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Evaluating the Cost-Effectiveness of Green Infrastructure for Mitigating Diffuse Agricultural Contaminant Losses

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Abstract: New Zealand's agricultural sector faces the challenge of maintaining productivity while minimizing impacts on freshwaters. This study evaluates the cost-effectiveness of various green infrastructure systems designed to reduce diffuse agricultural sediment and nutrient loads. Utilizing a quantitative economic and contaminant reduction modeling approach, we analyze the impacts of five interceptive mitigation systems: riparian grass filter strips, constructed wetlands, woodchip bioreactors, filamentous algal nutrient scrubbers, and detainment bunds. Our approach incorporates Monte Carlo simulations to address uncertainties in costs and performance, integrating hydrological flow paths and contaminant transport dynamics. Mitigation systems are assessed individually and in combination, using a greedy cyclical coordinate descent algorithm to find the optimal combination and scale of a system for a particular landscape. Applying the model to a typical flat pastoral dairy farming landscape, no single system can effectively address all contaminants. However, strategic combinations can align with specific freshwater management goals. In our illustrative catchment, the mean cost to remove the full anthropogenic load is NZD 1195/ha for total nitrogen, NZD 168 for total phosphorus, and NZD 134 for suspended solids, but results will vary considerably for other landscapes. This study underscores the importance of tailored deployment of green infrastructure to enhance water quality and support sustainable agricultural practices.

Keywords: Nature Based Solutions; economic optimisation; edge-of-field mitigation; water quality; riparian buffers; constructed wetlands; algal turf scrubber; denitrifying bioreactor; detainment bunds

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1. Introduction

Agricultural producers in Aotearoa-New Zealand must increasingly align their practices with the country's freshwater policy frameworks to reduce inputs of nutrients, sediment, and fecal microbes into water bodies. The freshwater policy frameworks, including the 2020 National Policy Statement for Freshwater Management [1], provide guidelines and standards for water quality management. Diffuse runoff and drainage are the predominant and most challenging sources of water pollution to manage in New Zealand [2] and most developed countries [3]. Mitigations for diffuse pollution may be broadly categorized as preventative farm system mitigation practices (source control) or interceptive (edge-of-field or flow path) mitigation systems, and it is generally recognized that both types of mitigation may be required to achieve freshwater goals [4,5].

Green infrastructure involves the use of natural or semi-natural systems to provide ecological, economic, and social benefits. It is defined by McWilliam and Balzarova [6] as "networks of mostly vegetated areas outside of pastures and supporting croplands, that together enhance ecosystem health and resilience in the landscape". Green infrastructure plays a crucial role in our focus on interceptive solutions for reducing diffuse sediment and nutrient inputs to surface waters at various points on flow paths. This paper presents a

quantitative economic and contaminant reduction model for assessing five interceptive mitigation systems: riparian grass filter strips (RB), constructed wetlands (CW), woodchip bioreactors (WB), filamentous algal nutrient scrubbers (FANS), and detainment bunds (DB). The contaminants assessed are total nitrogen (TN), total phosphorus (TP), and total suspended sediment (TSS). Although not explicitly quantified here, these systems may also remove other contaminants (e.g., fecal microbes and pesticides) and offer additional environmental benefits, such as biodiversity enhancement, enhanced landscape esthetics, flood risk management, and net reduction in greenhouse gas emissions (Tanner et al. [7]). Our focus in the current assessment is on grazed pastoral agriculture, which is the predominant land use in New Zealand, but this approach is equally adaptable to other types of agricultural land use and other regions of the world.

There are significant bodies of research associated with these green infrastructure options. For example, the Constructed Wetland Practitioner Guidelines [8] and Riparian Buffer Design Guide [9] synthesize results from case studies in New Zealand and international literature. Some studies have developed and assessed bundles of farm management practices and edge-of-field mitigation systems [10,11]. These studies accounted for the impact of different farm systems but not different hydrologic landscape types. Other studies have compared a subset of mitigation systems for a specific catchment. For example, Weeber et al. [12] assessed the feasibility and cost-effectiveness of implementing CW, WB, and aluminized zeolite filters in the Waituna catchment. Tanner et al. [7] provided a semi-quantitative framework for selecting interceptive mitigation systems for different hydrological landscape types. They highlighted the relative benefits of each individual mitigation system type in terms of landscape fit, flow-path(s) intercepted, contaminant(s) targeted, efficacy, associated co-benefits, relative cost, operation and maintenance requirements, longevity, and consent requirements.

This study builds on the Tanner et al. [7] framework to include quantitative dimensions of costs and performance, utilizing a mix of case study data and lifecycle costing models. It is noteworthy that, to date, no study has conducted a direct comparison of the cost-effectiveness across such a broad spectrum of interceptive mitigation systems for a particular catchment. Additionally, prior studies have not fully considered the uncertainties associated with both costs and performance, often relying on average values. While such variability may become less significant over larger catchments, it can be a critical factor for individual landowners.

We have developed a model to simulate interceptive mitigation costs and performance for a specific New Zealand catchment and farming system type. We recognize that effective solutions often require the use of multiple tools in multiple locations [13], so our model allows for tool stacking and considers the impacts of contaminant removal on the efficacy of other tools further down the flow path. The purpose of the model is to (a) assist landowners in identifying the mitigation systems that may help them achieve their water quality targets and (b) help policymakers understand the potential aggregated costs and benefits of interceptive mitigation systems at a larger scale.

In Section 2, we describe each interceptive mitigation system and summarize cost data and the costing model used. In Section 3, we present the farm and hydrologic landscape typologies and explain how the hydrological flow paths are modeled for each contaminant. In Section 4, we introduce a hypothetical farming catchment and present model results showing the least-cost combination of green infrastructure to reduce TN, TP, and TSS.

2. Framework and Approach

Our model focuses on annual outcomes at the spatial scale of an individual farm or a small catchment area, predominantly agricultural in nature. The approach is best suited to an area that is relatively homogeneous in terms of slope, soil type, and nutrient and sediment losses. For catchments where this is not the case, the assessments we conduct provide a useful order of performance magnitude for farmers considering mitigation systems. However, it would be advisable to model different areas separately. To assess the

range of probable costs and performance of various mitigation systems, both individually and in combination, we employ a Monte Carlo simulation approach. This method allows us to simulate a wide array of scenarios, accounting for the uncertainties in cost and performance estimations.

For modeling the interception potential of these mitigations, we extend the framework proposed by Tanner et al. [7]. This extension involves incorporating quantitative estimates of the proportion of N, P, and sediment transported by each hydrological flow path, thereby enhancing the precision of our cost-effectiveness estimates for a particular landscape. Each modeled mitigation system reduces contaminants from a particular interception point, which in some cases reduces how much can be removed by a mitigation system further along and has implications for relative cost-effectiveness.

The simulation of costs for the mitigation systems is derived from a dual approach. Firstly, we examine a range of case studies that provide real-world data on the construction and maintenance costs of mitigation systems. Secondly, to address the gaps in data for certain systems and to add depth to our understanding of cost variations, we employ detailed costing models developed by agricultural engineers [14]. This hybrid approach helps provide a robust and realistic cost assessment.

To ensure a consistent comparison of costs, we employ the lifecycle costing method and convert all costs to an annualized equivalent with a 5 percent discount rate (the current New Zealand Treasury default rate[1]) and a 25-year assessment period. The expected effectiveness of each mitigation system, along with the range of uncertainty, is sourced from published literature specific to each system. All costs are in 2023 New Zealand dollars (NZD).

The input parameters for the model include catchment area, climate, land slope, drainage class, farm type, some basic fluvial characteristics, such as total lengths of permanent and impermanent waterways, and flow volume for any second- or third-order streams. To illustrate the model, we use a scenario with typical values for input parameters. Another important parameter is accounting for the scale of pre-existing mitigation systems, such as riparian buffers, which may already be in place. However, for simplicity, our illustrative scenario assumes there are none.

Standard farm contaminant losses less natural environmental baseline losses of TN, TP, and TSS are estimated using published values specific to the type of farm and hydrologic landscape under consideration, but these can be overridden if actual farm contaminant losses are known. Annual nutrient losses are assumed to be fixed in this scenario, but the model also allows for the simulation of varying contaminant losses if the degree of annual variability is known. This provides a reference point against which the effectiveness of mitigation systems can be measured.

For each type and combination of interceptive mitigation systems we simulate cost and performance for a range of implementation scales from zero to a recommended maximum scale. For example, 100% implementation for a constructed wetland is equal to 5% of the catchment, while 50% implementation is 2.5% of the catchment. Some mitigation systems have diminishing returns at larger scales of implementation, so the most cost-effective option can be the partial implementation of multiple systems.

To find the most cost-effective combination of mitigation systems at any level of mitigation required, we implement a greedy cyclical coordinate descent algorithm [15]. This algorithm iteratively evaluates the efficacy of incorporating a marginal amount of each mitigation and chooses the one offering the most significant reduction per unit cost, indicated by the steepest gradient. This relatively simple optimization approach is sufficient due to the convex nature of the objective function.

The model output includes summary statistics and confidence intervals for each type, combination, and scale of interceptive mitigation system employed. Results can, therefore, be used to estimate both the expected cost and the probability of achieving a particular target level of contaminant reduction.

3. Hydrologic Landscape Typologies (HLTs)

Our model builds on the framework outlined by USEPA [16] and adapted by Tanner et al. [7], based on four different archetypal hydrologic landscape types. The four types (Figure 1) are based on different combinations of permeable or impermeable soil and subsoil. HLTs with permeable soils (A) lose more nutrients to groundwater in dissolved forms, which makes these contaminants difficult to intercept with any mitigation system. HLTs with impermeable soil layers (B–D) result in flow being transported by hydrological flow paths that are more readily intercepted by one or more mitigation systems. However, several different types of mitigation systems may be needed in these situations to target the different surface and subsurface drainage system pathways in operation. For illustrative purposes, we focus on a scenario involving type D (low permeability topsoils and subsoils) in this paper because this is the only HLT where all five interceptive mitigation systems are potentially applicable.

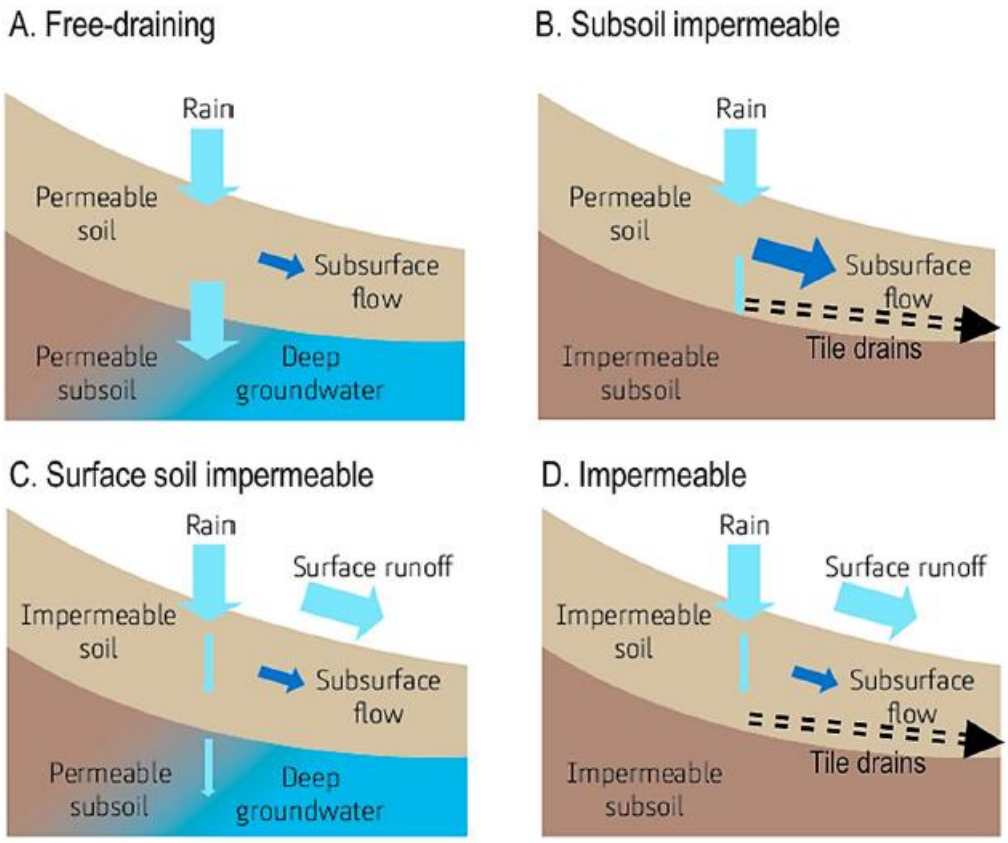


Figure 1. Diagram summarizing four basic hydrologic landscape types (Reprinted with permission from Tanner et al. [7]. Copyright 2023, N. Z. J. Agric. Res.

In order to quantitatively model the performance of different mitigation systems, we extend the semi-quantitative framework of Tanner et al. [7] with simplified numerical estimates of the proportion of TN, TP, and TSS losses that travel via six hydrological flow pathways: surface runoff, tile drainage, interflow/shallow groundwater, surface drains/ditches/first-order streams, second/third-order streams, and deep groundwater. The nutrient loss estimates for each flow path and HLT are presented in the Supplementary Information (Tables S2–S4), distinguishing between two slope classes: flat/undulating (<7°) and rolling/hilly (7–15°).

4. Farm Typologies

We draw on the work by Monaghan et al. [10], which categorized average contaminant losses from New Zealand farms by type (dairy or sheep and beef), climate, slope,

drainage, and wetness. We considered that HLT-D corresponds with "poor" drainage in the Monaghan classification. To illustrate the model, we use farm type D2, which is a dairy farm in a warm climate zone on flat, moist, poorly drained soil. D2 farms have the largest total area (166,727 ha) of any farm type corresponding to HLT-D. The average annual losses per hectare estimated for a D2 farm in 2015 are 29 kg TN, 1.4 kg TP, and 2.19 t TSS (Supplementary Information [17]). The average operating profit is NZD 2931/ha/year with an interquartile range of NZD 1744–6353.

5. Interceptive Mitigation System Costs and Performance

Our 25-year annualized lifecycle costing includes capital costs, maintenance costs, opportunity costs of lost grazing (if applicable), and replacement of assets that are not expected to last the entire period. All costs are simulated as random normal variables unless otherwise stated.

5.1. Natural Attenuation

The farm typologies provide estimates of TN, TP, and TSS losses from a sub-catchment. However, depending on factors such as climate, topography, soils, vegetation, and hydrologic characteristics, some of these losses will be attenuated by natural biogeochemical processes as they are transported through the catchment. In particular, significant attenuation of nutrients can occur in shallow, slow-flowing, low-order streams where the water column has close contact with bottom sediments [18].

Natural attenuation of TN predominantly occurs through nitrification and denitrification processes in both streambed sediments and adjacent wetland soils. These processes are more effective in smaller streams due to slower flow rates, allowing more extended contact time between the water column and reactive surfaces [18]. Phosphorus attenuation in stream beds primarily occurs through adsorption processes onto sediments and, to a lesser extent, by uptake through aquatic biota. This mechanism is especially effective in shallow, slow-flowing streams [19] and ditches [20] where there is extensive contact between water and sediments. We note that similar processes also occur in soils and groundwater. However, attenuation that occurs on land is already incorporated into the estimates of farm losses.

Our estimates, based on data from the Waikato River Basin in New Zealand, suggest that for permanently flowing waterways with a mean flow of 0.01 to 1 m^3/s, N removal is 0.223 ± 0.194 kg per km transported and P removal is 0.426 ± 0.148 kg/km [21]. Attenuation is much less for streams with a flow greater than 1 m^3/s, but most streams of Strahler order 1–3 are below this flow threshold. We model natural attenuation as a stochastic process based on the length of permanently flowing waterways in the catchment of interest.

For sediment contamination, it is important to distinguish that low-order streams generally act as net sources rather than sinks due to bank erosion [22]. The farm typology-based loss estimates described in the previous section already include stream bank erosion [23], so we do not need to model an additional process for bank erosion. Although not accounted for in the present study, in practice, buffering storm flows by interceptive mitigations could reduce the incidence of streambank erosion [24].

5.2. Riparian Buffers

A riparian buffer (RB) describes a band or strip of vegetated land (grass, trees, or shrubs) established and managed as a buffer between land and water [25]. RBs may be managed for a range of functions or outcomes, including economic return [26], cultural values, habitat, biodiversity, esthetics, storm flow moderation, and, importantly, reducing contaminant loads delivered to waterways from pasture.

RBs may be divided into three classes: grass-filter riparian buffers (GRBs), focused on intercepting and filtering ephemeral runoff; planted riparian buffers (PRBs), comprised of a mix of deeper-rooted trees and shrubs that can intercept shallow subsurface flows and provide shade; and multi-function riparian buffers (MRBs), combining a GRB and a

PRB. Costs are relatively simple to estimate for each type of buffer; however, there is no strong relationship between PRB scale and contaminant reductions [25]. We only include GRBs in this quantitative analysis, because they have greater applicability for the example landscape and farming system we use to illustrate the model.

5.2.1. Performance

GRBs function by slowing surface runoff, allowing nutrients (TP, TN) and sediments to settle and infiltrate the soil [9]. The filter performs well if runoff enters along the entire edge of the buffer at shallow depth and velocity. Fast flow or flow convergence into channels makes the buffer less effective, so to be most effective, the slope should be low to gently rolling (<11°). There is more uncertainty regarding RB performance in catchments with high clay content soils (>28.5% clay). Such soils are expected to generate finer suspended sediments with poorer settling characteristics and have higher rates of surface runoff and lower infiltration rates. We use performance estimates from the Riparian Buffer Design Guidelines [9], which provide performance confidence intervals based on site suitability, infiltration performance, and filter width relative to the contributing hillslope length (Figure 2).

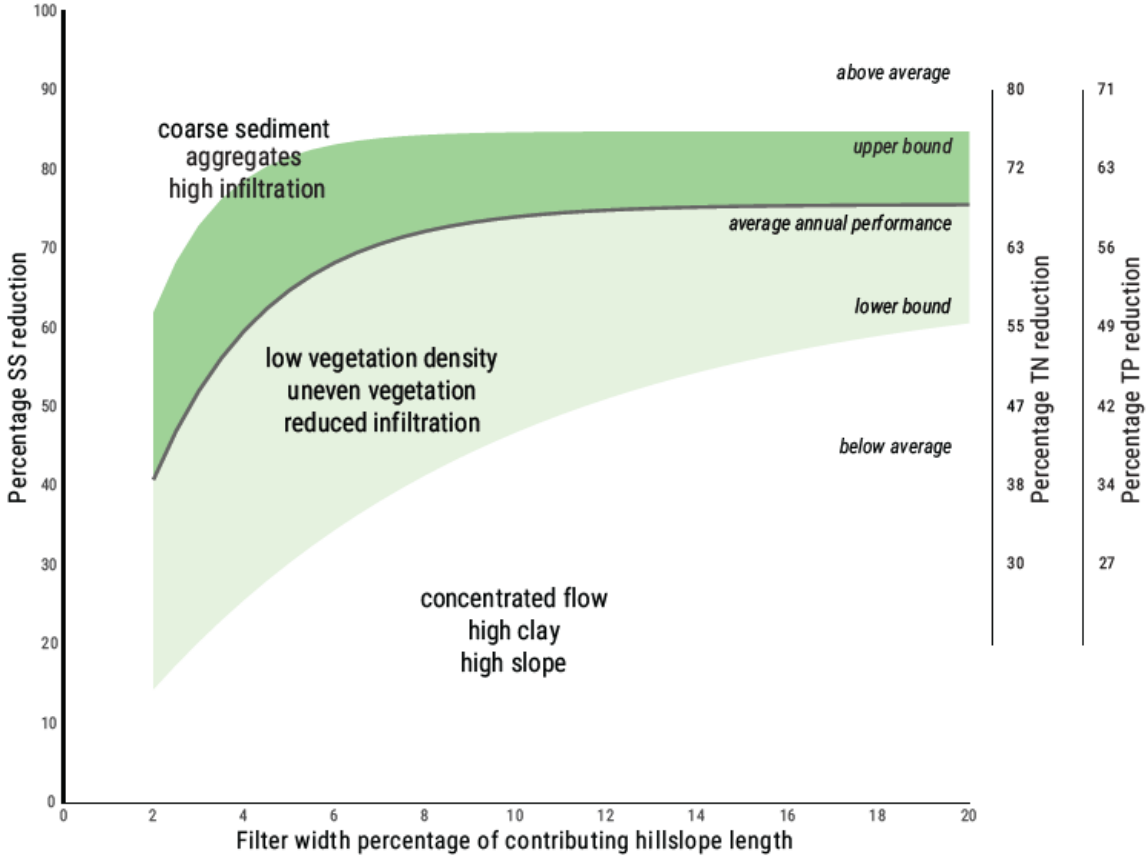


Figure 2. Long-term median annual reduction and 95% confidence interval for TN, TP, and TSS from riparian filter strips. Shaded areas represent 95% confidence intervals. Reprinted with permission from McKergow et al. [9]. Copyright year 2022, NIWA.

In our model, we have included parameters for landscape slope category (flat or rolling), hillslope length, and Boolean parameters for site suitability and infiltration performance. We simulate performance by drawing a random normal variable from the appropriate shaded area where the mean is the mid-point and the standard deviation is such that the boundaries represent 95% confidence. If the site is unsuitable due to clay or slope, performance is simulated by a random normal draw from the lower white-shaded

area. If the site is suitable but infiltration is less than ideal, then the draw is from the light green area. If infiltration is high (good vegetation coverage and coarse soil), then the draw is from the dark green area.

To achieve the theoretical maximum contaminant reduction across the catchment, GRBs must capture surface runoff from agricultural land before it enters any permanent or intermittent stream, and the width should be 20% of the hillslope length for each stream on both sides. The minimum width is 3 m as per legal requirements [27][2]. When we scale GRB implementation in the model we proportionally increase both GRB width and hillslope length up to this maximum. In a real catchment, 100% implementation may not be achievable due to excessive slope or the presence of farm infrastructure. On average, there are 15.5 m of permanent stream length per catchment hectare in New Zealand [28]. However, the total length, including intermittent and ephemeral streams is larger—by four times in the case of Auckland region [29]. The cost of GRB interception, therefore, varies greatly by landscape.

5.2.2. Costs

The cost of installing GRBs comprises fencing, maintenance (including weed control), and the opportunity cost of lost grazing. Fencing cost averages and ranges are sourced from the stock exclusion costs report [30] and adjusted for inflation using the Producer's Price Index for farming published by Statistics New Zealand. Fence maintenance is assumed to be 1% of average capital (Supplementary Information, Table S1). The expected lifetime of the fence is 25 years.

The opportunity cost of lost grazing is assumed to be 50% of the operating profit for the relevant farm type (see typologies in the next section). This is because riparian land is assumed to be less productive than the farm average due to erosion and saturation, and 50% is a typical assumption used in stock exclusion analyses [11,30]. We also assume a cost of NZD 500/ha/year for weed control of the buffer area. Based on these data, the average annualized cost for applying GRB varies from NZD 1.10–3.05/m (2023), depending on buffer width, slope, and farm type.

5.3. Constructed Wetlands

Constructed wetlands are artificial wetlands designed to intercept farm runoff and drainage and typically include extensive shallow areas ($\leq$0.5 m deep) vegetated with emergent aquatic plants [8]. They are particularly efficient in removing nitrate, principally via microbial denitrification, and reducing sediment and particulate phosphorus loads through settling [31]. They can be implemented at various scales, from headwaters to paddock to farm, and at the bottom of catchments, intercepting water from multiple properties before water enters lakes and estuaries. Constructed wetlands offer ancillary benefits, including habitat and biodiversity enhancement, storm-flow buffering, carbon sequestration, landscape esthetics, and cultural benefits (e.g., provision of traditional weaving materials) [7].

5.3.1. Performance

Woodward et al. [32] reviewed the contaminant reduction effectiveness of constructed wetlands of varying sizes in relation to the size of their respective contributing catchments. This assessment was based on data gathered from both local and international studies, which included field-scale monitoring and modeling. Nutrient and sediment removal rates were dependent on the wetland area percentage of the contributing catchment area. Reductions increased only marginally above a 5 percent area, so 5 percent is assumed to be the maximum size that would be implemented. In addition, TN reductions depend on whether the wetland is in a "warm" zone (most of the North Island except for the central plateau) or "cool" zone, which includes almost all of the South Island (Figure 3).

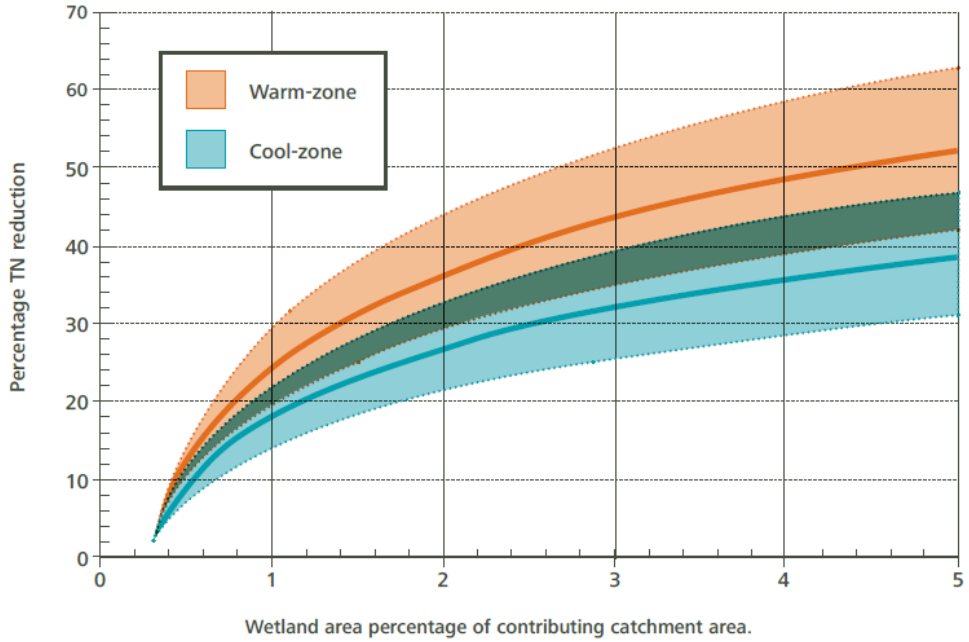


Figure 3. Long-term median annual total nitrogen (TN) reduction performance expectations. Reprinted with permission from Tanner et al. [8]. Copyright 2022, NIWA.

We assume a scenario in which wetlands are used to intercept contaminants transported in surface drains, ditches, and first-order streams. This may require more than one wetland to achieve total interception and water treatment, with the total area being up to 5 percent of the contributing agricultural catchment. In some natural first-order streams, it may not be possible to use CWs (or there may be additional costs) due to the need to protect fish passage. We set the minimum size as 0.5 per cent of the catchment. CWs that are too small struggle to treat high-flow events that tend to have the largest contaminant concentrations and may be scoured out and damaged.

We model performance by drawing a random normal variable from the TN, TP, and TSS reduction curves in Tanner et al. [8]. The shaded areas are considered to represent a 90% confidence interval considering both inter-annual and inter-site ranges of performance.

5.3.2. Costs

Wetland costs were sourced from project managers of eleven New Zealand case studies ranging from 0.19 ha to 6 ha in size, several of which are described in the Constructed Wetland Practitioner Guide [8]. We group up-front costs into four categories: consenting, planning and design, implementation, and "other", which includes monitoring and other non-standard costs (Figure 4). Consent costs were available for only two sites and were not a large proportion of the total cost, so we excluded these from total cost estimates. We also exclude "other" costs because monitoring is not necessary for typical use. The average capital cost for planning, design, and implementation was NZD 203,000 per hectare in 2023. The distribution is strongly positively skewed, so we use a lognormal distribution to simulate capital cost.

Constructed wetland costs depend on several factors, including size, geological conditions, and site-specific considerations. There was no detectable systematic relationship between wetland size and cost per hectare (i.e., no discernible economies of scale in costs from increasing wetland size within the case study size range). Much of the cost variation can be attributed to differences in site characteristics and remoteness (Brian Ellwood and Henry Van der Vossen, Lowe Environmental Impact Ltd., Christchurch New Zealand pers. comm. November 2023). In addition, some CWs formed part of research programs with specialized design and monitoring requirements. In our model, based on considerations in a detailed costing model [14], we include a parameter for slope class, which increases the

mean cost by 10% when changing from flat (0°) to low slope (1–2°), another 10% from low to moderate (2–5°), and another 10% for slopes above 5 degrees. There is a second parameter for remoteness level, which adds 10% to the cost if the site is more than 45 min from a city and another 10% if it is more than 1.5 h. Remoteness increases travel and delivery costs for labor and materials. A flat and accessible site, therefore, has an expected cost of NZD 126,000/ha with a 90% confidence interval of NZD 100,000–160,000/ha. Conversely, a steep and remote site has an expected cost of NZD 197,000/ha with a 90% confidence interval of NZD 156,000–248,000.

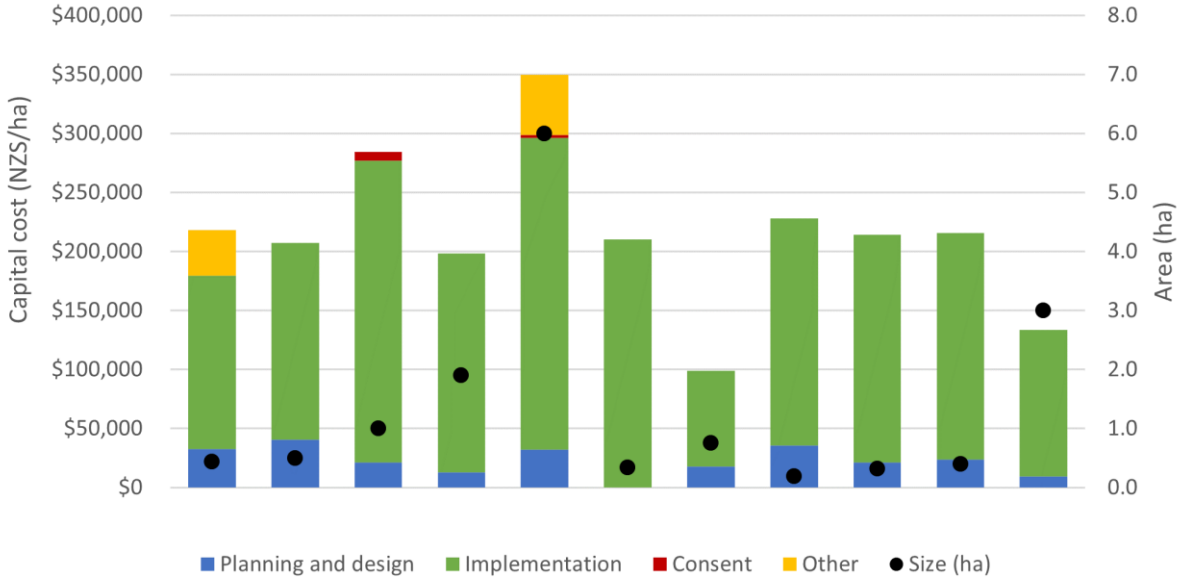


Figure 4. Actual constructed wetland capital costs by anonymized site.

Annual maintenance and operational costs comprise fence maintenance and weed and pest control. Weed control costs decrease after plants become established, but the annualized average is NZD 4040 ± NZD 810/ha/year.

5.4. Detainment Bunds

Detainment bunds are a type of sediment trap involving low dams (<3 m) strategically positioned across ephemeral flow paths to accumulate runoff. This temporary pooling process aids in capturing sediment, particulate phosphorus, and bacteria [33,34]. The accumulated water either seeps into the ground in areas with permeable soil or is slowly discharged through a restricted outlet or manually released after a few days. This design ensures that the land used for ponding can still support pasture and regular agricultural activities when not ponded. When flow exceeds the pond's capacity, water overflows through a riser pipe or, in extreme situations, over a spillway that safely directs excess water downhill below the bund.

The recommended sizing for a detainment bund is a ponded volume of 120 m^3 per hectare of catchment area draining to the bund [35]. However, detainment continues to increase at larger volumes because more extreme flow events can be accommodated. In our model, the size can be adjusted depending on the scale of removal needed. The system requires the use of a natural valley; otherwise, a large storage volume would be difficult to achieve with low earth dams. We assume a realistic implementation range of 90 to 180 m^3/ha for rolling landscapes. Completely flat landscapes are unlikely to accommodate a DB [36], but a mostly flat catchment may have areas of suitably undulating terrain. We have assumed for this scenario that 50% of the farm was gently undulating, and DBs were applicable to 50% of this area (i.e., 25% overall).

5.4.1. Performance

Levine [37] reported that DBs with a volume ratio of 120 m^3/ha mitigated an estimated 51–59% of the annual suspended sediment loads and 47–68% of the annual TP loads delivered to the DB. Smith and Muirhead [38] reviewed the performance of a wider range of DBs and found a power curve relationship between volume and performance. We adjusted the curve to include only the New Zealand pastoral farming data points and made it through (0, 0) by changing it to a symmetric sigmoidal function (Figure 5). Smith and Muirhead [38] show a similar sediment reduction as Levine [33] at a volume of 120 m^3/ha, but they do not report impacts on TN or TP. Therefore, we use the same shape curves for TP going through the midpoints of the Levine [37] results. Although TN reductions in surface runoff flows have also been reported for DBs [37], we have not included this in our assessments. The major pathway for N loss from grazed pastures is generally leaching of nitrate-N, and the forms of N infiltrated and settled from surface runoff in DB ponded areas are also likely to be subsequently transformed and significant quantities lost to groundwater via these processes (i.e., ultimately not removed).

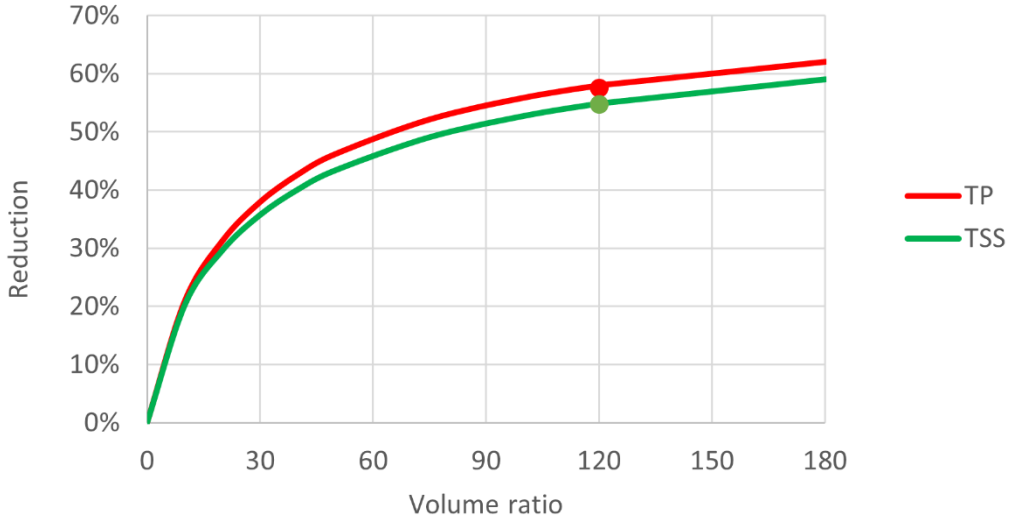


Figure 5. DB mean expected removal performance by volume ratio.

For uncertainty bounds we used a coefficient of variation of 0.14 from the pastoral data points in Smith and Muirhead [38].

5.4.2. Cost

We obtained costings from four detainment bund case studies in different areas of New Zealand. Sizes range from 900 to 1800 m^3 of ponded volume. As with the constructed wetland case studies, we included only planning, design, and capital costs. The average inflation-adjusted cost of a detainment bund was NZD 21 $\pm$ 8.50/m^3 of the ponded volume created. Operation and maintenance costs are assumed to be essentially zero. Likewise, there is no opportunity cost of lost grazing because the water is only held for three days, which should have minimal impact on pastoral production [39].

5.5. Woodchip Bioreactors

Intercepting nitrogen with a woodchip bioreactor (WB) involves redirecting drainage water containing nitrates through a porous subsurface bed of high-carbon material, most commonly woodchips. This setup fosters an environment rich in carbon but low in oxygen, promoting the microbial transformation of nitrate into nitrogen gas via denitrification [40]. We assume a scenario where WBs are used to intercept water from tile drains because it requires flow convergence into a relatively small area.

5.5.1. Performance

The mean TN removal rate from the treated flow is modeled as a random normal draw with a mean of 40% and a standard deviation of 26%, as per Christianson et al. [41]. There is no impact on TP or TSS. The treatable load is only what is transported through tile drains. An optimal sizing is assumed to be 11 m^3 of woodchip media per contributing ha of catchment (tile drained) area, consistent with the case study site in Dougherty et al. [42] and the Pirie farm site in Southland [43]. For sizes below optimal, performance is reduced along a similar-shaped curve to DBs and CWs.

5.5.2. Cost

New Zealand costings were only available for three case study sites where the WBs ranged in size from 78 to 430 m^3. The inflation-adjusted capital cost (excluding consent and "other" costs) ranged from NZD 289 to 532/m^3. Due to the small number of case studies and high cost variability, we augmented the case study data with a detailed costing model developed based on recent agricultural construction contracts in the Canterbury region of New Zealand [14]. In this model, a "low" expected cost is NZD 525/m^3, representing the 5th percentile for a flat and accessible site. Slope and remoteness shift the expected cost upwards. The 95th percentile cost for a steep, remote site is NZD 728/m^3. The minimum size for a WB is set at 25 m^3. To convert WB volumes to areas, we assumed the woodchips in the beds would be 1 m deep.

We assumed that there is no maintenance required for a bioreactor. The land area above woodchip bioreactors is generally backfilled with soil and could conceivably be grazed as normal. However, to avoid compaction of the bed, these areas are not trafficable and may experience slumping over time, so we assume that these areas are taken out of active production and, therefore, incur an opportunity cost. The expected lifespan of the woodchip media has been estimated as 10 years [41], which is the interval we have assumed that the media will need replacement. Replacing the media also requires excavation and liner replacement and costs approximately 1/3 the cost of the initial installation, assuming the liner is not damaged during the woodchip replacement process. For a 25-year assessment period, the discounted and annualized cost of woodchip replacements is NZD 21/m^3, plus a multiplier for a steep or remote site.

5.6. Filamentous Algal Nutrient Scrubbers (FANS)

FANS are a relatively new approach for reducing contaminant inputs to waterbodies from agricultural pollution [44]. These systems utilize the natural growth processes of filamentous algae to reduce nutrients from agricultural waters that are passed through the system. The algae growing in the FANS system are periodically harvested and repurposed as biofertilizers or feed supplements for livestock.

5.6.1. Performance

Filamentous algae can be efficient at nutrient uptake [45,46] but need consistent flows and the right balance of N and P for optimal growth. We have assumed that FANS are only applicable for nutrient removal from an appropriately sized system. The algae will absorb N and P at a ratio of 10:1 by mass [45] until one or the other nutrient is depleted. At a catchment level, the critical factor determining performance is, therefore, what proportion of flow can be taken from a second- or third-order stream for treatment. FANS treat only that proportion of the flow and contaminants transported by that flow for the catchment upstream of that point in the stream network. In this study we assume a scenario where FANS treat water from a second-order permanently flowing stream via pumping.

5.6.2. Costs

Due to a lack of field-scale applications, the costs for implementing FANS were derived from a detailed costing model developed for this study [14]. The base size for this costing is a 4-unit system, one hectare in overall size (1.2 ha including embankments). With a

hydraulic loading time of 3 days, the system can treat a flow of 2000 m^3/day. The system needs a permanent flow to maintain operation, so we assume the system is sized to treat 75% of the mean annual low flow (MALF) of the stream being treated. Flow data are sourced from NZ River Maps [47]. During seasonal low flow periods, particularly in drought periods, this may not be a realistic quantity to extract, even temporarily. However, N export is likely to be low under these conditions, and we assume for illustrative purposes that inflows would be restricted under these conditions but with FANS still be able to achieve equivalent nutrient removal rates due to higher summer growth rates (elevated day lengths and temperatures). All costs are scaled proportionally. For example, if the MALF is 3000 m^3/day then size and cost are scaled 150% compared to the 2000 m^3/d base unit.

We simulate the mean daily flow to estimate how much of the flow is treated for a system sized for 75% of MALF. Our model uses input parameters for the characteristics of the second-order stream (if there is one) flowing through the catchment of interest. These parameters are daily flow mean and standard deviation, MALF, and nutrient concentrations. A spring-fed stream with low flow variation is more cost-effective because a higher proportion of total flow can be treated in a fixed-size system. Flow variability is the main reason why we assume FANS will be placed in second- or greater-order streams.

Capital costs include design, excavation, liners, pipework, pumps, supports, and harvesting equipment. A "low" capital cost of NZD 284,000/ha is assumed to be the 5th percentile of the range in which actual costs may fall. Expected cost increases with slope class and remoteness. There is also an extra cost multiplier if the site is more than 20 m from the waterway. A base system on a non-ideal site has an expected cost of NZD 393,000/ha. We assume the 95th percentile is the highest quoted cost plus a 20% contingency. The minimum size in our cost model is 0.25 ha.

Harvest is by quadbike using a vacuum head, hose, filter, and pump (included in the capital costs). The harvest volume is calculated by working backward from the amount of nutrients entering the system. We assume algae is 8% N and 0.8% P by dry weight [44], and dry weight is 6.7% of wet weight; i.e., 500 kg N and 50 kg P would yield around 100 tonnes of wet algae per year. Operating costs for harvest include labor, fuel, and maintenance and are estimated to be NZD 19 per ton of wet biomass. We assume a moderated 20% coefficient of variation for operating costs. There is also an opportunity cost of lost grazing for the area occupied by the FANS, which is half the average annual operating profit per hectare of the farm.

We assume the harvested algae will be spread for fertilizer, in a manner similar to land application of effluent and using existing effluent spreading equipment. We use fertilizer prices to determine the value of the harvest and subtract it from annual costs. Urea cost NZD 809 plus NZD 230 freight per ton in 2023[3]. Urea is 46% N so this is equivalent to NZD 2.26/kg N. Superphosphate was NZD 449 plus NZD 230 freight per ton. Superphosphate is 9% P, so the value is NZD 7.54/kg P. At these prices, the nutrient value of the harvest is approximately two-thirds of the cost of harvest. However, the cost of harvest is almost entirely labor, so recycling nutrients using owner labor could help reduce farm cash outflows in difficult economic times.

6. Illustrative Scenario

To illustrate the cost-effectiveness of the different mitigation systems using our model, we use a hypothetical typical D2 dairy farm in the Waikato Region. This farm has a catchment area of 67 hectares. It is not remote, has no existing edge-of-field mitigation systems, and has a total length of 2200 m of waterways, including intermittent. Despite being a mostly flat landscape, we assume a quarter of the catchment has sufficient undulating terrain to accommodate DBs.

Because the soils have low permeability, we assume that the farm has extensive tile drainage[4], and 70% of the N and 20% of the P is routed by this pathway and the remainder by surface runoff. Because D2 farms have poor drainage, we assume riparian margins have a high clay content (>28.5%), so GRB performance is in the lower zone [9]. The average

hillslope length[5] is assumed to be 150 m. The catchment also features a typical second-order stream with a mean daily flow of 9500 m^3, an annual standard deviation of 2000 m^3, and a minimum annual low flow (MALF) of 2200 m^3/day. The TN and TP concentrations in this stream are 1.72 and 0.07 g/m^3, respectively, which is in the 95th percentile estimated for all second-order streams in New Zealand[6].

Because the landscape is HLT-D, there are several mitigation systems suitable for the treatment of each contaminant [7]. The diagrams in Figure 6 show the average quantities of TN, TP, and TSS available for interception at each point and by each mitigation system. Interception of surface runoff and tile drainage reduces the amount of contaminant in subsequent first-order streams and surface drains. When both a GRB and DB are implemented, we assume the DB will treat what would have otherwise flowed into a GRB, so DB removals are subtracted before calculating GRB removals[7].

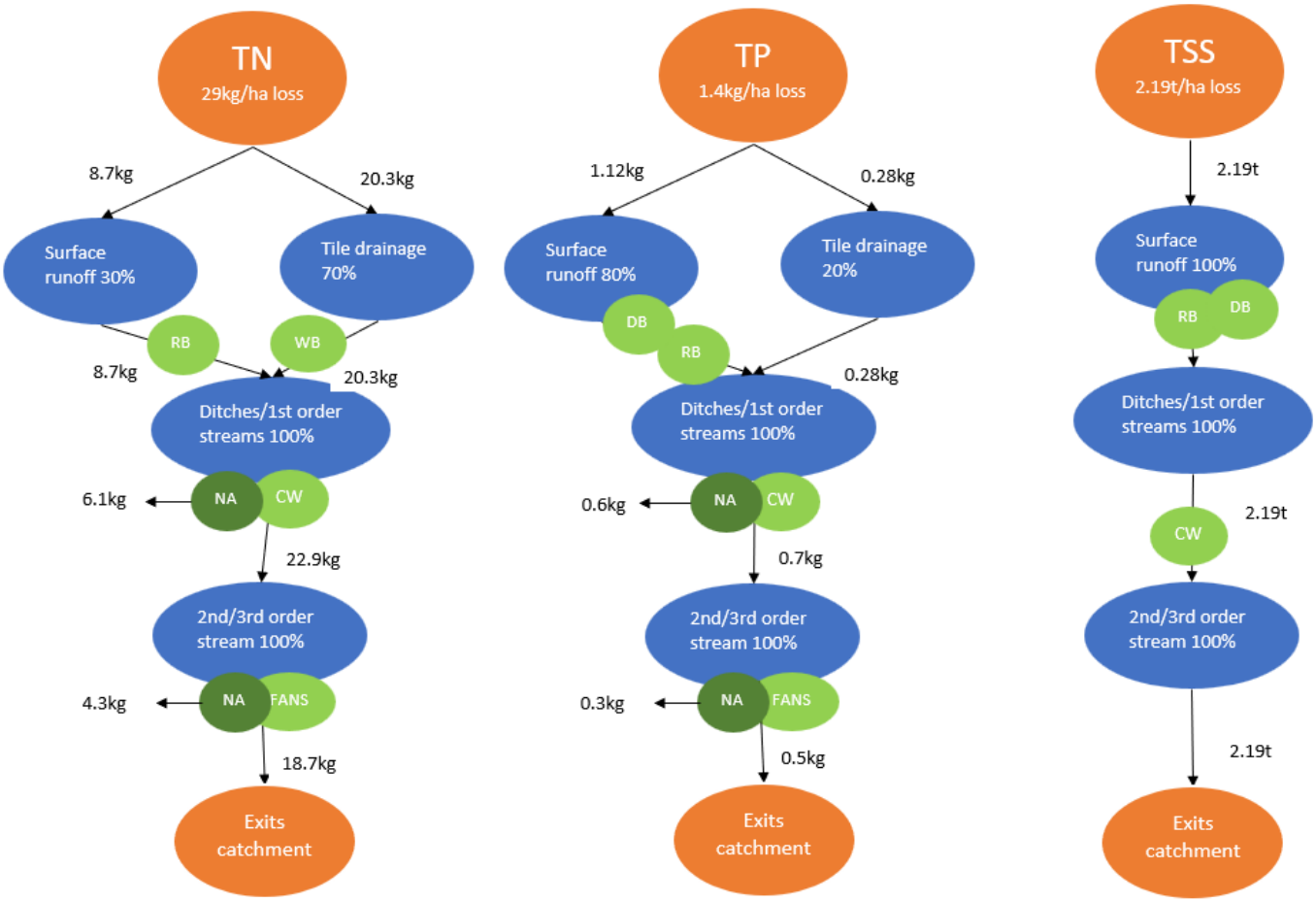


Figure 6. Contaminant loads (orange), flow paths (blue) and estimated proportional loads for TN, TP, and TSS for landscape type Flat D and farm type D2 with potential interception points for relevant interceptive mitigation systems (green) and natural attenuation (NA) processes (dark green). RB = riparian grass filter strips, CW = constructed wetlands, WB = woodchip bioreactors, FANS = filamentous algal nutrient scrubbers, and DB = detainment bunds.

A reasonable maximum mitigation target for this hypothetical farm might consider non-anthropogenic loads as a baseline. Natural baseline loads of TN and TP in areas of the Waikato corresponding to our HLT4, D2 example have been estimated [49] to be 3.4–3.7 and 0.21–0.34 kg/ha/year, respectively (Table 1). An equivalent baseline for TSS is 1.4 t/ha/year [50]. Given our farm losses and stream lengths, natural attenuation in stream beds is assumed to remove 10.25 kg/ha/year TN and 0.86 kg/ha/year TP. The remaining anthropogenic loads are 15.2 kg/ha/year TN, 0.27 kg/ha/year TP, and 0.8 t/ha/year TSS.

Removing these anthropogenic load amounts would bring the catchment load back to "natural" levels and is the maximum level of removal that we assess. The specific catchment target set may be less ambitious and is outside the scope of this paper.

Table 1. Scenario farm losses, natural baseline load and attenuation losses, and anthropogenic load.

Contaminant	Farm Losses	Natural Load	Natural Attenuation	Anthropogenic Load
TN (kg/ha)	29.00	3.55	10.25	15.20
TP (kg/ha)	1.40	0.28	0.86	0.27
TSS (t/ha)	2.20	1.40	0	0.80

6.1. Results

The model was first run for each mitigation system individually, with 1000 simulations of cost and efficacy distributions at each level of implementation from zero to the defined reasonable maximum scale. Implementation size was discretized into 100 steps, an acceptable trade-off between curve smoothness and simulation time. Subsequently, the next step was to use the optimization algorithm to find the most cost-effective combination of systems for any given level of reduction. With 5 systems, each having 100 size steps, there are 10 billion possible combinations of systems and sizes. However, the greedy cyclical coordinate descent algorithm was able to find the optimal solution within a thousand steps in most cases. After finding the optimal solution using mean values, we again ran 1000 simulations to create confidence intervals.

6.1.1. Mitigation of TN

Figure 7 shows the most cost-effective mix of systems across the range of potential N reductions, with the *x*-axis starting at the amount that would be removed by natural attenuation alone. For low reduction targets, we find that using WBs alone is the most effective. When a WB reaches the maximum size we have assumed, CWs are added. A CW is more expensive than a WB, but it treats the same flow as a WB plus surface runoff as well, so it becomes more cost-effective for a moderate level of reduction. For higher reductions, it is more cost-effective to use GRBs, WBs, and smaller amounts of CWs and FANS. It is theoretically possible to completely remove the anthropogenic load with WBs, GRBs, and CWs at close to 100% of maximum size and FANS at 51%. The following Figure 8 shows the relative contribution of each system to total removals. When mitigations remove TN before it reaches streams, the subsequent quantity removed by in-stream natural attenuation reduces.

The relative cost-effectiveness of individual options and bundles of mitigation systems in mitigating TN is displayed in Figure 9. Accounting for natural attenuation, which averages 10.3 kg/ha N for this scenario, Figure 9 only shows costs incurred from this point. The individual curves show that single-mitigation systems have limited efficacy, but the combined mitigations can achieve much larger reductions. Removal of the anthropogenic load could be achieved, on average, for an expected mean cost of NZD 1200/ha. This would be a relatively large proportion of farm operating profit for the least profitable quartile of farms of this type (NZD 1744/ha). The confidence interval for cost is NZD 1020 to NZD 1320 (dark grey area). The uncertainty around mitigation performance (light grey area) should be read horizontally and shows that performance uncertainty is larger than cost uncertainty. The mitigation bundle with mean removal of 25.45 kg/ha has a 90% confidence interval of 21–28.5 kg/ha.

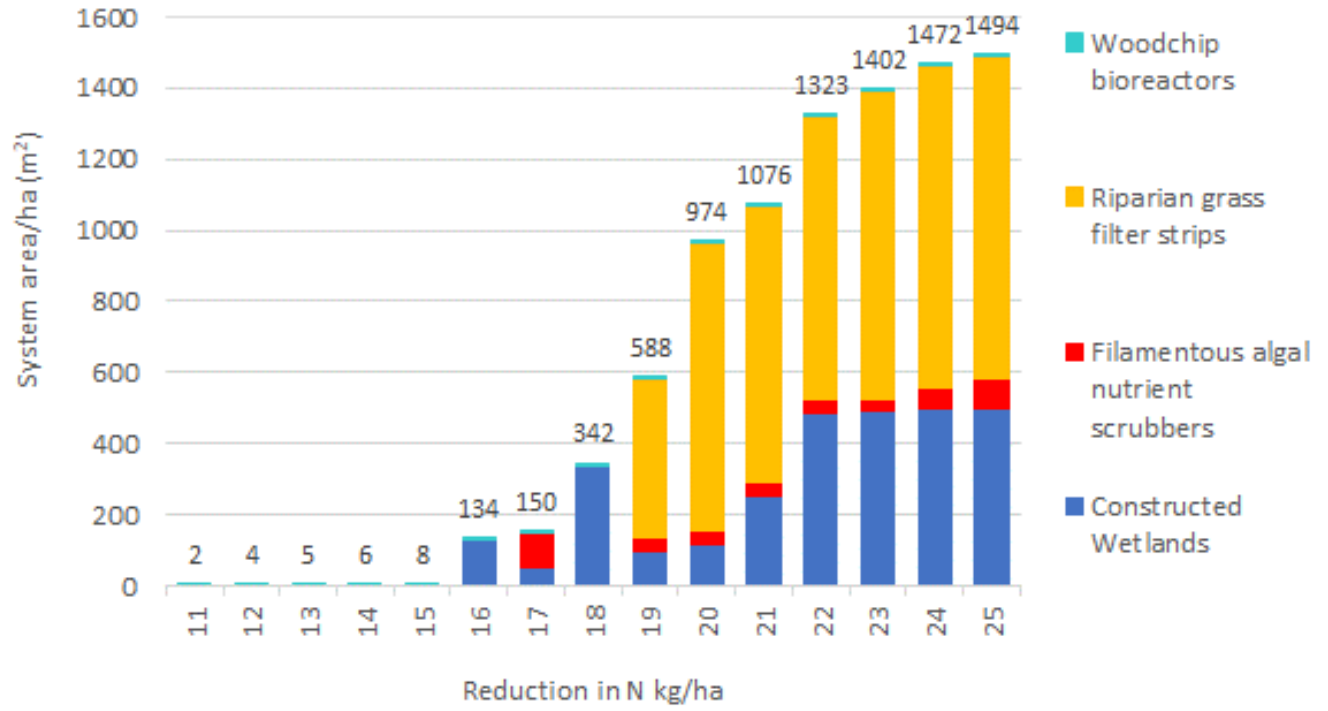


Figure 7. Most cost-effective mix of systems for reduction of the anthropogenic TN load in a low gradient dairy catchment with low permeability soils.

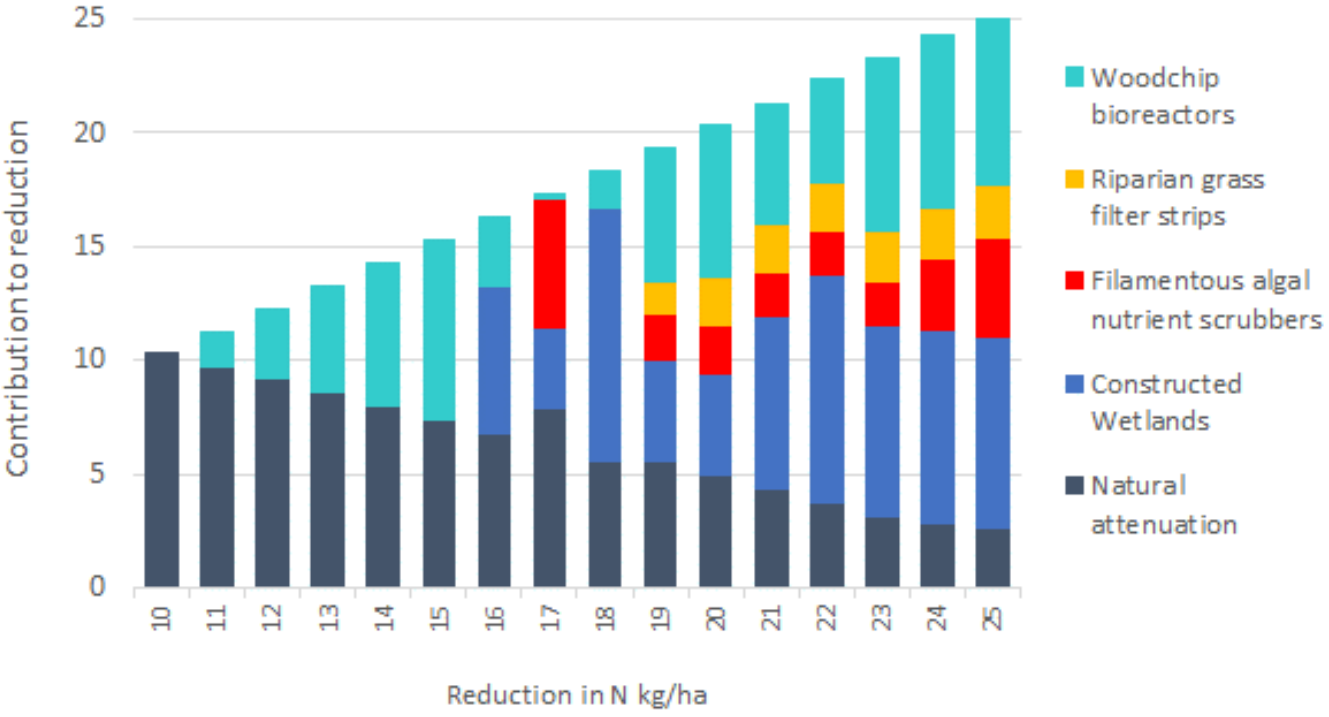


Figure 8. Contribution to TN removals of systems in a low gradient dairy catchment with low permeability soils.

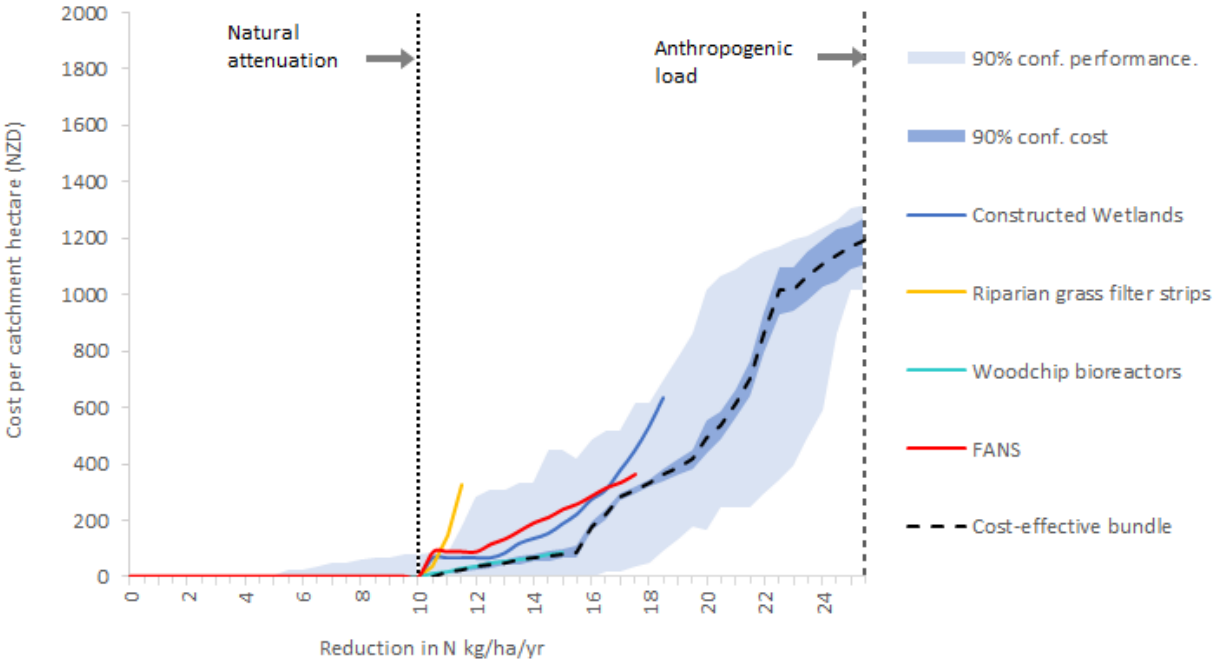


Figure 9. TN reductions for scaled implementation of individual mitigations and the least-cost combination in a low gradient dairy catchment with low permeability soils. Note that all costs are annualized and in New Zealand dollars.

6.1.2. Mitigation of TP

For the smallest TP removal targets, the most cost-effective system is a minimum-size DB (Figure 10). Due to the limited suitable area for a DB in the relatively flat landscape being evaluated, CWs or FANS are required for larger reductions. Removal capacity in a GRB is assumed to be limited by the low permeability of riparian soils in these hydrogeological conditions, so GRBs only feature to make up small differences when the specified reduction is not quite achieved by other mitigations. The entire anthropogenic load can be removed most cost-effectively using only DB and FANS, but the removal uncertainty is relatively high.

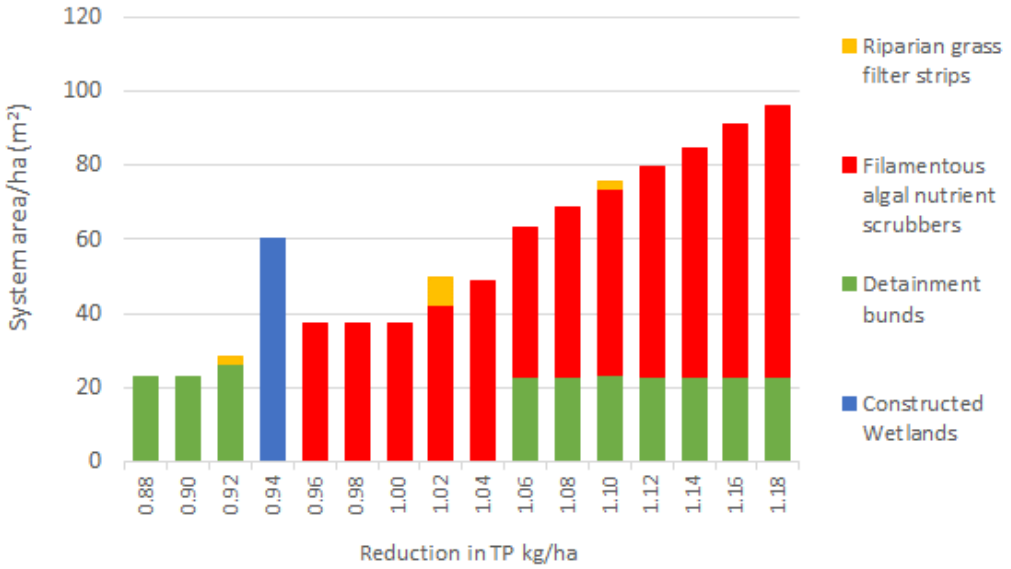


Figure 10. Most cost-effective mix of systems for reduction of the anthropogenic TP load in a low gradient dairy catchment with low permeability soils.

About a 50% reduction in catchment TP load is predicted due to natural attenuation (in stream beds) alone. Removal of the remaining anthropogenic load (i.e., Figure 11 returning to baseline levels) is expected to be able to be achieved for a cost of NZD 168/ha, with a cost confidence interval of NZD 159–177/ha.

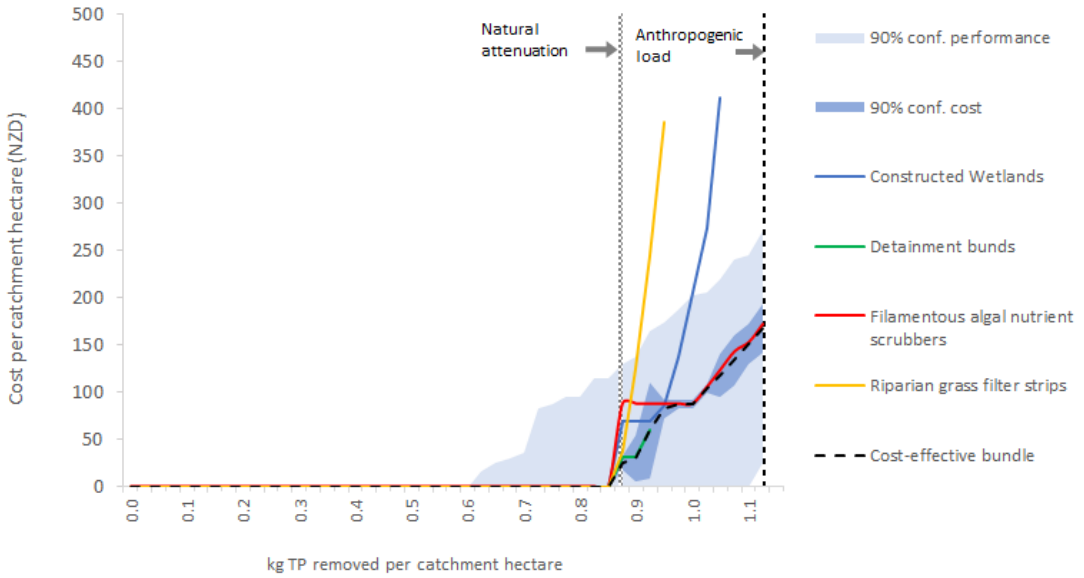


Figure 11. TP reductions for scaled implementation of individual mitigations and the least-cost combination in a low gradient dairy catchment with low permeability soils. Note that all costs are annualized and in New Zealand dollars.

6.1.3. Mitigation of TSS

For TSS, a DB is again the most cost-effective removal option. Due to landscape limitations on DB applicability for this scenario, a CW is the most cost-effective if larger removals are required. It is more cost-effective to have only a small CW rather than to stack a DB and CW. As the CW's size increases[8] above 0.9% of the catchment, it becomes more cost-effective to add a GRB rather than increase the size further. The anthropogenic load is only 0.8 t/ha because sediment losses are not massively elevated compared to the natural baseline in this relatively flat landscape. The entire anthropogenic load (0.8 t/ha) could theoretically be removed using a 95 m^2 CW and 8 m^2 GRB per farm hectare (~1% of catchment area; see Figure 12).

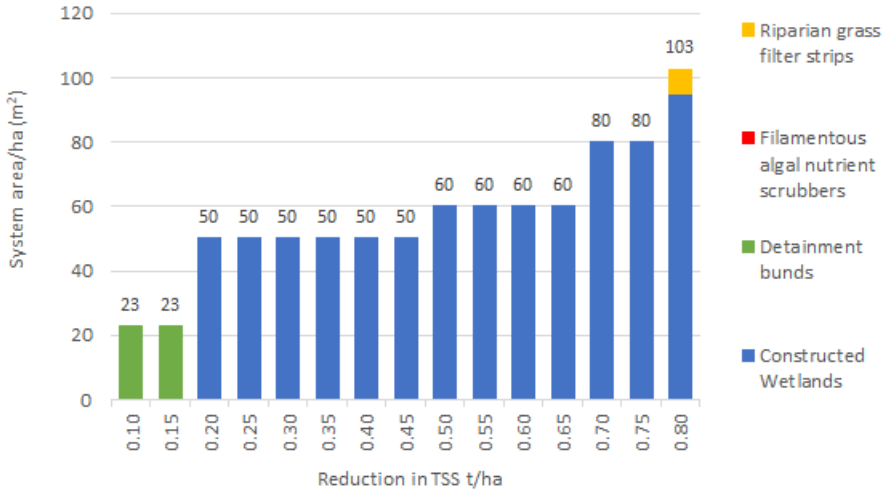


Figure 12. Most cost-effective mix of systems for reduction of the anthropogenic TSS load in a low gradient dairy catchment with low permeability soils.

The expected cost of removing the anthropogenic TSS load is only NZD 134/ha (Figure 13). The 90% confidence interval on the cost of this bundle is NZD 121–147, and the 90% confidence interval on removals is 1.087–1.276 t TSS/ha.

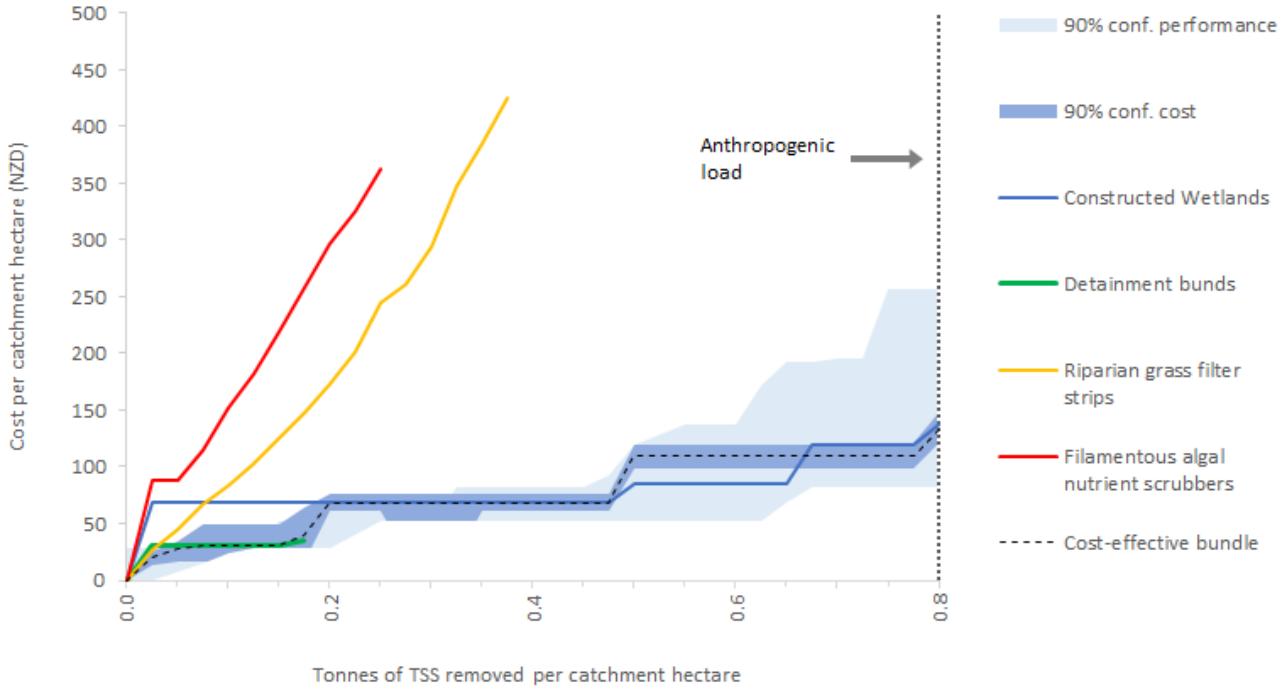


Figure 13. TSS reductions for scaled implementation of individual mitigations and the least-cost combination in a low gradient dairy catchment with low permeability soils. Note that all costs are annualized and in New Zealand dollars.

6.1.4. Comparison of Cost-Effective Systems for Each Contaminant

The most cost-effective systems for each individual contaminant will also remove other contaminants to varying degrees (Table 2). The system optimized to remove anthropogenic TN load, for example, also removes the anthropogenic TP and TSS loads. The system optimized for TP has only a small impact on TSS. Finally, the system bundle optimized for TSS offers low TN and TP reductions for a relatively low price of NZD 134/ha.

Table 2. Comparison of optimized system sizes per catchment hectare for removal of full anthropogenic loads. The 90% confidence intervals are in parentheses; removals include natural attenuation.

Focus	System Size per Catchment ha	Cost (NZD /ha/Year)	TN Reduction (kg/ha)	TP Reduction (kg/ha)	TSS Reduction (t/ha)
TN	Constructed wetlands (500 m^2) Filamentous algal nutrient scrubber (84 m^2) Riparian grass filter strips (904 m^2) Woodchip bioreactors (7.3 m^2)	1195 (1112–1278)	25.3 (22.9–27.7)	1.4 (1.3–1.5)	1.6 (1.3–1.9)
TP	Detainment bunds (23 m^2) Filamentous algal nutrient scrubbers (59 m^2)	168 (142–194)	13.0 (6.4–19.7)	1.1 (1.0–1.3)	0.3 (1.0–1.2)
TSS	Constructed wetlands (95 m^2) Riparian grass filter strips (8 m^2)	134 (121–147)	14.1 (7.2–21.0)	1.0 (0.8–1.2)	0.8 (0.6–1.0)

7. Discussion

The results of our scenario show that, for each contaminant, it is generally more cost-effective to combine small amounts of different mitigation systems if moderate reductions

are required than to target a single mitigation system. However, the optimal mix and order of mitigation systems in terms of cost-effectiveness is different for each contaminant. Trade-offs must, therefore, be made if the combination of mitigation systems is to be affordable. We show that TP and TSS could theoretically be reduced to natural baseline levels for less than NZD 200/ha, well below the operating profit of the majority of D2 farms. However, TN reductions are less affordable in this landscape type.

We find that including FANS in second-order streams allows the reduction of nutrients above and beyond the ability of other interceptive mitigation systems. In HLT types, where more nutrients are transported by shallow groundwater (and can only be intercepted by a planted buffer), FANS may be even more important. However, the cost-effectiveness of FANS depends critically on nutrient concentrations and flow variability.

The illustrative scenario also highlights that targets are important for determining the most cost-effective bundle of mitigation systems. A more ambitious target requires a more diverse set of mitigation systems, and the optimal bundles vary depending on which contaminant(s) have priority. Our model allows the user to specify a target for each contaminant and generate the optimal bundle of mitigation systems for meeting that target, subject to a specified level of certainty.

7.1. Comparative Analysis

Previous economic analyses in New Zealand, such as those by Muller et al. [11], have evaluated the cost-effectiveness of some of the mitigation measures discussed in this paper. However, their analysis did not consider the variability in costs and effectiveness arising from different HLTs and slope classes or include a measure of uncertainty. Furthermore, their cost estimation assumes near universal suitability across the catchment, which does not recognize limitations in which flow paths different mitigations are able to intercept effectively. Their estimated implementation costs and performance estimates were also limited to a small number of studies and did not consider the potential effects of varying sizes and combinations of mitigation systems. Our modeling approach thus provides a significant advance from this earlier work by incorporating a broader range of variables and scenarios, thereby providing a more comprehensive understanding of the economic impacts of these mitigation strategies.

7.2. Policy Implications

The 2020 National Policy Statement for Freshwater Management [1] emphasizes the need for a tailored approach to managing freshwater resources. It acknowledges that water bodies and ecosystems vary greatly across different regions and that the impacts of activities on water quality and quantity are context-specific. The modeling approach we describe in this paper can help regional councils to set objectives and limits that reflect local conditions and priorities. For instance, our results show that combinations of varying-sized mitigation systems can be optimized for cost-effectiveness for different landscape and farm types and different contaminant reduction targets. McWilliam and Balzarova [6] suggest that dairy companies often lack motivation to encourage farm-scaled green infrastructure. However, we contend that our modeling approach, when integrated with appropriate catchment-level targets, can more effectively highlight the benefits and justify strategic investments in green infrastructure. This could foster greater industry motivation to invest in these systems due to the clearer demonstration of their environmental and economic benefits.

7.3. Limitations and Future Opportunities

We acknowledge the following limitations in our modeling approach and analysis. Firstly, the availability of detailed costings is limited to a small number of case studies for some mitigation systems, which could affect the robustness of estimated costs. In particular, the full costs associated with implementation may be underestimated for newer mitigation systems that have rarely or never been implemented at the scale simulated in this study

(e.g., FANS). Conversely, we also note that some case study sites (e.g., for CWs and DBs) were designed for research outcomes, which may have inflated costs.

Cost-effectiveness is sensitive to key assumptions, such as slope class (for CWs and Bs), stream nutrient concentrations (for FANS), and total waterway length and suitability (GRBs). Special provisions to enable fish passage (e.g., online CWs) may provide further limitations on applicability and incur additional costs above those estimated. The cost of mitigations requiring earthworks or temporary water abstraction from streams (e.g., off-line CWS or FANS) may also be higher where resource consents or oversight by a Chartered Engineer are required. This sensitivity will result in wider variations in cost-effectiveness across different real-world catchments. The use of cost-effectiveness analysis also limits the scope of assessment to a single outcome metric. In real-world applications, other factors may be considered in addition to cost-effectiveness.

Lastly, the scarcity of information regarding the current level of implementation of edge-of-field mitigations poses a challenge in generating realistic results for actual catchments. For example, dairy farms are meant to already have livestock excluded from all waterways wider than 1 m and deeper than 30 cm [51]. It would be ideal to use the model in collaboration with land managers who can provide specific data inputs, enhancing the model's applicability and accuracy in real-world settings.

The estimated cost-effectiveness of individual and bundles of mitigation options in this study is based solely on expected changes in N, P, and total suspended sediment. However, the mitigation options also have other co-benefits, which can be important to landowners and/or communities. For example, CW can also remove a range of other contaminants and provide esthetic, biodiversity, and Māori cultural benefits [8]. DBs and CWs provide storm-flow moderation benefits that may help protect agricultural productivity and infrastructure (e.g., roading) downstream in lowland catchments. Planted buffers, which were not modeled for this exercise, also provide ecological and carbon sequestration benefits, although they are not presently eligible for carbon credits.

Another limitation is that we assume the sub-catchment either has a single land manager or that managers are able to take communal action to enact the most cost-effective combination of systems. The potential for collective responses to diffuse pollution mitigation, as evidenced in the Waituna catchment study by Weeber et al. [12], suggests a re-evaluation of traditional individual farm-based approaches. Collective strategies across catchments could offer cost efficiencies but also introduce complexities for equitable implementation across diverse agricultural landscapes. This aspect underscores the necessity for adaptable policies that can mediate the balance between collective benefits and individual farm impacts, particularly in scenarios where mitigation responsibilities might be unevenly distributed.

Seasonal variation in flows, farm losses, or mitigation system effectiveness present opportunities for future research and model refinement. The main barrier to including seasonal variation in our model is a lack of data about impacts on system performance. Similarly, we do not model planted buffers or pathogen reductions due to a lack of reliable data about the relationship between mitigation scale and effectiveness. There is also an opportunity to incorporate a cost-effectiveness measure that is a weighted average of different contaminants rather than a separate measure for each contaminant.

8. Conclusions

This study presents a quantitative model for assessing the cost-effectiveness and uncertainty of green infrastructure for reducing nutrient and sediment losses from agricultural catchments. Extending the framework of Tanner et al. [7], it incorporates case study data and detailed costing models for a nuanced assessment of interceptive mitigation systems. Using Monte Carlo simulation to address uncertainties, we integrated hydrological flow paths and contaminant transport dynamics, evaluating each mitigation system individually and in combination to determine the optimal mix and scale for specific landscapes. In a typical lowland tile-drained pastoral dairy catchment, reducing the anthropogenic nitrogen

load is the costliest, requiring multiple mitigation systems at up to NZD 1300/ha/year. However, targeting a 50% reduction costs around NZD 300/ha/year and also moderately reduces phosphorus and sediment loads. Targeting phosphorus or sediment specifically costs less than NZD 200/ha/year but only slightly reduces nitrogen. The most cost-effective mix of mitigation systems depends on landscape characteristics and water quality targets, emphasizing the need for a tailored approach to freshwater management. Our unique contribution lies in simulating variability in cost and performance, providing insights into the potential outcomes of implementing these systems. While we illustrate a scenario using a single farm type and hydrologic landscape, the model can be applied to various catchments. The findings highlight the importance of combining different mitigation systems strategically, especially given the diminishing returns at larger scales. This study contributes to agricultural water management by developing a decision-support model for assessing the cost-effectiveness of green infrastructure options. Integrating green infrastructure into farm and catchment management plans holds promise for improving freshwater ecosystems and supporting the sustainability of agricultural landscapes.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/land13060748/s1, Table S1: Fence costs by farm and slope type; Table S2: Estimated proportion of catchment contributed TN in each flow path for two slope classes and four hydrologic landscape classes; Table S3: Estimated proportion of catchment contributed TP in each flow path for two slope classes and four hydrologic landscape classes; Table S4: Estimated proportion of catchment contributed TSS in each flow path for two slope classes and four hydrologic landscape classes.

Author Contributions: Conceptualization, R.J.C. and C.C.T.; Methodology, Y.S.M.; Validation, F.E.M.; Investigation, R.J.C. and C.C.T.; Data curation, P.H.; Writing—original draft, Y.S.M.; Writing—review & editing, Y.S.M., P.H., F.E.M. and C.C.T.; Project administration, P.H.; Funding acquisition, C.C.T. All authors have read and agreed to the published version of the manuscript.

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Data Availability Statement: The data presented in this study are available on request from the corresponding author, subject to commercial and privacy restrictions.

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Conflicts of Interest: The authors declare no conflicts of interest.

Notes

1 https://www.treasury.govt.nz/information-and-services/state-sector-leadership/guidance/reporting-financial/discount-rates, accessed on 1 November 2023.

2 The regulations do not apply to small or ephemeral streams, or where there is an existing fence with a setback < 3 m. A farm may be compliant but not have buffers wide enough to be an effective filter.

3 Source: https://www.ravensdown.co.nz/ and https://mchort.co.nz, accessed on 1 November 2023.

4 Information about the extent and location of tile drains in New Zealand is limited. Likely areas based on soil and slope characteristics and available records have been mapped by Manderson [48].

5 Based on a representation of the catchment as a long rectangle bisected by the entire length of waterways.

[6] Source: NZ River Maps (https://shiny.niwa.co.nz/nzrivermaps/, accessed on 1 November 2023).

[7] Despite being at the bottom, FANS are not affected much by removals by other mitigations because the 2nd-order stream brings nutrients from upstream catchments. This external dependency would require consideration if we were aggregating results across multiple catchments.

[8] Due to the discrete size steps used in optimisation, the size is flat across a range of removal ambitions.

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