020 to July 2024). ▪ The surface drain flow remained active throughout most of the year due to the shallow groundwater level. Therefore, the bioreactor received active flow for most of the year.
Additional information:	▪ The bioreactor had its inflow from a surface drain installed in shallow groundwater conditions. ▪ There were issues with waterlogging noticed on the bioreactor's surface following its coverage with geotextile and soil layer. As a remediation, the geotextile layer was removed from the first part of bioreactor where drainage inflow pipes were spread over the woodchips. ▪ The bioreactor flows were managed to research the potential effects of different flow rates on nitrate removal efficiency. Since this was a research trial, the bioreactor performance was not optimised to achieve maximum nitrate removal (Singh et al. 2025).
* $ < 20K, $$ 20–40K, $$$40–80K, $$$$>100K	



Figure C–2: The Coastal Rangitīkei woodchip bioreactor before covering (left) and after covering with geotextile, soil, and grass (right). Photos provided by Ranvir Singh, Massey University.

13.1.3 Dannevirke, Upper Manawatū–Whanganui

Location:	Dairy farm near Dannevirke, Upper Manawatū–Whanganui
River catchment:	Manawatu River Catchment
Year constructed:	June 2021
Bioreactor configuration:	In–line, located in the gulley head, along a subsurface drain outflow
Bioreactor catchment area:	▪ 65 ha catchment used for dairy farming ▪ The subsurface tile drain has an irregular network
Bioreactor size and woodchips:	▪ Rectangular shape with a 131 m^2 surface area. ▪ Surface dimensions 15.0 m long x 11.0 m wide x 1.12 m deep with 1.12 m woodchip depth, and a mesh cover. Aspect ratio L:W of 1.36 ▪ 180 m^3 of coarse pine woodchips with 20–60 mm nominal chip size were sourced from a feed pad supplier.
Project scope:	▪ The project sought to develop and demonstrate novel drainage management technologies. ▪ A field monitoring campaign was conducted during 2020 to establish a baseline of drain flow and water quality (nitrate and dissolved phosphorus concentrations). The collected field info was used to select the site and design the pilot woodchip bioreactor demonstration. ▪ Bioreactor performance was monitored weekly between July 2021 and July 2024.
Approximate cost*:	• *< $20k (includes construction, pipes, woodchips, liner, cover, and fencing). Cost excludes research–grade monitoring equipment.
Bioreactor performance:	▪ Approximately 15% reduction in the total tile drain nitrate–N load, and a 49% reduction in the nitrate–N fluxes diverted through the bioreactor. ▪ Estimated nitrate–N removal rates were 0.68 g N m^{-3} d^{-1}. Outlet nitrate–N concentrations ranged <0.001 to 18.4 mg L^{-1}, with an average of 2.44 mg L^{-1} versus inlet concentrations of 1.26 to 20.60 mg L^{-1}, with an average of 7.41 mg L^{-1}. ▪ The bioreactor was 1 to 4 years old during the monitoring study. ▪ Field observations suggested an increase in outlet dissolved reactive and total phosphorus concentrations, especially during warm, low–flow periods. Long residence times at high temperatures may enhance internal phosphorus release. Therefore, operation of bioreactors under low flow summer conditions should be avoided in the study area.
Fencing, liner, and covering:	▪ Fully fenced off into the gulley head. ▪ The bioreactor was lined using plastic silage wrap. The top was covered with nylon geotextile mesh cloth.
Operating depth and active woodchip volume:	▪ The designated saturated operating depth was 1.10 m. ▪ Water level was monitored. However, it was checked during field visits in the inflow and outflow sumps in the bioreactor. ▪ The bioreactor inlet flows to a sump which distributes inflow water at two depths 10 and 50 cm from the bioreactor surface. ▪ The outlet height was designed as adjustable to operate at two heights, generally set at 10 cm below the bioreactor surface.

	▪ Inflows into a sump, which dispersion via a set punched AgPipe laid across the bioreactor width at two depths 10 and 50 cm from the bioreactor surface. ▪ Outflow collection by a set of perforated plastic pipes laid across the bioreactor width. The outflow collection pipes drain to a sump, which is connected to the height adjustable (two-stages) outflow.
Operational flow rates and bypass flow:	▪ The drain flow ranged from 0 (no flow) to 19 L s^{-1}, with an average of 3.90 L s^{-1}. ▪ Overall, the bioreactor captured about 32% of total drain volumes measured over the three years (July 2021 to July 2024). More than half of the weekly observed drain flow rates were fully (100%) captured. ▪ The bypass flow varied from 0 (no flow) to 17 L s^{-1}, with an average of 2.66 L s^{-1}. The bypass mainly occurred during winter (drainage) season and following occasional high rainfall events in spring and summer seasons. ▪ The bioreactor inflow ranged from 0 (no inflow) to 7.7 L s^{-1}, with an average of 1.2 L s^{-1}. ▪ The bioreactor inflow remained active mainly during the winter (drainage) and spring seasons.
Additional information:	▪ Since the bioreactor was constructed in the head of a small gully, channelised surface runoff posed a risk of scouring the woodchips during storm events. A nylon mesh was used to cover the woodchips to mitigate this risk. ▪ It is planted with native vegetation downstream in the gully, done as part of other on-farm riparian management. ▪ The bioreactor flows were managed to research the potential effects of different flow rates on nitrate removal efficiency. Since this was a research trial, the bioreactor performance was not optimised to achieve maximum nitrate removal (Singh et al. 2025).
* $ < 20K, $$ 20-40K, $$$40-80K, $$$$>100K	



Figure C-3: The Upper Manawatū-Whanganui woodchip bioreactor covered with nylow mesh (left) and during a farmer field day (right). Photos provided by Ranvir Singh, Massey University..

13.1.4 Lowcliffe, Canterbury

Location:	Waimanu Dairy Farm, Lowcliffe, Canterbury
River catchment:	Yeatmans Drain, Coastal Canterbury Plains
Year constructed:	Constructed October 2015
Bioreactor configuration:	Built on-line in a low spot at the edge of a paddock where the paddock meets a farm track. The tile drain outlet discharges to the drainage ditch opposite the farm track.
Bioreactor catchment area:	▪ Catchment land use is intensive, irrigated dairy farming with 3.5 cows per ha on a 160-ha dairy farm. ▪ The tile drain network is irregular; a 20-cm diameter clay tile drain buried ~1m deep intercepts shallow groundwater from a much larger catchment area. Hence, the tile outlet had year-round flow of 1-3 L s^{-1}.
Bioreactor size and woodchips:	▪ The bioreactor had a rectangular footprint of 12.5 m long x 2.2 m wide, corresponding to a length:width aspect ratio of ~5.7 and 27.5 m^2. ▪ Approximately 28 m^3 of untreated *Pinus radiata* woodchip were added; woodchips were dried to 30% moisture content and were on average 12-25 mm in diameter.
Project scope:	▪ This pilot-scale bioreactor was constructed along with two smaller woodchip denitrification 'walls' intercepting riparian groundwater seeps along a 1-km reach of a 2nd-order artificially drained waterway. The streambanks along the reach were re-battered and planted with native riparian plants as part of the Canterbury Waterway Rehabilitation Experiment (CAREX project). ▪ The waterways, tile outlet, and bioreactor were measured ~monthly using a before-after-control-impact study as part of a PhD project from January 2014 to October 2017 (18 months before the bioreactor and 24 months after the bioreactor). Monitoring encompassed manual flow gauging at the tile drain outlet and collection of filtered grab samples to measure concentrations of nitrate-N, dissolved reactive phosphorous, and dissolved organic carbon. ▪ The bioreactor design was a compromise between size, cost, and practicality, and it was tailored to align with the local context. The bioreactor was installed in collaboration with the farmer to avoid obstructing ditch maintenance or creating flood risks. It occupied minimal land area and was located to intercept a preferential flow path with nitrate-N >10 mg L^{-1}.
Approximate cost*:	• *$ < 10K (includes earthworks, pipes, woodchips, and geotextile cover)
Bioreactor performance:	▪ Over two years of operation, the bioreactor removed approximately 10% of the tile drain load (95% confidence interval: 5.9-14.2%). ▪ Performance estimates are conservative due to the possibility of untreated nitrate entering the bioreactor via groundwater and/or mixing with the treated water from the bioreactor outlet, since the outflow measurement point was after an additional 10 m of tile drain between the bioreactor outlet and the tile drain outlet at the edge of the waterway. ▪ Variable bioreactor performance observed was likely due to very short hydraulic retention times of ~ 1 hour, rather than cold temperatures or low influent nitrate concentrations. ▪ The average saturated volumetric removal rates was 50.9 (95% CI: 22.5-79.3) g nitrate-N m^{-3} d^{-1}. This is extremely high amongst New

	Zealand bioreactor studies and sits within the upper range of woodchip bioreactor studies (Cameron & Schipper, 2010; Hassanpour et al. 2017). This is likely due to the very small, but highly reactive volume of saturated woodchip that received very high nitrate–N loading. Most of the time, the majority of the woodchip were unsaturated and decomposing aerobically, which likely provided an additional carbon source in the saturated zone. ▪ No significant changes in the concentrations of dissolved reactive phosphorus and dissolved organic carbon were observed.
Fencing, liner, and covering:	▪ The bioreactor was fenced to exclude livestock and protect the scientific monitoring set–up. ▪ No liner was used due to the site having heavy soils with good water retention properties ▪ The woodchips were covered with geotextile cloth and a 30 cm topsoil cap with pasture grasses.
Operating depth and active woodchip volume:	▪ The inlet and outlet depth were the same elevation as the existing tile drain, which was approximately 1.2 m below ground surface. ▪ Two 15–cm plastic novaflow pipes were used to disperse the inflows across the width of the bioreactor; the same set–up was used to collect flows across the width of the bioreactor at the outlet. ▪ Water level was monitored during ~monthly sampling events at the inflow and at two other locations along the length of the bioreactor. The average saturated operating depth was ~0.5 m, corresponding to ~15 m^3 of wetted, active woodchip in the bioreactor.
Operational flow rates and bypass flow:	▪ Flows measured at the tile drain outlet were typically 2–3 L s^{-1}. ▪ Due to the flat topography, poorly drained soils, and shallow groundwater table, it was accepted that the bioreactor would likely impede drainage in the immediate vicinity of the bioreactor. This was observed during seldomly occurring high flow events in the first two years after installation. ▪ The high flow bypass conveyed negligible flow.
Additional information:	▪ This pilot study showed that pragmatically implementing bioreactors and accepting some uncertainties around their performance likely provides greater net environmental benefits than not implementing bioreactors in the first place. ▪ Lessons learned helped to inform design limitations for bioreactors in flat landscapes (such as the Canterbury Plains). The site served as a demonstration to the local irrigation board and catchment group to inspire further uptake of bioreactors in the region. ▪ Greenhouse gas fluxes measured during summer low flows from the tile drain bioreactor were similar to the surrounding pasture (CO_2–C mean: 185–286 mg C m^2 h^{-1}; N_2O–N mean: 49–90 µg N m^2 h^{-1}), suggesting no negative impacts from the bioreactor (Goeller et al. 2019).

* $ < 20K, $$ 20–40K, $$$40–80K, $$$$>100K



Figure C-4: The The Lowcliffe bioreactor during construction. Photo credit: Brandon Goeller, ESNZ.

13.1.5 Waituna Lagoon, Southland

Location:	Waituna, Southland (~40 km east of Invercargill). Privately owned dairy farm.
River catchment:	Waituna Creek catchment draining to Waituna Lagoon, part of the Awarua wetland complex (Ramsar Wetland of International Importance)
Year constructed:	Installed in 2015; commissioned and instrumented November 2015. Continuous monitoring through to early 2018.
Bioreactor configuration:	• In-line woodchip filter bed treating combined flow from two subsurface tile drains. Existing tile drains above the bioreactor site were re-graded to provide sufficient head to build bioreactor at ground level and still achieve a relatively high level of woodchip saturation. • Components: inlet flow measurement structure using perforated pipe inlet manifold to distribute flow across the bioreactor • No formal high-flow bypass was provided in the original design (a design lesson learned in the project).
Bioreactor catchment area:	▪ ~9.6 ha catchment with intensively grazed dairy pasture. ▪ Drainage type: subsurface tile drainage typical of Southland dairy systems (60–80 cm depth, 20–100 m spacing). ▪ Bioreactor footprint represents ~0.1% of the contributing catchment area; sized at ~10.4 m^3 woodchip per ha catchment.
Bioreactor size and woodchips:	▪ Rectangular bed with 1:1 length:width aspect ratio. ▪ Dimensions: 10 m long × 10 m wide × 1 m deep. ▪ Total bioreactor volume: 100 m^3; total woodchip volume: 96 m^3. ▪ Average saturated woodchip volume ~48 m^3 (range across years 30–70 m^3: 70 m^3 in 2016, 45 m^3 in 2017, 30 m^3 in 2018). ▪ Coarse *Pinus radiata* woodchips were used.
Project scope:	▪ Installed by NIWA, DairyNZ, and Dairy Green Ltd with co-funding from the Living Water Partnership (Fonterra and Department of Conservation) to quantify the mass load of N from a representative tile-drained dairy paddock and the load reduction achievable with a passive woodchip filter, in support of the Waituna Lagoon water quality strategy. ▪ The site was selected based on having adequate slope for gravity-fed flow, a defined catchment source area, soils and lithology susceptible to nitrogen leaching, and existing tile drainage that could be intercepted. ▪ Continuous monitoring (15+ months) included flow at inlet and outlet, water temperature, electrical conductivity, turbidity, rainfall, and flow-proportional grab samples (ISCO automatic samplers) analysed for ammoniacal-N, nitrate- and nitrite-N, total nitrogen, dissolved and total organic carbon, dissolved and total phosphorus. Loads were estimated using rating-curve and LOADEST modelling techniques to fill gaps between samples.
Approximate cost*:	• $$$ ($40–80K) — excavation of the 100 m^3 trench, woodchips, inlet/outlet manifolds and flow-control structures, fencing, labour. Costs exclude the research-grade monitoring equipment.
Bioreactor performance:	▪ Inflow nitrate-N (annual median): 2.19 mg L^{-1}. Outflow nitrate-N (annual median): 0.42 mg L^{-1}. Median daily inflow load: 47.1 g N d^{-1}. Median daily outflow load: 2.9 g N d^{-1}. Annualised inflow nitrate-N

	yield: 5.4 kg N ha^{-1} yr^{-1} (3–year mean; cf. 1.83 kg ha^{-1} yr^{-1} median for first 15 months). ▪ Annualised outflow nitrate–N yield: 0.11 kg ha^{-1} yr^{-1}. Median load–based nitrate–N reduction ~94%. Mean nitrate–N removal rate (volumetric) 2.4 g N m^{-3} active woodchip d^{-1} (3–year mean). ▪ Annual mean nitrate–N reduction (3–year mean) ~66% load reduction; efficacy exceeded 90% for ~70% of the monitoring period. Total nitrogen removal ~54% (3–year mean). ▪ Other contaminants: net ammonium production observed (mean annual −314% removal — i.e., increase across the bed), consistent with mineralisation of organic N within the saturated woodchip media. Sediment, phosphorus and microorganism removal not systematically reported. Subsurface drainage typically carries low sediment and phosphorus loads. ▪ Removal efficacy was strongly dependent on hydraulic retention time. Removal efficacy (% reduction) decreased markedly during high–flow events when HRT was short. Temperature was a secondary factor.
Fencing, liner, and covering:	▪ Fenced to exclude livestock and protect monitoring equipment. ▪ A high–quality EPDM (Ethylene Propylene Diene Monomer) synthetic rubber liner was used with careful sealing of all pipework to create a water–tight experimental system for research purposes. ▪ Covered with a geotextile cover over the woodchips.
Operating depth and active woodchip volume:	▪ Saturated operating depth ~0.7 m — the water level was maintained at this depth via outlet weir setting. ▪ Average active (wetted) woodchip volume ~48 m^3 (range across years 30–70 m^3). ▪ Water level was monitored continuously at the inlet and outlet.
Operational flow rates and bypass flow:	▪ Drainage flows were strongly seasonal, varying between ~0.1–10 L s^{-1}. Nine of 15 monitored months exceeded 100 mm rainfall. Discharge was a better indicator of soil moisture than rainfall alone. ▪ Bioreactor inflow and outflow were similar in all months (small differences attributable to measurement accuracy). ▪ ~355 days of flow per year (essentially year–round flow) due to the wet Southland climate. ▪ No formal bypass installed during this trial. High–flow events reduced HRT and treatment efficacy. Subsequent design recommendations include a buffer chamber, upstream flow throttle, or secondary bypass weir for future installations.
Additional information:	▪ A flagship New Zealand field–scale demonstration of a passive woodchip filter treating real subsurface drainage from an intensively grazed dairy farm. Provides some of the most comprehensive multi–year, continuous–flow monitoring data for any New Zealand bioreactor. ▪ Key insights: despite a relatively low volumetric removal rate, the bioreactor achieved very high load reduction efficacy (~94% median) because of long HRT under typical flow conditions. The lack of a high–flow bypass meant high–flow events caused short–circuiting and a temporary reduction in efficacy — a key design lesson. Net ammonium production was observed, consistent with international experience. Modelled scenarios in the Waituna catchment (Weeber et al. 2023) reported a median annualised cost–effectiveness of ~$44 NZD kg^{-1} N

	reduced for woodchip bioreactors — among the more cost-effective edge-of-field mitigations modelled. ▪ After 10 years operation, some localised flooding problems were observed due to accumulation of fine sands in the inflow distribution pipes. The woodchip media was still largely intact and proper operation was able to be resumed by removal of the accumulated sands. ▪ Key references: McKergow et al. (2015); Hudson et al. (2019); Weeber et al. (2023).
* $ < 20K, $$ 20-40K, $$$40-80K, $$$$>100K	

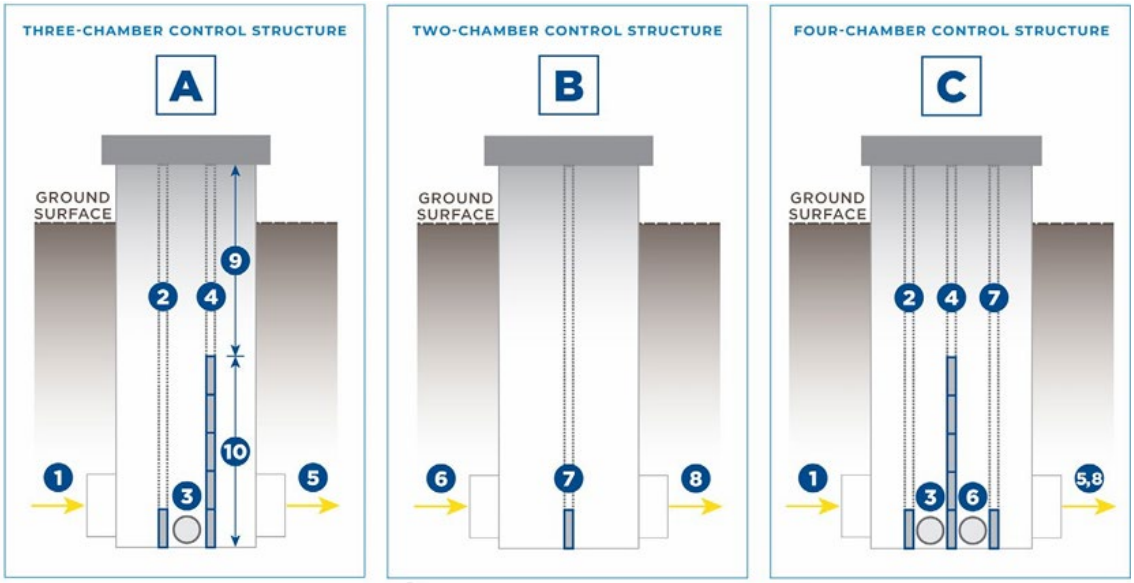


Figure C-5: A view of the Waituna Lagoon 'Nitrate catcher' bioreactor before coverage, showing the locations of the inlet (foreground) and outlet (background) and scientific equipment used to measure rainfall, water levels and flows, and automatically sample the inflows and outflows.
Photo credit: Chris Tanner, ESNZ.



Figure C-6: Living Waters farmer field-day at the Waituna Bioreactor in March 2015. The surface cover has been rolled back to expose the woodchips. Photo credit: Chris Tanner, ESNZ..

Appendix D Other examples of bioreactor water level control structures



1. **Drainage coming from the field.**
2. **Field-side control stoplogs:** To limit sediment entry into the saturated buffer or bioreactor, keep the bottom board installed and temporarily raise the stoplogs when doing work on the field drainage system.
3. **Inlet pipe to the saturated buffer or bioreactor.**
4. **Bypass stoplogs:** Used to set the control elevation for water into the saturated buffer or bioreactor. Anything over this level bypasses directly to the outlet.
5. **Bypass flow to the outlet.**
6. **Outflow from the bioreactor.**
7. **Bioreactor outlet stoplogs:** Used to control the speed of water through the bioreactor.
8. **Flow from the bioreactor to the outlet.**
9. **Distance from the top of the control structure to the top stoplog. One way that stoplog settings may be shown.**
10. **Height of the stoplogs. Another way that stoplog settings may be shown.**

Common control structures commonly for saturated buffers and bioreactors. Saturated buffers typically use three-chamber structures (A), but in some cases, two-chamber structures without the field-side control stoplog tracks (2) are used. For bioreactors with separate inlet and outlet structures, a three-chamber structure (A) is used for the inlet, and a two-chamber structure (B) is used for the outlet. Some bioreactors use a single four-chamber control structure (C) as a combined inlet and outlet structure.

Figure D–1: Schematic of control structures used in the United States of America. Reproduced from Maintaining the Edge (2025).
https://www.iasoybeans.com/PDFLibraryUploads/Manual_Edge%20of%20Field.pdf.

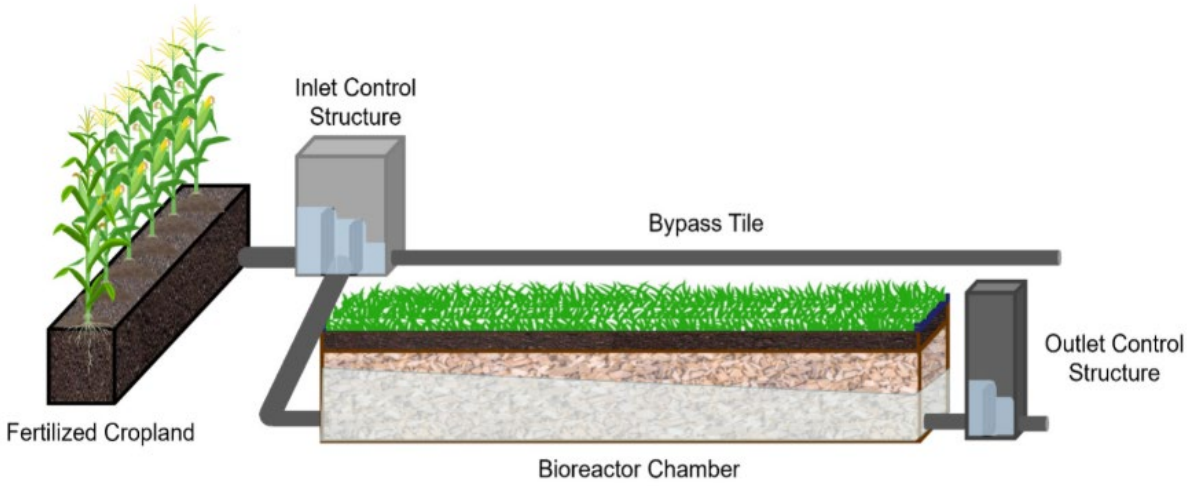


Figure D–2: Example of a US bioreactor showing on off–line woodchip bed with inlet and outlet control structures to regulate water levels in the bioreactor. Source: https://conservationdrainage.net/resources/conservation–drainage/practice–management/ .

Appendix E Advanced bioreactor design options for flat sites or pumped sites

Bioreactors at flat sites are only advisable if they are excavated deeper (deep-set bed, Figure E-1) or if they use some form of pumping (surface-pumped bed, Figure E-2). These represent less preferred options, as they do not rely on gravity-fed drainage. However, they remain plausible alternatives that could be adapted to specific site conditions, provided their associated caveats are clearly acknowledged and managed.

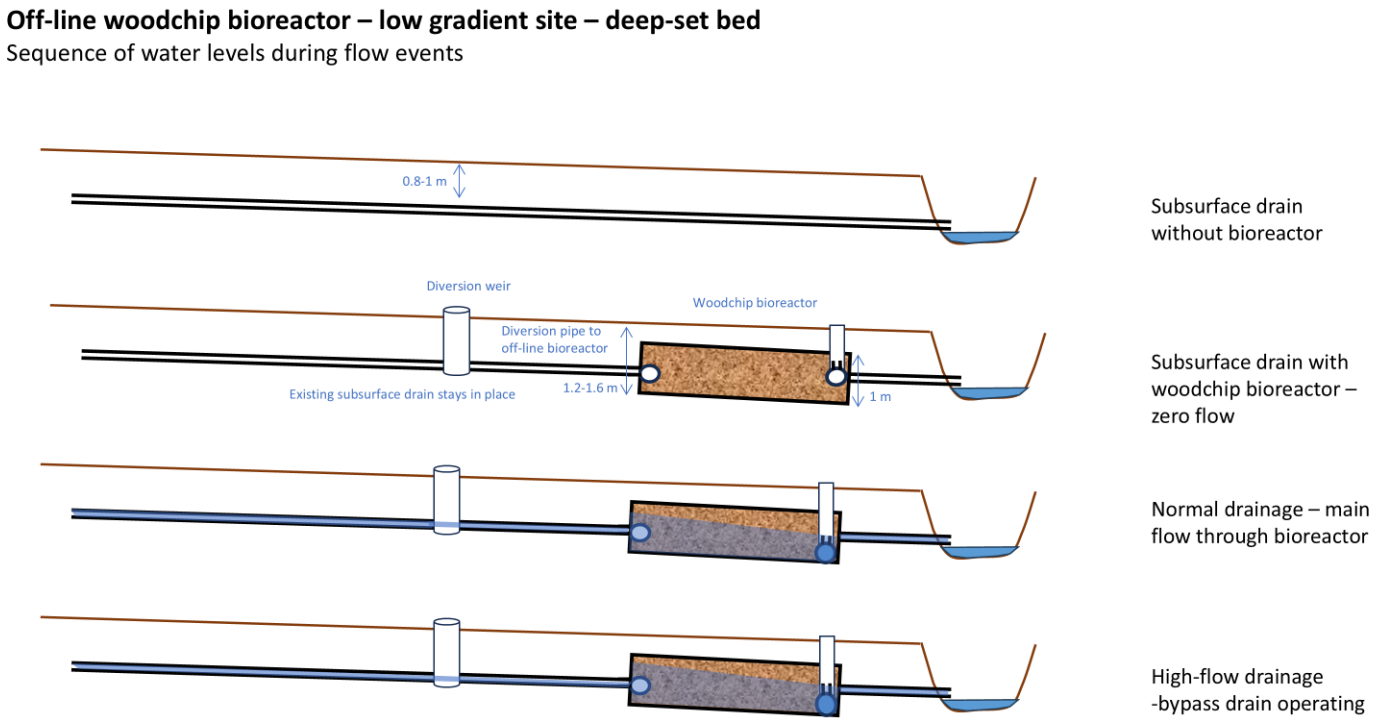


Figure E-1: Flat-site longitudinal cross section of a subsurface drain before/after woodchip bioreactor installation under no flow, normal flow, and high-flow conditions. To maintain drainage and gravity flow, the bioreactor must be installed deeper. Systems treating intermittent flows may not fully drain by gravity during dry periods, requiring seepage or pumping to prevent stagnation. This design may be unsuitable for high water tables or impermeable clay soils and has not yet been tested in New Zealand.

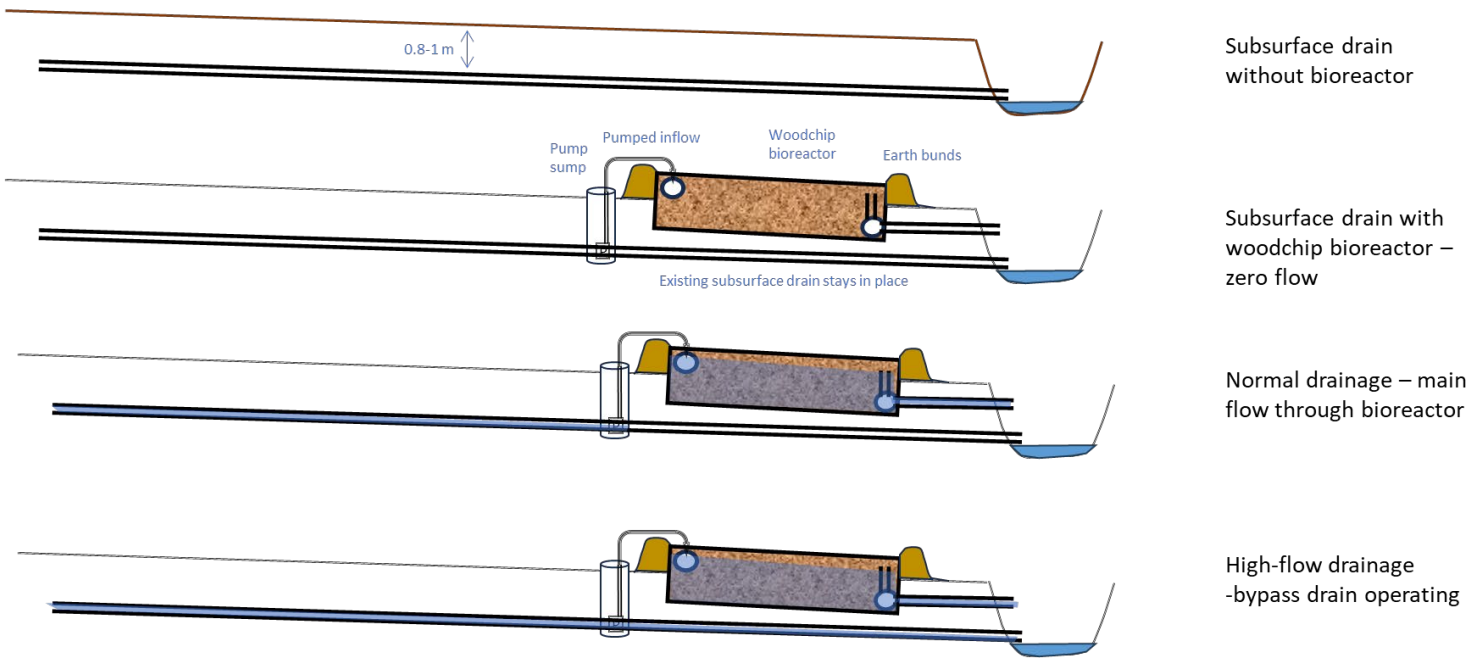


Figure E–2: Alternative flat site sequence from top to bottom showing a longitudinal cross section of a subsurface drain before and after installation of a woodchip bioreactor, and under no flow, normal flow, and high flow conditions Such a system would be reliant on a pump (possibly solar powered diurnally or battery assisted) to feed drainage into the bioreactor. It has the advantage that it could be constructed near the soil surface with minimal excavation to construct embankments, and it would be able to drain fully via gravity during periods without drainage flows. This kind of set–up could be applicable to many artificial drains on lowland plains, but it will have higher costs and logistical requirements. .