Memorandum: Assessment of potential for modelling an extended set of *E. coli* mitigation measures for Horizons and Taranaki regions

From: Sandy Elliott, Rebecca Stott, Chris Tanner (NIWA); Richard Muirhead (AgResearch)

To: Marie Patterson, Horizons; Thomas McElroy, Taranaki Regional Council

23 May 2024

Purpose and scope

This memorandum documents findings from an Access2Experts project conducted by NIWA and AgResearch for Horizons Regional Council (HRC) and Taranaki Regional Council (TRC). The advice addresses modelling of the microbial indicator *E. coli* in rivers and streams in these regions. The project follows from earlier 'Stage 2' modelling by NIWA (Semadeni-Davies et al., 2023a; Semadeni-Davies et al., 2023b; Semadeni-Davies et al., 2023c) in which the effects of several mitigation measures were assessed to inform regional planning. The mitigations included removal of dairy shed effluent discharge to streams in Taranaki, completion of TRC's riparian protection and stream fencing programme, reduction of point sources in the Horizons Region, and stock exclusion according to stock exclusion regulations applied in the Horizons Region. The catchment model CLUES was applied for that purpose.

The current project examines whether additional mitigations could improve water quality further, and provides advice on whether additional modelling is warranted. The project scope covers the following work items:

- Review relevant industry and literature information sources and draw on experience to advise on additional *E. coli* mitigations that could achieve reductions beyond what is currently modelled by TRC and HRC.
- Consider prioritisation and relative efficacy of different mitigations being put forward for consideration of modelling to inform recommendations.
- Prepare a joint guidance memorandum summarising findings and recommendations to help TRC and HRC make decisions regarding whether they should progress into their next stage of proposed modelling. The joint guidance memorandum shall summarise the findings of the literature review and provide an assessment of the *E. coli* mitigation strategies that have been considered.

The project provides a preliminary assessment of these matters, and the analysis is restricted in its depth due to the nature of the funding.

The assessment focusses on the implications of mitigation measures for *E. coli*, and does not address further considerations such as cost, co-benefits of the mitigation for improving other environmental conditions, and acceptability or cultural, social and economic benefits of the mitigations.

Methodology

A list of potential mitigation measures was generated based on systems that are used or could be reasonably used in New Zealand pastoral systems, based on knowledge of the authors and HRC and TRC staff, and examining existing summaries such as McDowell et al. (2013), Tanner et al. (2023a), (Collins et al., 2007) and the Our Land and Water list of Farm Environment Plan actions[1].

The potential mitigations considered in this assessment are listed below:

1. Additional SLUI land retirement
2. Additional stock exclusion from streams
3. Additional improvements to wastewater treatment
4. Detainment bunds
5. Vegetated riparian buffers
6. Constructed wetlands
7. Exclusion from natural wetlands
8. Improved onsite domestic waste disposal
9. Diffuse urban source management
10. Woodchip bioreactors
11. Bridge stream crossings
12. Deferred and low-rate effluent irrigation
13. Enhanced dairy effluent ponds
14. Off-pasture confinement
15. Restricted winter grazing
16. Reduce stocking rate
17. Stock exclusion from critical source areas
18. Strategic grazing of fodder crops
19. Alternative deer wallows
20. Feral animal control

In some cases, the additional mitigation is simply extending the current modelling approach to a larger area, so the current modelling approach and extension is summarised. For other mitigation measures, the following information was assessed:

- Brief description of the mitigation measure
- Key existing literature
- Assessment of removal efficiency for the flow paths and areas treated by the mitigation
- Consideration of potential overall removal of the mitigation taking into account the efficiency above, the flow paths treated, and the degree to which the mitigation could be implemented
- Potential for modelling the mitigation measure
- Overall assessment of the effects of the mitigation and the priority to model it.

For some mitigations, there was minimal information available, in which case not all of the points above were addressed.

The information was then summarised to provide an overall assessment of the potential for the mitigation to improve water quality and to be included in a model. Removals were classified as very

[1] https://ourlandandwater.nz/fep-actions/

low (<10%), low (10 to 25%), moderate (25-50%), good (50-80%) and high (>80%), where the numbers are coarse indications rather than intended to be used directly in modelling.

As a general note, reductions are sometimes expressed as x $\log_{10}$ removal, denoting that loads or concentrations are reduced by a factor of 10^x. So, for example, 2 $\log_{10}$ corresponds to 99% removal.

Assessment by mitigation type

We begin this section with some mitigation measures that are extensions of measures currently modelled, or that are able to be modelled with the existing model parameters, and hence do not need to be addressed in the same manner as other mitigation measures.

1. Additional SLUI land retirement

In the Horizons region, a soil conservation scheme has been underway for many years to reduce erosion from hill-country land with high erosion potential. Soil conservation plans may also be extended. The erosion control measures include planting, sediment traps, and fencing. Sediment traps and stream fencing could be modelled as discussed elsewhere in this memorandum, provided that the degree of implementation associated with SLUI is known. Reversion of pasture land into native forest (with appropriate fencing of the retired area) could be modelled by replacing land uses in microbial model, specifically the proportion of pasture within the catchment, provided that the area of retirement is known. There was some uncertainty in the model calibration about the yield of microbes from forest, and the yield from retired forest. There is also some uncertainty about the yield of microbes from steep hill areas that are likely to have low stocking rates. The current model does not differentiate pasture areas based on slope or stocking rate. Hence there will be uncertainty about the degree to which retirement will reduce microbial loads. With these provisos, it would be straightforward to estimate the benefit of SLUI land retirement. The extent of land retirement may be modest overall, so that the overall effect on loads might be small, but given the interest and simplicity of additional modelling, and this item should be given high priority.

2. Additional stock exclusion from streams

The previous modelling considered a range of options for stock exclusion from streams.

For Horizons, the modelled degree of stock exclusion for Horizons was to implement the Resource Management (Stock Exclusion) Regulations 2020, which increased the proportion of streams fenced from 13% to 23%. The existing extent of fencing was taken into account in the scenarios. There is scope for increasing the extent of fencing since pasture covers 52.3% of the catchment area (according to the model calibration report). The removal efficacy of stock exclusion was assessed, with a range of removal efficiencies of 24%, 62%, and 92% for low, medium and high removal scenarios respectively. The effect of additional fencing could be modelled in the same way as in the original modelling (with similar uncertainty and provisos). The cost would likely need to be assessed differently, considering that the additional fencing would be in steeper areas. There is political interest in the cost and benefit of full fencing, so that modelling of additional fencing in the Horizons region remains a medium or high priority.

In Taranaki, current fencing and riparian planting has so far largely targeted the intensive dairy farming zone (mostly in the Ringplain and Coastal Terrace FMUs). Properties <20 ha have not been targeted, and nor have Hill Country areas outside of what is required under the Resource Management (Stock Exclusion) Amendment Regulations (2023). The past stock exclusion scenario for Taranaki was based on completing the planned fencing and riparian planting programme, so that an additional 17% of the stream length in the region would have new or improved fencing, beyond the fairly high current degree of fencing. There may also be room for further improvements with improved riparian margins and buffer widths in accordance with new regulations. Additional fencing may have modest impacts on water quality overall, based on Stage 2 modelling. However, this is a scenario of high interest. Hence, modelling of additional fencing in the Horizons region is recommended.

3. Additional improvements to wastewater treatment

Reductions of wastewater loads though improved wastewater treatment were considered in the earlier modelling reports. For Horizons, anticipated reductions in wastewater loads, with 100% reduction for sites such Palmerston North and Foxton. This resulted in a small reduction in the regional *E. coli* load and very little change in microbial attribute values or bands even locally, except for Woodville and to a small degree Rongotea. Considering the small amount effect of anticipated significant improvements, and the small scope for further source reduction, further modelling of point source reductions is not warranted for Horizons.

For Taranaki, reduction of municipal discharges was not considered. Discharges from Discharges from New Plymouth, Hāwera and Ōpunake are to marine outfalls, and it is expected that removal of discharges from small inland towns like Stratford would make little impact on water quality for the region overall. Even locally, the discharge from the Stratford WWTP had little effect on microbial water quality of the Patea River based on upstream-downstream monitoring (Taranaki Regional Council, 2022). Nonetheless, there may still be local political reasons for improving wastewater treatment.

The overall assessment is that there is low priority on modelling wastewater treatment further, for either region.

4. Detainment bunds

Description of the mitigation

A detainment bund is an embankment across small intermittently-flowing channels in headwaters, to create temporary ponding areas (Paterson et al., 2020). They are also sometimes referred to as silt traps or earth bunds. *E. coli* removal mechanisms include settlement (for microbes associated with sediment particles) and infiltration. They primarily treat surface runoff from small catchments.

Key literature on removal efficiency

Smith and Muirhead (2023) reviewed information on effectiveness of sediment traps, including detainment bunds in a range of New Zealand and international studies (including a recent study by Levine et al., 2021). Sedimentation removal efficacy was assessed, with a typical removal of 55% of sediment and a range of values depending on size of the device and other factors. Removal for *E. coli* was not assessed, and is likely to be lower than the sediment removal if there is no infiltration because only part of the microbial load is associated with coarser sediment that the bunds preferentially trap. The performance is likely to be highly dependent on the degree of infiltration, which will in turn depend on soil characteristics, the location of the site (e.g. subject to saturation) and the degree of reduction of infiltration due to sealing by prior sedimentation.

For a summary of information on *urban* retention ponds, (Tilman et al., 2011) found a wide range of removal efficiencies for *E. coli*, with representative values of 78% and 90%. The urban Stormwater BMP database indicates a removal of approximately 45% for detention basins (Clary et al., 2020). Although urban ponds are likely to be different from rural detainment bunds, the results give an order-of-magnitude removal from a generally larger database than is available for rural detainment bunds.

Daigneault and Elliot (2017) assessed a removal efficiency of 50% for sediment traps, based on literature review, which was applied to the total *E. coli* load.

Stott et al. (2022) reported recent measurements of *E. coli* removal for a detainment bund for a site with free-draining soil at Rotorua. Concentrations were reduced by 50-75%, and for one event 55% of the load was removed. Typically 30-40% of inflow infiltrates at this site (based on other studies at the site).

Tanner et al. (2023a) assessed a detainment bund removal efficacy of 70-90% for *E. coli* transported in surface-flow pathways based, on studies by Stott et al. (2022) and preliminary results from the PMP project (Andrew Hughes, pers. com. Feb, 2024).

Conditions of applicability and overall removal

The removal efficiencies above apply to the surface runoff components of microbial loss from headwater areas and does not intercept runoff from other sources or pathways such as saturated areas near streams. Detainment bunds can only be constructed in areas where the flow has converged and where there is a suitable configuration of the topography at the bund site (so that a bund can be constructed and will pond water). Also, bunds will not be appropriate on very steep or flat catchments (Tanner et al., 2023b) due to difficulties such as constructing suitable storage areas. They will therefore only be applicable in some topographies (gently undulating to rolling hill-country) and will only treat a portion of the catchment even when topography is suitable. They are unlikely to be effective in areas with subsurface drainage. Assessment of the applicability will require some closer consideration of these factors, probably by detailed spatial analysis in some representative locations.

Bunds are likely to be more efficient on free-draining soils to maximise infiltration. However, such areas are likely to have a lower proportion of overall microbial loading coming from surface runoff, which will diminish their overall effectiveness taking other sources into account.

Detainment bunds are likely to achieve good removal (50-90%) of *E. coli* from surface runoff, with removal on the higher end of this range in permeable areas where runoff can infiltrate. Detainment bunds are applicable in only some regions and only treat one flow path, so that overall their removal efficiency will be moderate (25-50%).

Ability to model

Modelling would require assessment of suitable locations based on topographic analysis, and consideration of the importance different flow paths. Techniques to achieve this are available. Flow path considerations could be added to CLUES.

Overall assessment

It is considered that there is moderate priority to pursue further assessment and modelling of this mitigation measure considering the moderate removal and that they may be amenable to representation in models. A high-priority precursor would be to conduct an assessment of the locations where detainment bunds would be suitable as a preliminary exercise, and to assess how different flow paths could be incorporated into the assessment of load reduction.

5. Vegetated riparian buffers

Description of the mitigation

Vegetated Riparian Buffers (VRB) are perennial vegetated strips of land strategically positioned along or above streambanks to provide a protective barrier between potential pollutant sources and waterways. VRBs can be generally classified into three types: grass filter strips; planted riparian buffers; and multi-function buffers comprising a combination of filter strip and planted buffer (McKergow et al., 2022). Typically, stock would be removed from such areas by fencing.

Grass filter strips (GFS) are managed bands of dense vegetation (commonly grass) which act as physical barriers that intercept and slow ephemeral surface runoff trapping contaminants and promoting infiltration. Planted riparian buffers (PRB) involve the establishment of a mix of usually native deeper-rooted trees and shrubs which intercept shallow subsurface flows through the root zone. Combining grass filter strips and planted buffers in multi-function riparian buffers effectively intercepts both surface runoff and subsurface flows.

Key literature on removal efficiency

Removal of faecal contaminants in VRBs is through a combination of physical and biological processes including infiltration, deposition, filtration, adsorption and die-off.

Vegetated riparian buffers are effective in trapping particulate contaminants. The deposition of these contaminants within a GFS are influenced by their size particularly where dense vegetation cover reduces water velocities allowing particles more time to settle. Once released into overland flow, *E. coli* has the potential to be transported either as individual cells or attached to manure or soil particles. There remains some uncertainty regarding the proportion of *E. coli* transported in overland flow as attached or unattached cells, but the majority of *E. coli* are likely transported as unattached from fresh manure sources (Soupir and Mostaghimi, 2011). Although *E. coli* may therefore not settle to the same degree as other particulates due to lower settling velocities, some microbial die-off of trapped microbes within the GFS environment would occur.

Grass filter strips

Concentrations of *E. coli* in surface runoff entering GFS can be highly variable ranging from 10 – 10^7 *E. coli* MPN/100mL (Collins et al., 2004; McKergow, 2008). Although GFS may be useful in reducing these concentrations, the efficacy of GFS can vary significantly, depending on slope, surface area, width, flow rates, preferential flows during saturation, vegetation type and density, soil infiltration capacity, rainfall intensity and duration, topography and hydrology and flow channelization (Guber et al., 2009; McKergow et al., 2007).

The performance and mechanisms of GBS for *E. coli* removal was reviewed by Olilo et al. (2016a). Reported performance shows substantial variability with load and concentrations reductions ranging between 0 and 99% (Collins et al., 2004; Olilo et al., 2016a); Olilo et al. (2016c). Olilo et al. (2016c) reported *E. coli* load reductions for GFS (based on international data) varying between 68-99.6% depending on grass type with higher removal rates attributed to surface vegetation slowing overland flow velocity thereby enhancing filtration of *E. coli* by vegetative leaf litter and infiltration and detention of *E. coli* in the subsoil horizon. Most of the reduction occurred within the first 10 m of the GFS. Similar *E. coli* load reductions of 61-94% were reported by Miller et al. (2015) for GFS widths 1.5 m to 6 m (parallel to surface runoff). The removal efficacy of GFS declines with increasing flow as large flows overwhelm the filter strip compromising their trapping efficiency and reducing infiltration rates (McKergow, 2008; Tate et al., 2006). Lower flow velocities reduce the momentum of surface runoff facilitating the entrapment and infiltration of overland flow and microbes into the soil profile. Nevertheless, in subsequent events, trapped microbes may be mobilised and washed from filter strips, transforming GFS into both a sink and source of *E. coli* (Beck et al., 2013; Collins et al., 2004).

Though large reductions in *E. coli* concentrations and loads have been reported for GFS, studies typically focus on small plots in controlled environments with simulated rainfall events for short duration trials. Real-world strips are likely to have considerably lower performance due to factors such as bypassing or concentrated flow paths and degradation of the condition of the strip over time.

Few *E. coli* removal rates have been reported for field-scale GFS. If removal is assumed to follow relationships for sediment removal (see McKergow et al., 2022), for a filter with a width:hillslope length ratio of 3% and surface runoff for a non-clay soil with low to gentle rolling slopes (<11 degrees), *E. coli* removal could average at least 50% increasing to 70% for a 6% hillslope ratio based on hillslope length for the same conditions. This is, however, dependent on factors such as filter strip condition and bypass flow, and the proportion of microbes associated with settleable material.

Planted riparian buffers

For PRB, removal of microbial contaminants from subsurface flows may occur through improving soil permeability, increasing the ability of water to infiltrate deep into the soil, adsorption onto soil particles and biofilms in the root zone system, and inactivation due to soil micro- and macro-fauna and potentially by antimicrobials produced by native plants such as Mānuka (Gutierrez-Gines et al., 2021; Prosser et al., 2016; Wheatley and Poole, 2018). Limited data exists on the removal efficiency of PRB. Generally, studies assess microbiological water quality in streams with PRB. Parkyn et al. (2003) compared stream quality in unbuffered reaches to fenced and planted reaches to assess the efficacy of planted riparian buffers ranging in age from 2 to 24 years at six sites in the Waikato. Changes in *E. coli* varied in buffered streams relative to control reaches (-60% to 100%) possibly due to limited one-off sampling and/or upstream contaminants entering unprotected reaches. The study emphasized the need for long contiguous buffer lengths to enhance efficacy. Corroborating this finding, a recent study by Lim et al. (2022) reported the longer the length of PRB (along the stream) the greater the improvement of water quality for minimum 15 m width planted buffers. For studies in areas with grazed pasture, it is difficult to separate the effect of riparian planting from the effect of stock exclusion, because riparian planting usually entails stock exclusion.

In Stage 2 modelling for Taranaki, a 10% load reduction (for buffers around all streams) was applied for riparian buffers beyond the value with fencing only, based on previous literature reviews and expert opinion. This figure was uncertain, and is small in relation to the removal that was related to stock exclusion from streams.

In the Taranaki region, the implementation of fencing to exclude stock from streams, which is typically combined with native vegetation planting, has led to a decrease in *E. coli* concentrations in waterways (Graham et al., 2018).

Conditions of applicability

Application of VRBs needs to consider prominent hydrologic transport pathways for microbial contaminants such as the relative importance of surface runoff as a pollutant pathway compared to subsurface pathways. Therefore, knowledge of the “hydrologic landscape” is needed.

GFS target contaminant loss in surface runoff and function best when surface runoff is uniformly dispersed as sheet flow. Thus GFS should ideally intercept contaminants close to the point of generation and before channelisation occurs (McKergow, 2008). GFS function poorly in areas that are often saturated due to limited filtration.

Application sites for GFS are riparian zones around streams, creeks, drains, wetlands, ponds and lakes. Situations where they are likely to have significant benefit are low-moderate permeability soils, gently undulating to rolling landscapes and areas with high intensity rainfall where surface runoff is a significant pathway for microbial contaminant transport (McKergow et al., 2007; Tanner et al., 2023a). GFS are unlikely to be effective in attenuating *E. coli* within overland flow under heavy rainfall on steep pastoral land (Collins et al., 2005) as flow convergence on steeper slopes (e.g. >15 degrees) promoting high flow velocities limits ability of filter strips to trap microbes. The likely applicability of grass filter strips in NZ is considered moderate (McKergow et al., 2007) as careful consideration of landscape characteristics is required in order to effectively intercept contaminants in surface runoff which often converges into channelised flow bypassing or locally inundating GFS typically located in riparian areas.

GFS are less useful where flatter slopes and permeable soils promote the vertical movement of water thereby limiting the generation of surface runoff and where bypass flows (e.g. rapid movement through subsurface soils via cracks and macropores) aids rapid vertical transport of microbes, with minimal attenuation through filtration or adsorption. In these situations, PRBs are more appropriate. Neither GFS or PRBs are suitable when there is tile drainage bypassing the riparian area.

PRB are designed to manage lateral subsurface flows. They are best suited where an impermeable soil layer forces subsurface flow to move through the root zone of riparian soils and in flat to moderate slopes. Faecal microbe loads in subsurface flows are generally less than in surface runoff but may be elevated in soils prone to bypass flows and where effluent irrigated or manure applied.

GFS and PRBs only intercept one flow path. However, multi-function riparian buffers that include GFS upslope of PRB, are designed to intercept both surface runoff (sheet flow) and subsurface flow and are considered to have high applicability in NZ (McKergow et al., 2007).

Overall removal

GFS have potentially high removal for microbes in surface runoff under ideal conditions. However, the limited extent of suitable locations, and difficulties with maintaining their condition over time, means that the overall removal may be low.

For PRBs it is presumed that *E. coli* concentrations will be minimal in the subsurface flows due to filtration occurring through the soil matrix and prolonged residence times fostering die-off. Median values are anticipated to be < 50 *E. coli* MPN/100ml in shallow subsurface flows, assuming *E. coli* levels are as low as observed in shallow subsurface flows in Waikato soils at Toenepi and Owl farm (Stott et al., 2023; Tanner and Sukias, unpublished data), and therefore moderate reduction of 50-90% might be expected.

For vegetated riparian buffers intercepting both surface and subsurface flows at baseflow, conservative estimates based on expert opinion suggest high removal reduction of at least 90% for *E. coli* through multi-function VRBs under ideal conditions where applicable.

Ability to model

Both mechanistic (process based) and empirical (regression based) models have been developed to describe and quantify the removal of contaminants from surface runoff by vegetated filter strips such as GFS (Yu et al., 2019).

Munoz-Carpena and Parsons (2011) developed the model VFSMOD-W as a dynamic event-based model system for overland flow and infiltration in GFS. However, such models are highly parameterised with site-specific parameters used to model experimental scale systems. Whilst the model has been used for sediment trapping and infiltration under field conditions, it may hold promise to model transport of microbes in overland flow and through VFS under a range of hydrologic conditions (Olilo et al., 2016b). Park et al. (2013) developed a Web GIS-based VFSMOD to design effective widths of VFS for trapping sediment in an agricultural catchment.

The dynamic catchment model SWAT has also been used for modelling GFS. It considers flow that bypasses the strip, and infiltration and sediment trapping of the sediment-associated microbes based on empirical formulae. In an application, Parajuli et al. (2008) used an older version of SWAT that had removal modelled as an empirical function of filter width, and predicted reduction of 60% from overland flow, but not by that much at the catchment outlet. SWAT models have also been used to explore the efficacy of vegetative filter strips in reducing faecal coliform abundance throughout a watershed (Bai et al., 2016), demonstrating sensitivity to input parameters such as the bypass fraction and fraction associated with sediment, and noted that targeted validation data is needed. Setting up such models is time-consuming, and appropriate parameterisation would be needed before there could be high confidence in the predictions.

There are several empirical relationships for filter strip microbial removal (e.g., Moore, 1988)

A major issue to consider when modelling the removal efficacy of VRBs and estimating potential improvements in water quality is the disjunct between hillslope scale and stream reach/catchment scale studies. For the latter, there may also be livestock exclusion providing *E. coli* reduction benefits (O'Callaghan et al., 2019) and setback of pastoral activities from the stream.

A consideration for modelling is assessment of conditions of suitability of GFS, in the Taranaki and Horizons regions; until this is quantified, modelling would not be appropriate.

Overall assessment

Comprehensive understanding of *E. coli* removal in vegetated riparian buffers (VRBs) remains limited with a scarcity of catchment scale investigations and a lack of knowledge about their long-term impacts, especially in New Zealand.

Most of the effect of PRBs (planted riparian buffers) is likely due to the associated stock removal. The current modelling approach involved applying a small amount of additional load removal for PRBs. This has already been incorporated into the Taranaki Stage 2 work, and could be explored further as part of an examination of extended stock exclusion using the existing modelling approach.

GFS have the potential to achieve high removal under ideal conditions, but there is considerable uncertainty about how widespread and suitable GFS would be, and how well they would operate in real pastoral system, and whether the wide (about 5m) widths are practical. Therefore, it is recommended that further modelling of GFS have a low priority at this stage. However, further assessment of VRBs as mitigation measures is warranted given their potential for moderate removal, current prevalence of riparian management, and parameterisation in international modelling efforts. Practical considerations may also dictate the degree to which VRBs could be introduced. As a preliminary step, it is recommended that spatial analysis could be conducted to determine whether there are suitable topographic and soil conditions for GFS in the region.

6. Constructed wetlands

Description of the mitigation

Constructed wetlands (CW) are shallow (<0.5 m) impoundments vegetated with emergent aquatic plants that intercept and pond water. They are most effective for removal of nitrate (through microbial denitrification) and moderate sediment and particulate P loads (through settling). CWs can treat surface runoff, surface and subsurface (i.e. tile) drainage, interflow and shallow groundwater seepage. They can be employed at multiple scales, including at the bottom of catchments before flows enter lakes and estuaries, and can provide ancillary habitat/biodiversity, flow moderation, carbon sequestration, aesthetic, and cultural benefits (Tanner et al., 2023a).

Key literature on removal efficiency

NZ constructed wetland practitioner guidelines for CW treatment of farm run-off were recently updated (Tanner et al., 2022). They do not provide information on *E. coli* removal performance, because reliable data was lacking for CWs treating pastoral runoff.

Stott et al. (2023) recently reviewed information on *E. coli* removal in constructed wetlands from various wastewater, stormwater and run-off sources in context of their use for diffuse agricultural run-off. This is summarised as follows. Studies on surface-flow CW treating continuous point-source inflows of domestic wastewaters have generally reported effective removal of at least 90% of faecal indicator bacteria (FIB), particularly when influent concentrations are high (in the range of 10^4–10^6 CFU/100 ml) (Ghermandi, 2007; Kadlec and Wallace, 2008). However, it has frequently proved difficult to achieve effluent concentrations of *E. coli* consistently below 550 CFU/100 ml in the final effluents. Additionally, wetland removal of FIB appears related to influent concentration (Wu et al., 2016) and where influent concentrations are less than ~1000 CFU/100 ml, effluent concentrations sometimes exceed influent values. A review of surface-flow (free-water) CW systems reported outflows from 13 out of 32 systems had increased levels of *E. coli*, sometimes exceeding 1000-fold (Kadlec and Wallace, 2008). Levels of *E. coli* have also been found to increase in a wetland receiving chlorinated wastewater, and in treated effluent after UV disinfection (Hallmich and Gehr, 2010; Orosz-Coghlan et al., 2006). This apparent increase of FIB has been attributed to photoreactivation, inputs from resident and visiting wildlife, re-suspension of bacteria which have previously settled in the CW, or bacterial growth within the wetland (Kadlec and Wallace, 2008).

Stott et al. (2023) undertook a detailed study of a NZ CW receiving tile drainage from a Waikato dairy farm. Determination of inflow and outflow *E. coli* fluxes and loads for ten storm events concluded that the wetland was consistently a net exporter of *E. coli* with increases ranging from 1.5 to 26-fold (median 10.2-fold). Whilst wildlife defecation in the wetland may account for a proportion of the rise in *E. coli*, it is likely that (consistent with associated genetic profiling studies outlined in Stott et al., 2023) much of the increase was due to mobilisation of environmentally adapted strains of *E. coli* persisting and growing within the sediment, decomposing plant litter and organic detritus of the wetland. Such naturalised growth in the wetland, which is decoupled from live animal inputs (although likely originally sourced from them), brings into question the utility of *E. coli* as a faecal microbial indicator of health risk for wetland discharges. In this case an increase in *E. coli* is likely to not result in an equivalent increase in zoonotic pathogen risk.

Monitoring of the Okaro constructed wetland intercepting agricultural streamflow from sandy loam soils in a hilly landscape near Rotorua (Hudson and Nagels, 2011) showed reductions of 92, 96 and 89 % over 3 consecutive years. Removal varied with flow and was closely correlated with TSS removal in the wetland.

Monitoring over 2 years at Baldwin's wetland at Litchfield, Waikato (Sukias and Heubeck, 2020) showed 85% and 65% reduction of *E. coli* over 2 years (2017, wet and 2019, dry). This wetland received baseflows from natural seepage wetlands on the farm and sediment-rich stormflows from a significant dairy farm track.

E. coli data has also been measured in the Owl Farm wetland (near Cambridge in Waikato; C. Tanner, unpublished data), which receives a mixture of diffuse groundwater and tile drainage inputs. .*E. coli* data was collected for baseflow and during 6 rainfall events over 3 years. Concentrations of *E. coli* measured in tile drain and groundwater inflows to the wetland were relatively low (commonly < 35 /100 ml). Increased concentrations were measured in the outflows of the wetland, similar or greater than those reported by Stott et al. (2023). However, unlike the Toenepi wetland studied by Stott et al. (2023), substantial waterfowl populations were observed in this wetland at times (e.g., Goeller et al., 2023) potentially contributing to faecal microbial loads to the wetlands.

Conditions of applicability

Constructed wetlands are applicable in a wide range of situations (Tanner et al., 2022) where they can intercept surface run-off, subsurface seepage, tile drainage and/or surface drainage and stream-flow up to the scale of approximately 1st and possibly 2nd-3rd order streams (latter likely to require provision for fish passage). Costs for construction are likely to be lower in gently to moderately rolling landscapes where the natural landform can be modified to form the wetland.

For larger streams there is the option of off-line deployment, treating a proportion of the streamflow (e.g. 50-75% of mean annual low-flow) and returning the treated water back to the stream near where it was extracted. Connecting streamflows to off-line wetlands in relatively low gradient areas is likely to require either deep excavation of the wetland base down to the level of the stream or pumping up to the wetland. Preliminary economic assessment of annualised costs suggest that pumping is likely to be more cost-effective in these cases, providing a power source is available.

Overall removal

Wetlands can be both sources and sinks for *E. coli*. In general, when influent concentrations are high, wetlands will reduce them, but when they are low they can increase them. The human health consequences of these increases are not clear where increases are due to naturalised variants of *E. coli*. On the other hand, increases due to waterfowl and other resident animals are likely to present zoonotic health risks to humans.

In general terms, then, CWs receiving a high proportion of surface run-off with high concentrations (>1000 /100 ml) will likely reduce concentrations and load by 60-90%. CWs receiving subsurface flows with low *E. coli* concentrations (<100 /100 ml) are likely to increase loads by around 10-fold, but this will vary depending on waterfowl presence and other factors.

Ability to model

Various dynamic process-based modelling approaches have been used for modelling *E. coli* removal in CWs processes (Boutilier et al., 2011; Kadlec and Wallace, 2008; Khatiwada and Polprasert, 1999). Hamaamin et al. (2014) applied an adaptive neuro-fuzzy inference systems (ANFIS) modelling approach to describe *E. coli* removal in pulse-loaded constructed wetlands. This model performed better than a mechanistic model which was also tested. Such models require considerable set-up effort and calibration data.

In terms of load-reduction-factor approaches, there is considerably uncertainty about how to represent the effects of influent concentrations and waterfowl on treatment performance.

Overall assessment

Further research is warranted to properly assess the performance of CWs in different landscape and farming situations. Additional data for at least 4 additional CW systems is currently being collected (along with flow, sediment and nutrient data) as part of NIWA research in collaboration with councils, with support from MPIs SLMACC-FM fund.

Despite significant potential, especially for treating areas producing high concentrations, at this stage it is difficult to provide reliable guidance on CW attenuation of *E. coli* loads for catchment-scale modelling. Accordingly, such modelling is given a low priority until more suitable data is available.

7. Livestock exclusion from natural wetlands

Description of the mitigation

Natural wetlands relevant to interception and mitigation of diffuse agricultural loads primarily include wetland seeps, riverine swamps and flood plains, fens and lake-edge wetlands that receive surface or groundwater flows from upslope or adjacent farmland. Apart from seepage wetlands, these types of natural wetland, where they are connected to watercourses, share similarities with surface-flow constructed wetlands and can be considered in the same way in terms of faecal contaminant attenuation. Natural peat bogs have little interaction with runoff, so can generally be discounted in terms of contaminant attenuation.

Livestock ingress to wetlands can increase contaminant loading by two mechanisms. First, livestock may defecate into the wetland and stir up sediment, thereby acting as a contaminant source. Second, they may reduce the condition of the wetland (by, for example, compacting soils), thereby reducing their contaminant removal efficiency.

Under current planning rules, stock are supposed to be excluded from natural wetlands. The current degree of compliance is unclear.

Key literature on removal efficiency

Restored natural wetlands have been shown to reduce nutrient losses from agricultural landscapes (Audet et al., 2020; Page et al., 2023), and remnant natural wetlands have been promoted as farm attenuation assets in New Zealand (Tanner et al., 2015). Although nutrient removal in natural seepage wetlands has been summarised by McKergow McKergow et al. (2017), *E. coli* removal was not included in that assessment.

Hughes et al. (2013) presented information on *E. coli* concentrations at the top and bottom of a natural wetland in hilly dairy pasture country in the Waikato that receives flow from shallow groundwater springs and surface run-off events. Concentrations at the bottom of the wetland were commonly <10% of the value measured at the top of the wetland during baseflows and for summer flow events. Reductions were generally less (~30%) for winter flow events that are larger and thus provide lower hydraulic residence time in the wetland. However, it should be noted that only around 10% of the water flow reaching the bottom of the wetland was measured at the top. That is, water and further contaminants were entering this wetland at multiple points down its length, which complicates interpretation. This is, however, a situation likely to be common in many similar natural wetlands. Multi-cell dynamic modelling of such systems is required to properly assess their contaminant removal performance, as has been done for nitrogen in this wetland (Uuemaa et al., 2018).

Hughes (2016) also studied the response of the wetland to specific grazing events and could find no effect, except when a cow became entrapped in the wetland overnight upstream of a monitoring site. Cow entry to the wetland was otherwise generally only superficial, suggesting that fencing of the margins of the wetland would be unlikely to have a significant effect on its contaminant removal performance.

In contrast, (McKergow et al., 2012) studying a larger, more shallow-shelving natural wetland in a Lake Taupo catchment found cattle commonly entered the wetland. Their disturbance and defaecation in the wetland caused measurable increases in nitrogen losses from the wetland during grazing events, but *E. coli* was not measured.

Natural wetlands, like constructed wetlands, can be both sources and sinks for *E. coli*. In general, they are likely to function in a similar way. When influent concentrations are high, they will reduce them, but when they are low they can increase them.

The functioning of natural wetlands in agricultural landscapes can be significantly compromised by drainage works around and within them which can cause short-circuiting of flow through the wetland (reducing residence time and contaminant removal) and/or in some cases leave wetlands disconnected or perched above main flow paths. This is a broader aspect of restoration on wetlands, of which stock exclusion as part.

The benefits of stock exclusion from a wetland are difficult to quantify, because stock access causes variable disturbance and sources, the removal efficiency of natural wetlands for *E. coli* is likely to be variable, and stock exclusion is just one aspect of wetland restoration.

Conditions of applicability

Exclusion of stock from wetlands could occur from all locations where stock current access wetlands. It is not clear how widespread such access is within Taranaki and Horizons regions, especially for smaller wetlands. Before human development, there were large areas of swamp on the Manawatū plains, and there may be remnants that would benefit from stock exclusion. Under current planning rules, stock are supposed to be excluded from natural wetlands. The current degree of compliance is unclear.

Overall removal

It is difficult to assess the overall reduction in loading from removing stock access to streams, due to the variable removal by wetlands, uncertainty about how stock access would improve removal by wetlands that are currently degraded, and the degree to which stock act as a source, and limited knowledge of where stock have access.

Ability to model

It would be difficult to model the effects of stock removal due to uncertainties as outlined above.

Overall assessment

Modelling of stock removal would not be practical.

However, stock are supposed to be removed from natural wetlands anyway, and potential reduction in microbial loading is one of the benefits of this measure.

8. Improved onsite domestic waste disposal

Description of the mitigation

On-site wastewater treatment (OSWT) refers to the decentralised and localised processing of sewage or wastewater generated by homes, businesses or facilities. Common on-site wastewater treatment systems include septic tanks, aerobic treatment units and advanced treatment technologies. These systems typically involve physical, biological and chemical processes to treat contaminants in the wastewater before discharging to the receiving environment typically via a land application system such as a soakage trench or subsurface drip irrigation scheme. When on-site treatment is combined with a disposal system it constitutes an on-site wastewater management system (OWMS).

Key literature on removal efficiency

Discharge of poorly treated wastewaters can act as sources of *E. coli* to streams. Septic tanks, widely used to provide primary treatment of household wastewaters before application to soil, make minimal difference to faecal bacterial loads discharged (compared with raw wastewater). Previous studies have shown a positive correlation between septic system density and pollution levels in streams (Sowah et al., 2014). Poorly designed disposal systems using land treatment can also create potential for leaching and runoff to impact surface and groundwaters.

The microbial removal efficacy of improved on-site wastewater treatment systems can vary based on the specific technology employed and the level of treatment achieved.

Close et al. (2020) provides modelling scenarios for OWMS (treatment + disposal) and microbial removal of 0.6 $\log_{10}$ for conventional primary treatment (settling tank + effluent disposal field), 1.6 $\log_{10}$ (secondary treatment + effluent disposal) and 2.6 $\log_{10}$ (advanced secondary treatment + effluent disposal). However, advanced on-site effluent treatment (OSET) systems often provide higher microbial removal rates. Trials of commercially available OSET systems reported removal of FIB greater than 2 $\log_{10}$ with some treatment systems (e.g. membrane bioreactors) achieving more than 5 $\log_{10}$ (Scholes, 2006). These systems may follow on from septic tanks or incorporate both primary and secondary treatment.

A range of mechanised small package treatment plants are available providing advanced secondary or tertiary treatment options. However, these have relatively high capital and operational costs and may not cope well with intermittent and fluctuating usage. In contrast, ecotechnologies such as horizontal subsurface flow constructed wetlands (HSSF CW) are a viable alternative for the secondary treatment of septic tank discharges prior to land application. They commonly achieve 99% reduction of faecal indicator bacteria (FIB) from fluctuating inflows (Stott et al., 2018). Where woodchip bioreactors (WB) were paired downstream of vertical-flow wetlands with sand media, *E. coli* removal improved to >4 $\log_{10}$ removal across the treatment system (Stott et al., 2018).

Conditions of applicability

Improved on-site wastewater treatment would be widely applicable to locations with conventional septic treatment systems. A reasonable estimate of the reduction in load could be obtained.

In general, onsite wastewater systems are widespread in regions lacking centralised sewer infrastructure, notably in small towns and settlements in rural and coastal areas. The subdivision of rural land is resulting in an increased prevalence of wastewater systems on smaller sections whilst existing systems may also be less effective due to inadequate maintenance. New systems are likely to be of a higher standard than legacy systems.

Problems with safe disposal of septic tank effluents are prevalent in areas with either poorly permeable soils leading to hydraulic failure, or highly permeable soils that enable rapid bypass flows. Challenges intensify in situations of shallow groundwaters limiting vadose zone attenuation, and high housing densities (e.g. clustering of on-site treatment systems) overwhelming the assimilation capacity of the aquifer. Inadequate separation distances from water supply wells or sensitive waterways exacerbate the problems.

Horizon Regional Council have provided guidance for the design of OSWT systems, land application disposal and the assessment of effects from OWMS in the Manawatu-Wanganui Region (Barnett and Ormiston, 2007). Rules in the One Plan require better systems for new sites and for when old systems are upgraded and set minimum standards for performance of existing systems.

Poor treatment from existing systems could be improved with adequate servicing and improved operation management as end users have little knowledge about how OSWT systems operate (Dakers et al., 2016). Failing systems could also be improved with the provision of additional treatment which can increase the sustainable acceptance rate of soil absorption fields thereby reducing the incidence of hydraulic failure and increasing operational lifetime of soil infiltration systems.

Secondary and particularly advanced secondary OSWT systems are effective in removing *E. coli*. But, despite the high reduction rates achieved by some OSWT systems, the *E. coli* levels they attain may fall short of regulatory recommendations. Auckland Council GD06 advised a limit of $< 10^4$ *E.coli* /100mL for secondary treatment from OSWT systems (Chen and Silyn-Roberts, 2021). However, a recent survey of OSWT systems in the Canterbury region found that secondary treatment systems typically failed this guideline with average concentrations of 10^5 /100mL in secondary treated effluent (Humphries et al., 2023).

Overall removal

Properly managed and maintained systems ensure a higher level of treatment of point-diffuse discharges, thereby preventing contamination of groundwaters and nearby surface waters. The degree of improvement of water quality will depend on the density and condition of on-site facilities. It is suspected that significant improvements in water quality could be achieved in some locations where there is a high density of facilities with poor condition. For example, local river contamination associated with on-site disposal has been noted for Urenui, prompting actions for formal wastewater collection and disposal[2]. In sparsely populated areas, the improvements would be insignificant. In townships, there is likely to be central wastewater treatment in most cases. Therefore, the overall improvement in swimmability for the region overall is likely to be modest.

Ability to model

GIS based tools developed as a planning tool for new subdivisions in Canterbury can provide a preliminary assessment of OWMS discharge impacts to groundwaters and surface water catchment condition (Trewartha, 2023).

Microbial risks associated with multiple land use practices such as on-site wastewater treatment systems for domestic or community size have been modelled by (Close et al., 2020). The authors simulated risk assessment for OWMS (treatment and land application) for three types of OWMS in

[2] Urenui/Onaero wastewater project (npdc.govt.nz)

New Zealand with a focus on transport of microbes including *E. coli* to groundwaters for drinking water supply risk.

Modelling for impacts on surface water would require information on the number, location and type (or density) of onsite wastewater management systems, and estimates of their condition. A key uncertainty would be around loading that reaches surface watercourses. Some initial analysis with conservative assumptions would be appropriate to gauge risks. The baseline concentration model underlying CLUES does not account for the impacts of septic tanks, so it may be necessary to first add the sources and estimate increases in concentration based on load increases, and then remove the load to compare with and without discharges from on-site wastewater.

Overall assessment

It is appropriate to quantify expected inputs of *E. coli* from small towns and settlements using on-site disposal because many of these will be in lowland areas and may have a disproportionate effect on water quality at downstream sites where monitoring is commonly carried out. This would take the form of a preliminary loading analysis based on conservative assumptions, and would require information on the density and type of management systems for on-site wastewater disposal.

9. Diffuse urban source management (source control and stormwater treatment)

Description of the mitigation

A variety of methods can be used to reduce diffuse microbial contaminant sources in urban areas. The methods range from reducing faecal sources such as animal faeces (from pets or birds), reducing sewage cross-connections and overflows, and treating stormwater through devices such as filters and wetlands.

Key literature on removal efficiency

Clary et al. (2020) summarised *E. coli* concentration removal fractions for several types of urban treatment devices, with a high removal of about 85% by wetlands, about moderate to good 50% removal for bioretention, filter media and detention basins, and poor removal for grassed swales. Note that these are based on median concentrations, not on loads; for urban areas in particular, a large part of loading can occur during flashy runoff events.

Urban cross-connections can be removed effectively through management of the wastewater network, and sanitary surveys to identify and then remove illegal connections. Overflows can be managed through improvements to sewerage infrastructure and management of stormwater and groundwater ingress.

Overall removal (including consideration of flow paths and technical suitability).

Urban areas are only a small proportion (1% or less) of the land use in Taranaki or Horizons Regions. In previous modelling, a microbial yield roughly 10 times that of pastoral areas was used for urban areas, based on literature vales for urban areas and calibrated values for pasture, suggesting a significant overall contribution to microbial loading from urban areas. Concentrations increase significantly through Palmerston North, suggesting that urban areas do have an impact. Improvements are likely to be local to Palmerston North, because observed concentrations return to levels typical of intensive pasture downstream of Palmerston North. There may be high value placed on this, however.

Stormwater treatment devices are difficult to retrofit in an established urban area such as Palmerston North. Hence the good removal that can be achieved by wetlands (and their potential to intercept a range of flow paths) would have limited overall effect due to the low suitability. Other devices would also be difficult to implement on a widespread basis, or would have low removal.

There was not readily-available documentation the degree of cross-connections in Palmerston North, so it is difficult to ascertain its effect on urban microbial loads. Similarly, the degree of overflows is unclear.

There is considerable uncertainty about the dominant source of microbial contaminants in Palmerston North. Council staff have indicated that they are investigating microbial contamination in the Manganui Stream, which is suspected to be associated with a dominant microbial source.

Ability to model

The effect of urban wetlands could be modelled, albeit in an approximate fashion. However, the overall effect on microbial contamination would remain uncertain because significant sources that bypass wetlands (such as discharges into streams) would not be represented and the importance of such other sources is not known.

The effect of diffuse urban source control would be difficult to model because the sources are not quantified well, and the efficacy of the methods is not known well.

Overall assessment

Removing urban microbial sources could have a high influence locally, expecially around Palmerston North, but would have little impact on the region-total swimmability.

If urban contamination is considered important locally, then sources of the contamination should be investigated further before embarking on further modelling of mitigation measures. It would be of interest, however, to conduct some preliminary scenarios of urban yield reduction to gauge the impact, with the proviso that there is a considerable uncertainty in the urban component of the model.

10. Woodchip bioreactors

Description of the mitigation

Designed primarily to treat nitrate-laden waters, denitrifying bioreactors redirect drainage waters into and through a buried bed/trench of porous high-carbon materials typically woodchips (Christianson et al., 2021). Woodchips serve as a carbon and energy source fostering microorganism growth and creating anoxic conditions. This encourages nitrate conversion to nitrogen gas by microbial denitrification and anammox processes (Rambags et al., 2019a). Buried walls of comparable high-carbon materials can also be used to intercept shallow groundwater flows and seeps through flow rates tend to be lower than that of beds (Schipper et al., 2010). Woodchip denitrification beds have also been placed in open surface drains (Burbery and Abraham, 2022; Christianson et al., 2017). Designed primarily to treat nitrate-laden waters, denitrifying bioreactors redirect drainage waters into and through a buried bed/trench of porous high-carbon materials typically woodchips (Christianson et al., 2021). Woodchips serve as a carbon and energy source fostering microorganism growth and creating anoxic conditions. This encourages nitrate conversion to nitrogen gas by microbial denitrification and anammox processes (Rambags et al., 2019a). Buried walls of comparable high-carbon materials can also be used to intercept shallow groundwater flows and seeps through flow rates tend to be lower than that of beds (Schipper et al., 2010). Woodchip denitrification beds have also been placed in open surface drains (Christianson et al., 2021).

Key literature on removal efficiency

Bioreactors beds are designed to intercept and treat shallow concentrated subsurface flow paths such as tile drainage and interflow/shallow groundwaters and seepage likely to re-emerge on farms. Microbial contaminants leach through the soil horizon and move into tile drainage and subsurface flows mainly by infiltration via soil macro-pores (Hruby et al., 2016). While denitrifying woodchip bioreactors (WBs) are effective in removing nitrate from tile drainage waters, little consideration has been given to the fate of faecal microbial contaminants.

A few studies have assessed the microbial removal efficacy of WBs treating highly contaminated sources such as wastewaters. Rambags et al. (2016) found effective microbial removal in a full-scale WB treating partially-treated wastewater, with removal of 2.9 $\log_{10}$ *E. coli* and 3.9 $\log_{10}$ FRNA phage despite highly variable inflow concentrations. Most of the *E. coli* load reduction occurred within the first meter of the reactor distance (1.4 $\log_{10}$) indicating the potential capacity of WBs to manage higher microbial loads with shorter HRTs. Clogging and siltation will likely reduce bioreactor performance via flow restriction and development of preferential flow paths through the bioreactor. However, similar removal performance was reported for mature WBs (8 yr old systems) demonstrating the potential longevity of *E. coli* removal (Rambags et al., 2019b). Soupir et al. (2018) observed *E. coli* removal ranging from 75% to 96% in woodchip laboratory columns treated with synthetic agricultural drainage, with higher removal noted at elevated temperatures. (Mardani et al., 2020) reported *E. coli* removal rates from laboratory columns varying from 49% to 77% with greater removal observed under wet-dry flow conditions compared to steady flow conditions.

Tanner et al. (2023a) assessed that WBs sized for moderate nitrate-N reduction would generally demonstrate approximately 90% removal for *E. coli* for the flow path.

Field-scale performance of WBs has been tested in New Zealand primarily for interception of tile drainage water focussing on nitrate-N removal (Rivas et al., 2020). There are no field based studies on bioreactors targeting the removal of faecal microbial contaminants from agricultural tile drainage.

Conditions of applicability

WB are best applied as beds that intercept artificial drainage systems. They are vulnerable to clogging by fine sediments where they intercept surface flows, so tile drainage is preferable. Denitrification walls have potential to intercept and treat shallow subsurface flow paths where subsurface flow is within 3 m of the ground surface and are critical source pathways. Situations where they are likely to be of significant benefit are where subsurface flow is confined by low permeability layers beneath (e.g. clay or bedrock).

Application sites for WB beds in the Taranaki and Horizon regions are likely to be flatter sites with heavy subsoils, where tile drainage is an important flow path for contaminants (Tanner et al., 2023a). Currently they are primarily used for nitrate-N reduction and application for removal of *E. coli* is low.

WBs demonstrate a reasonably effective removal of microbial contaminants when there is a high concentration in the inflow (Rambags et al., 2019b). Typically, subsurface drainage flow is less contaminated than surface runoff with concentrations of *E. coli* varying by up to three orders of magnitude (Schreiber et al., 2015). The relevance of WB application to Taranaki and Horizon situations would be where faecal microbe loads may be elevated in drainage and subsurface flows for example where wastewater effluent is irrigated and manure/sludge applied and in soils prone to bypass flow respectively.

Overall removal

WBs can serve as effective mitigation tools for addressing microbial transport in areas with subsurface (tile) drainage in addition to reducing nitrate-N losses.

Ability to model

Modelling of WB performance has focused predominantly on nitrate removal rates resulting in a diverse range of reported models. (Jang et al., 2018) used an empirical model to estimate bioreactor efficiency for nutrient reduction. Microbial removal models have been developed specifically for stormwater biofilters and could be adapted for WBs using key processes and factors influencing microbial removal as demonstrated by (Shen et al., 2018). Otherwise expected removal rates for a given application intensity could be applied to tile-drainage areas of catchments.

Overall assessment

Woodchip bioreactors, exhibit potential as a mitigation tool for reducing faecal microbial losses from tile-drained areas in agricultural landscapes, especially hotspots where wastewater effluents are irrigated. Although promising, their application to reduce microbial loads lacks extensive field-scale validation, and their overall applicability is low unless reduction of nitrate-N is also a key focus.

11. Bridge stream crossings

Description of the mitigation

This involves placing a bridge over a stream so that animals and farm vehicles do not walk through the stream. This mitigation is very pertinent for a dairy farm where the cows may need to cross a stream up to 4 times per day to get to the milking shed. However, the mitigation can also apply to sheep & beef farm although the mobs of animals are not moved as often and dairy farms and hence will be less effective overall.

Key Literature

Davies-Colley et al. (2004) measured the impact of a 246 dairy cow herd crossing the Sherry River and a large spike in downstream *E. coli* concentrations when the herd crossed as a large mob on the way to milking. On the return crossing when the cows were considerably spread out the concentrations were lower, but for a longer period of time and hence the overall impact was similar. Muirhead et al. (2011) developed a Monte Carlo simulation model of a single dairy farm and predicted that – assuming the stream crossing was used for only 25% of days during the milking season – that the stream crossings would have a large impact on the 95th percentile of concentrations but not on the median downstream concentration. Muirhead (2015) then used the outputs from the Monte Carlo simulation model to develop an *E. coli* risk index and validated this against the data from the 5 dairying focus catchments. This analysis provided some evidence that changes in dairy farm management would lead to reductions in *E. coli* concentrations in a catchment. (Muirhead and Doole, 2017) adapted the Monte Carlo modelling approach from (Muirhead et al., 2011) to a sheep & beef farm systems in the Gisborne region and then developed a simplified spreadsheet model that can estimate the relative losses from a farm from: (1) full stream fencing, (2) stream cattle fencing only, (3) bridging stream crossings, (4) changing stocking ratios and (5) land use conversion to forestry. The Gisborne study however, estimated that bridging stream crossings was the least cost effective of the 5 mitigation options investigated.

Conditions of Applicability

Requires that the streams are already fully fenced off. If there is just a bridge for vehicle traffic and no stream fencing, then this will have a minimum effect.

Overall removal

Best expert opinion is <5% at the catchment scale.

Ability to model

This mitigation is difficult to model as its effectiveness is dependent on: (1) the amount of stream fencing on a farm, (2) the number of stream crossings on a farm and (3) the frequency of animal mob movements at these crossings. These inputs can be estimated by a farmer for an individual farm (Muirhead et al. 2011; Muirhead & Doole, 2017), but would be highly uncertain when applied to multiple farms in a catchment model.

Overall assessment

Low priority for modelling. Remains a common-sense mitigation measure supported by data for crossings that are used regularly.

12. Deferred and low rate effluent irrigation

Description of the mitigation

The irrigation of farm dairy effluent (FDE) by low rate (<10mm/h) irrigation systems and using a storage pond so the effluent is only applied with there is a suitable soil water deficit.

Key Literature on removal efficiency

The science of FDE management in NZ is very mature and this mitigation is the recommended method by DairyNZ. This knowledge has been synthesised into industry guidance that can be found at https://www.dairynz.co.nz/media/2iybfjqv/fde-planning-the-right-system-for-your-farm-dnz40-118-2023.pdf. Deferred and low-rate irrigation can achieve substantial reductions in losses from the soil or in overland flow compared with high-rate irrigation without storage (Muirhead et al., 2010; Muirhead et al., 2011). The reduction depends on the amount of storage, irrigation rate, and soil conditions, and can reduce the loading from dairy effluent to very low levels compared with other dairy farm sources.

Conditions of Applicability

This mitigation applies only to dairy farms. There is some flexibility in types of systems but when matched appropriately to the irrigated soil type, the different systems will achieve the same outcome of very little losses to the wider environment. It is likely to have most impact when there are soils that have little water storage and have high potential for flow bypass.

Overall removal

When managed correctly the environmental losses from one of these FDE management systems will be insignificant compared to losses from the grazed pastures of a farm. Deferred and low-rate application management is already practiced widely in Horizons.

Ability to model

Removal of direct discharges to streams in Taranaki has already been applied in the previous modelling analyses by NIWA. This assumed 95% reduction in loads discharged to streams, compared with direct discharge, and the reduction had a significant effect on loads and microbial concentrations. In that work, little emphasis was placed on the method of irrigation. A reduction of 95% could be more relevant to low-rate systems, but it is not clear what type of disposal conditions TRC would use. There are modelling techniques available to estimate losses from the soil or surface runoff, although modelling of transport to the stream is less certain.

Overall assessment

Low priority. In the Horizons region this FDE management system has been encouraged so almost all farmers will already be doing this – so little opportunity for improvement in water quality at the catchment scale as the mitigation is already adopted.

In the Taranaki region there is opportunity to encourage adoption of this mitigation approach on farms as there is still a large number of dairy farms still directly discharging FDE from ponds. The effect of removal of pond effluent has already been included in the stage 2 modelling work (Semadeni-Davies et al., 2023a). There could be existing systems that don't use low-rate or deferred irrigation, but in Taranaki this is likely to be of less consequence that in Horizons because there are generally well-drained soils with little potential for bypass flow or subsurface drainage. Some simulation of low-rate or deferred irrigation systems in Taranaki for locations with poorly-drained soils or artificial drainage, may be warranted to evaluate risks in those locations.

13. Enhanced dairy effluent ponds

Description of the mitigation

These are advanced or high-rate effluent ponds that provide a higher level of treatment than traditional 2-pond systems on dairy farms.

Key Literature on removal efficiency

Craggs et al. (2004) demonstrated that the advanced pond system (APS) could significantly reduce the *E. coli* concentrations in the discharge water from 70,000 to 918 MPN $100mL^{-1}$ for the APS and traditional 2-pond systems, respectively. Muirhead et al. (2011) included the APS in the Monte Carlo model and demonstrated that converting a 2-pond discharge to an APS system or the deferred-low rate irrigation system (as above) would result in similar downstream concentrations.

Conditions of Applicability

Applies only to dairy farms where FDE is being discharged directly to a drain or stream.

Overall removal

Significant removal of *E. coli*.

Ability to model

Could be achieved in CLUES by using the same reduction value as for the deferred-low rate irrigation system.

Overall assessment

Low priority. While these ponds have been shown to significantly reduce the *E. coli* concentration in the discharge the discharge still contains high levels of nutrients. So, any form of direct discharge is still losing nutrients to water, which will need to be replaced with fertilizer inputs, which is inefficient from a farming systems perspective. A better overall outcome for the farm and the environment is adopting a differed irrigation system as above and as recommended by the Dairy Industry.

14. Off-pasture confinement

Description of the mitigation

Keeping animals in an animal housing system all year round.

Key literature on removal efficiency

There is very little literature on *E. coli* impacts of this mitigation. The idea is that all of the dung is contain in storage where it can be applied to land as a fertilizer during summer when there is minimal risk of runoff events. The challenge for modelling is that we do not know the relative *E. coli* losses from a year-round grazing system versus manure spreading over the summer period. This runoff risk is complicated by the fact the manure spreading systems can result in a huge surface area exposed to rainfall impact compared to discreet dung pats. Furthermore, the increased capital cost of building the animal housing facility can result in increased intensification of the operation and hence a larger volume of dung will be generated than a grazed system.

Conditions of Applicability

Theoretically could apply to any animal species. However, in a NZ context is likely to only be an option for some dairy farms. Due to the high capital cost of these systems, they would be uneconomic for sheep or beef systems.

Overall removal

Unknown.

Ability to model

None.

Overall assessment

Low priority. These systems rely on mechanical harvesting of feed and bringing the feed to the animals which is much less efficient than the animals harvesting the feed directly from the pasture. This mitigation would be pushing NZ farms to more intensive systems with higher environmental footprint which is the opposite direction to which the market wants farmers to be moving.

15. Restricted winter grazing

Description of the mitigation

This is limiting the time animals are on pasture (usually to 8 hours/day) during the winter period to reduce the amount of urine deposited on wet soils.

Key literature on removal efficiency

There is no literature on reduction of *E. coli* for this mitigation, but some work out of Massey on nitrogen effects. This mitigation suffers from the same modelling issues as for fully housed systems where the runoff risk is complicated by the fact the manure spreading systems can result in a huge surface area exposed to rainfall impact compared to discreet dung pats. Furthermore, the increased capital cost of building the animal housing facility can result in increased intensification of the operation and hence a larger volume of dung will be generated than a grazed system.

Conditions of Applicability

Theoretically could apply to any animal species. However, in a NZ context is likely to only be an option for some dairy farms. Due to the high capital cost of these systems, they would be uneconomic for sheep or beef systems.

Overall removal

Unknown.

Ability to model

None.

Overall assessment

Low priority.

16. Reduce stocking rate

Description of the mitigation

This mitigation entails having fewer animals on a farm. This outcome could be achieved by two different options: (1) keeping the pasture area the same and reducing fertilizer inputs and/or imported feed and running less animals or (2) by removing some of the pasture area and retiring into trees or converting to cropping or horticulture systems. Note that option 2 can be achieved at a catchment scale by reducing the number of pasture farms and replacing these farms with horticulture or forestry.

Key literature on removal efficiency

There is no literature on *E. coli* impacts of this mitigation at a farm-scale. There is only one study on the effect of stocking rate on *E. coli* runoff concentrations which found that sheep camping areas with 4x stocking rate of typical grazing resulted in a 6x increase in runoff concentrations (Muirhead 2023). We cannot extrapolate this result to what would happen if stocking rates were decreased below typical grazing.

Conditions of Applicability

Could be applied to any pasture-based farm. Option 2 could be modelled in CLUES by assuming an area of pasture converted to forestry for carbon farming.

Overall removal

Will depend on the proportion of land converted to forestry.

Ability to model

Option 1 of reducing the number of animals grazing the pasture areas cannot be modelled. However, this option is unlikely to be adopted on pastoral farms as it will reduce the profitability of the farm. Option 2 is a more likely approach as it can provide an alternative income stream to offset the reduction in area in pasture farming. Option 2 could be done in CLUES using existing model parameters and changing the land-use distribution.

Overall assessment

High priority. Land use change is already happening and likely to accelerate in the future.

17. Stock exclusion from critical source area exclusion

Description of the mitigation

Removing animals from the wettest parts of the land that generates most of the overland flow.

Key Literature

Nothing published for pasture areas, but AgResearch has one study underway with results expected to be published in 2025. There is a publication when applied to winter forage crop grazing – see information in the fodder crop restriction mitigation section.

Conditions of Applicability

Can be applied to any pasture area but unlike to be applied to flat land. This would apply best to rolling to steep land where the wet areas are smaller and easier to identify in the landscape.

Overall removal

Unknown.

Ability to model

None.

Overall assessment

There is low importance on modelling this as the science is not yet developed.

It may be desired to promote critical source area management in the absence of a strong quantitative foundation, as a common-sense approach.

18. Strategic grazing of forage crops

Description of the mitigation

Management of grazing of fodder-crop/forage areas to reduce runoff risks, for example by removing animals from wet parts of fodder-crop/forage areas.

Key Literature

Monaghan et al. (2017) found that *E. coli* losses from strategic grazing of cow-grazed forage crop loads could reduce losses from the grazed area by about 50% compared with standard forage crop grazing. Assuming 10% of the farm is planted in forage crops this would be 5% reduction at the farm-scale. Losses of sediment and nutrients were greatly reduced by re-establishment of pasture in the following year, but surprisingly losses of *E. coli* increased, which was attributed to residual effects from the fodder crop trial. Recently-published work on sheep grazed forage crops measured a 63% reduction in runoff loads by protecting critical source areas (Ghimire et al., 2024). These experiments were conducted in South Otago so may not be directly applicable to the soils, farm systems and rainfall patterns in Taranaki and Horizons.

Conditions of Applicability

This mitigation only applies to forage crops that are grazed through winter. In Horizons and Taranaki one of the main supplementary feed crops is maize which is harvested, not grazed (although occasional grazing of maize is may occur in dry conditions with lower runoff risk). Hence fodder crop grazing restriction is not relevant to the feed crop of maize. However, there are also significant areas of winter grazing on fodder crops such as brassicas and forage beet in the Taranaki and Horizons areas[3]. Stats NZ figures show 4,159 ha of forage brassicas in Taranaki in 2022 and 14 883 ha in Horizons area (compared with 4,375 ha and 5195 ha for maize silage in Taranaki and Horizons respectively). There is about 270,000 ha of dairy and beef cattle in Taranaki, so forage brassicas are about 1.5% of those intensive pasture areas, while for Horizons the area of dairy and beef cattle is approximately 486,000 ha, so forage brassicas are about 3% of those intensive pasture areas.

Overall removal

Uncertain as the loss rates from grazed forage crops in the area are not known.

Ability to model

A percent reduction could be estimated (using results mentioned above) if we are provided with a local input of the proportion of farms that are winter grazed forage crops. The reductions are uncertain due to the very small number of trials and no trials in the North Island. We are also unsure of the losses from standard forage grazing without restrictions.

Overall assessment

Moderate priority for modelling.

19. Alternative deer wallows

Description of the mitigation

Providing a wallow for deer to “play in” that is not connected to a stream.

Key Literature

Natural deer behaviour will result in the creation of wallows which are basically a large mud bath and will often be formed in or near streams (McDowell, 2007). By fencing off the deer from the stream and then creating a “safe wallow” that is not connected to the stream can significantly reduce the impact on a stream (McDowell, 2007). However, the overall impact of this mitigation at a farm or catchment scale has not been assessed.

Conditions of Applicability

Applies only to deer farms.

[3] https://figure.nz.

Overall removal

Unknown. But could be estimated at 10 to 25% at the farm scale.

Ability to model

A reduction parameter could be applied to deer farmed land in CLUES.

Overall assessment

Low priority for modelling, given that deer farms are only about 2-3% of the land area in Taranaki and Horizons regions (Semadeni-Davies et al., 2023b). It is recommended that alternative wallows should still be promoted on deer farms to control local stream contamination, based on evidence from other regions.

20. Feral animal control

Description of the mitigation

This is the control of possums, deer, pigs and other animals in native forest areas.

Key Literature

This mitigation came out of the He Waka Eke Noa process where it was recognized that wild animals in forests consume biomass and therefore, controlling the animal populations could result in increased carbon sequestration. The assumption is that less animals in native forests should reduce *E. coli* (and other contaminants) from streams. There are no publications to support these assumptions.

Conditions of Applicability

Would only apply to native forested areas.

Overall removal

Unknown.

Ability to model

The effect of this mitigation is likely to have been covered by the “low reference source scenario” conducted in stage two of the modelling by NIWA.

Overall assessment

Low for further modelling.

Comments on difficulty of modelling mitigation effect on NPS-FM attributes using budget models such as CLUES.

Our ability to provide accurate assessments of the effectiveness of *E. coli* mitigations is severely limited by a lack of fundamental data and knowledge of microbial dynamics in catchments. This is further complicated by the complexity of the current water quality standards applied in NZ and how these relate to catchment scale modelling. Here we discuss several of these complications.

The NZ microbial water quality standards (NPSFM, New Zealand Government, 2023) are based on a complex assessment that requires the calculation of four sample statistics (median, %>260, %>540 and 95th %ile) that are then compared to a table of standards with gradings from A to E. The final grading for a site is based on the worst grading of the 4 metrics. In relation to median concentrations, there is only one threshold value (130 *E. coli* 100mL^{-1}), at the break between C and D/E grades. Hence other metrics are needed to determine if a site is in A or B bands, if the median is <130 *E. coli* 100mL^{-1}. This means that it is important to understand the shape of the probability distribution of concentrations, not just the median concentration. If the median value is low and the site fails due to a high 95th %ile, this indicates that the issue is caused by a sporadic *E. coli* source that is impacting on the river only occasionally. However, if the site is failing due to a high median, then this indicates that there is a relatively constant source of *E. coli* in the river. In these different situations one may want to target different mitigation options to address potentially different sources of *E. coli*. Furthermore, the current modelling approach using CLUES operates on an annual average basis and we assume that any percent reduction effectiveness of mitigations will apply equally across the range of concentrations. In practice this is unlikely to be true, but we don't have any better data to approach this question differently. Hence, for future modelling it would be desirable to have an approach that models time series of concentrations, or the probability distribution.

The second complication of modelling the impacts of mitigation options on microbial water quality is the difference between stormflow and baseflow conditions in rivers. During stormflow there is a large increase in total streamflow and the associated contaminant loads transported in the stream network, and this effect is greater for *E. coli* loads than for other contaminants (Ballantine and Davies-Colley, 2013; Davies-Colley et al., 2008). This means there is a much greater load of *E. coli* transported in the river during stormflow events. The total load of *E. coli* in a river has an impact on the waterbody (lake, estuary or ocean) that the river discharges into as the large pulses of storm water have to be diluted and dispersed over time. However, the river microbial water quality metrics are based on concentrations, not loads. Because a river spends more days per year in a baseflow state than stormflow state, *E. coli* concentrations during baseflow conditions have a large impact on the river water quality metrics – particularly the median concentration. For the same size of the annual load of *E. coli* discharged to a river, sources that occur during baseflow conditions (such as point source discharges and farm dairy effluent) will have a disproportionately large impact on the microbial river water quality metrics compared with sources that occur during infrequent storm events, and analysis of load reductions may underestimate the importance for microbial metrics of sources that occur during baseflow. It is unknown whether runoff from the land during storm events will have an impact on *E. coli* concentrations under all flow conditions, including baseflow. As yet we do not yet have a good understanding of the extent to which *E. coli* that enter a stream during storm flows impact on the water quality guideline metrics i.e. do all the *E. coli* that enter a stream in runoff flow all the way to the river mouth and therefore only impact on storm flows? Do these stormflow conditions only impact on the 95th percentile values in the monitoring datasets? Or do some of the runoff *E. coli* get trapped in the stream sediments and subsequently bleed out during baseflow conditions, thus contributing to elevated stream median concentrations (Davies-Colley et al., 2008;

Drummond et al., 2022; Pachepsky et al., 2018; RJ Wilkinson et al., 2011). Our understanding of the dynamics of *E. coli* concentrations in rivers is poor and this severely limits our ability to model and/or predict changes in microbial water quality that might occur in response to changes in land use or management (Oliver et al., 2016).

A third complication is the effect of scale on mitigation effectiveness. We will discuss this using the example of direct animal defecation into a stream. If we assume that a farm will have a large number of paddocks that the animals move around, and that only 25% of the paddocks have unfenced stream access. At a single farm-scale, we have shown that this direct animal access has a large impact on the 95th percentile value but no impact on the median downstream concentration Muirhead et al. (2011). However, if we expand to a larger catchment with 40 farms, all with 25% of paddocks unfenced from the river, then it is likely that there will be direct inputs from approximately 10 farms every day. Therefore, these direct inputs could increase the median concentrations at a catchment scale and have less impact on the 95th percentile as there is greater dilution at the catchment-scale.

A fourth complication is the importance of different flow paths, and representation of these within models. The assessment of mitigation measures above shows that some devices can treat runoff from one flow path effectively, but not for other flow paths. This suggests that understanding of flow paths, the loading associated with the different flow paths, and how this varies with land use and landscape conditions, is desirable to accurately predict the effect of mitigation measures. This is an active area of model development, but is hampered by limited data and ability to represent microbial dynamics.

When all of these complications are added together, means that it is difficult to assess improvements in NOF attributes due to implementation using a catchment model such as CLUES, which in most cases assessed sources based on land use and the applies source reduction factors for different mitigations. One advance on the modelling would quantify different flow paths and sources (not only in terms of hydrology, but also in terms of contaminants). A further advance would be to develop dynamic catchment models to assess mitigations, although such modelling is challenging.

Synthesis and prioritisation

The preceding assessment of mitigations options is summarised in abbreviated form in Table 1. We distinguish between the removal efficiency for the intended source and flow path in situations that are suitable for the mitigation, and the overall removal taking into account the extent and importance of the source, the degree to which the measure could be implemented. The recommendation for modelling takes into account the overall potential for improvement and the feasibility of modelling.

The table is accompanied with several notes, which in some cases summarise the reasoning for a particular grading, and other points such as the need for gaining further information before embarking on modelling.

Table 1. Summary of potential for load removal and priority for modelling of each mitigation measure. Notes for the table are on the next page.

	Mitigation	Potential for load removal for local source	Potential for load removal overall[1]	Priority for modelling
1	Additional SLUI land retirement	High	Low	High
2	Additional stock exclusion from streams	Moderate	Low	High
3	Additional improvements to wastewater treatment	?	Low	Low[4]
4	Detainment bunds	Moderate	Low-Moderate	Moderate[12]
5	Vegetated riparian buffers	Moderate	Low-moderate (GFS) Very low (PRB)[9] Low-moderate (combined)	Moderate (GFS)[7] Low (PRB)[8]
6	Constructed wetlands	Negative to high	Negative to moderate	Low[2]
7	Livestock exclusion from natural wetlands[10]	?	?	Low[9]
8	Improved onsite domestic waste disposal	Moderate to high	Very low or low[3]	Moderate
9	Diffuse urban source management	?	Very low[3]	Low-Moderate[5]
10	Woodchip bioreactors	Moderate to high	Very low	Low
11	Bridge stream crossings[10]	High	Very low	Low
12	Deferred and low rate effluent irrigation[10]	High	Very low	Low
13	Enhanced dairy effluent ponds	High	Very low	Low
14	Off-pasture confinement	?	?	Low
15	Restricted winter grazing	?	?	Low
16	Reduce stocking rate	Variable[11]	Low	High
17	Stock exclusion from critical source areas[10]	?	?	Low
18	Strategic grazing of fodder crop areas[10]	Good	?	Moderate
19	Alternative deer wallows[10]	High	Very low[6]	Low
20	Feral animal control	?	?	Low

Notes for Table 1:

[0]Anticipated local removal for the intended flow path or source and under conditions that are suitable for the mitigation, taking into account the removal efficiency, flow paths treated, and conditions of applicability. Removals are classified as very low (<10%), low (10 to 25%), moderate (25-50%), good (50-80%) and high (>80%), where the numbers are coarse indications rather than intended to be used directly in modelling.
[1]Anticipated region-wide implications for improving health metrics, taking into account the removal efficiency, flow paths treated, importance of the source, and conditions of applicability.
[2]Difficult to determine reliable attenuation estimates due to variability in performance. Await further monitoring data.
[3]High in some localised areas
[4]Low overall impact likely. May be desirable to include for localised impacts and political reasons.
[5]Recommend investigation of sources before implementing controls or modelling. Conduct some preliminary assessment of reduction of urban stormwater by reduction the urban diffuse yield.
[6]Limited extent of deer land use.
[7]Recommend spatial assessment of suitability of GBS before modelling
[8]Included in additional fencing options.
[9]Low additional removal beyond the effect of stock removal.
[10]Not practical to model this mitigation, but the measure should be promoted due to potential benefits
[11]Depends on degree of stock reduction, and whether there is a change in land use intensity.
[12]First assess the potential for suitable locations for bunds in the Taranaki and Horizons regions

Recommendations relating to progression to next stage of modelling

Based on the analysis above, we recommend that modelling be progresses for:

- Additional SLUI land retirement
- Additional stock exclusion from streams
- Detainment bunds, but contingent on assessment of suitable locations
- Moderate for grass buffer strip, but contingent on assessment of suitable locations and practicability
- Preliminary risk assessment of improved onsite domestic waste disposal, especially at locations where there is a moderate concentration of legacy systems. First establish the extent and condition of such systems.
- Reduced stocking rate (through land retirement or non-animal land use).

Some mitigation measures are encouraged, despite difficulty with quantitative understanding of the benefits, as 'common-sense' measures or because of evidence of their impact in other locations, or because they are included in recent regulations. These include:

- Deferred or low-rate dairy effluent irrigation in locations with risky soils
- Bridged stream crossing regulations
- Stock exclusion from wetlands and critical source areas
- Provision of alternative deer wallows.
- Strategic grazing of forage crops

Some further investigations (apart from those mentioned in relation to providing information for modelling) include:

- Assessment of causes of observed increases in river concentrations in Palmerston North

Advancement of microbial modelling to include explicit consideration of flow pathways would be desirable to enable refined and more nuanced assessment of the effects of mitigation measures.

Also, research into development and testing of dynamic microbial models should be advanced to enable better assessment of concentrations across the distribution of temporal scales.

The analysis in this report did not include consideration of costs or acceptability of mitigation measures, and co-benefits of mitigation measures. Implicitly, we have considered some of these aspects by, for example, calling for assessment of feasibility of grass buffer strips, and encouraging some mitigation measures that would have little overall impact but could have high local or political importance.

A final comment is that while individual mitigation measures may have limited or local effects, collectively a package of mitigation measures applied in appropriate conditions could improve NOF metrics.

References

Audet, J., Zak, D., Bidstrup, J. and Hoffmann, C.C. 2020. Nitrogen and phosphorus retention in Danish restored wetlands. Ambio 49, 324-336.

Bai, J., Shen, Z. and Yan, T. 2016. Effectiveness of vegetative filter strips in abating fecal coliform based on modified soil and water assessment tool. International Journal of Environmental Science and Technology 13(7), 1723-1730.

Ballantine, D. and Davies-Colley, R.J. 2013. Nitrogen, phosphorus and E. coli loads in the Sherry River, New Zealand. New Zealand Journal of Marine and Freshwater Research 47(4), 529-547.

Barnett, H. and Ormiston, A.W. 2007 Manual for On-Site wastewater design and management: Technical Report to support policy development. . Council, H.R. (ed), p. 95.

Beck, W.J., Schoonover, J.E., Williard, K.W.J. and Zaczek, J.J. 2013. Attenuation and persistence of fecal indicator bacteria in vegetated buffer strips in Southern Illinois, USA. Agroforestry Systems 87(6), 1405-1419.

Boutilier, L., Jamieson, R., Gordon, R. and Lake, C. 2011. Modeling E.coli fate and transport in treatment wetlands using the water quality analysis and simulation program. Journal of Environmental Science and Health Part A 46, 680–691.

Burbery, L. and Abraham, P. 2022 Results from an in-stream woodchip denitrifying bioreactor field trial in south Canterbury, p. 10, Farmed Landscapes Research Centre, Massey University, Palmerston North, New Zealand.

Chen, Z. and Silyn-Roberts, G. 2021 On-site wastewater management in the Auckland Region, p. 363.

Christianson, L.E., Collick, A.S., Bryant, R.B., Rosen, T., Bock, E.M., Allen, A.L., Kleinman, P.J.A., May, E.B., Buda, A.R., Robinson, J., Folmar, G.J. and Easton, Z.M. 2017. Enhanced Denitrification Bioreactors Hold Promise for Mid-Atlantic Ditch Drainage. Agricultural & Environmental Letters 2(1), 170032.

Christianson, L.E., Cooke, R.A., Hay, C., Helmers, M., Feyereisen, G.W., Ranaivoson, A.Z., McMaine, J., McDaniel, R.L., Rosen, T., Pluer, W.T., Schipper, L.A., Dougherty, H.L., Robinson, R., Layden, I., Irvine-Brown, S., Manca, F., Dhaese, K., Nelissen, V. and Ahnen, M.v. 2021. Effectiveness of Denitrifying Bioreactors on Water Pollutant Reduction from Agricultural Areas. Transactions of the ASABE 64, 641-658.

Clary, J., Jones, J., Leisenring, M., Hobson, P. and Strecker, E. 2020. International stormwater BMP database: 2020 summary statistics. The Water Research Foundation: Alexandria, VA, USA.

Close, M., Humphries, B., Tschritter, C., Sarris, T. and Moore, C. 2020 Model scenarios for a microbial risk assessment tool. Envirolink report, p. 61, ESR, Christchurch.

Collins, R., Donnison, A., Ross, C. and M, M. 2004. Attenuation of effluent-derived faecal microbes in grass buffer strips. New Zealand Journal of Agricultural Research 47, 565-574.

Collins, R., Elliott, S. and Adams, R. 2005. Overland flow delivery of faecal bacteria to a headwater pastoral stream. Journal of Applied Microbiology 99(1), 126-132.

Collins, R., Mcleod, M., Hedley, M., Donnison, A., Close, M., Hanly, J., Horne, D., Ross, C., Davies-Colley, R., Bagshaw, C. and Matthews, L. 2007. Best management practices to mitigate faecal contamination by livestock of New Zealand waters. New Zealand Journal of Agricultural Research 50(2), 267-278.

Craggs, R., Sukias, J., Tanner, C. and Davies-Colley, R. 2004. Advanced pond system for dairy-farm effluent treatment. New Zealand Journal of Agricultural Research 47(4), 449-460.

Daigneault, A.J. and Elliot, S. (2017) Land-use Contaminant Loads and Mitigation Costs: A Technical Paper, Motu Economic and Public Policy Research, Wellington, New Zealand.

Dakers, A., Clark, A. and Morrison, K. 2016 Domestic On-Site wastewater: real needs and relative risks.

Davies-Colley, R.J., Lidiard, E. and Nagels, J. 2008. Stormflow - dominated loads of faecal pollution from an intensively dairy-farmed catchment. Water Science and Technology 57(10), 1519-1523.

Drummond, J.D., Aquino, T., Davies-Colley, R.J., Stott, R. and Krause, S. 2022. Modeling Contaminant Microbes in Rivers During Both Baseflow and Stormflow. Geophysical Research Letters 49(8), e2021GL096514.

Ghermandi, A., Bixio, D., Traverso, P., Cersosimo, I., Thoeye, C 2007. The removal of pathogens in surface flow constructed wetlands and its implications for water reuse. water sci technol 56(3), 207-216.

Ghimire, C.P., Monaghan, R., Rutherford, A., Muirhead, R. and Lasseur, R. 2024. Grazing strategies for reducing contaminant losses in surface runoff from winter forage crop fields located in hill country and grazed by sheep. New Zealand Journal of Agricultural Research, 1-17.

Goeller, B.C., Sukias, J.P.S., Woodward, S.J.R. and Clarkson, B. 2023. 'Dual purpose' surface flow constructed treatment wetlands support native biodiversity in intensified agricultural landscapes. water 15, 2526.

Graham, E., Jones-Todd, C.M., Wadhwa, S. and Storey, R.G. 2018 Analysis of stream responses to riparian management on the Taranaki ring plain, p. 66, Report prepared for Taranaki Regional Council.

Guber, A.K., Yakirevich, A.M., Sadeghi, A.M., Pachepsky, Y.A. and Shelton, D.R. 2009. Uncertainty Evaluation of Coliform Bacteria Removal from Vegetated Filter Strip under Overland Flow Condition. Journal of Environmental Quality 38(4), 1636-1644.

Gutierrez-Gines, M.J., Alizadeh, H., Alderton, E., Ambrose, V., Meister, A., Robinson, B.H., Halford, S., Prosser, J.A. and Horswell, J. 2021. Phytoremediation of microbial contamination in soil by New Zealand native plants. Applied Soil Ecology 167, 104040.

Hallmich, C. and Gehr, R. 2010. Effect of pre-and post-UV disinfection conditions on photoreactivation of fecal coliforms in wastewater effluents. Water research 44(9), 2885-2893.

Hamaamin, Y.A., Adhikari, U., Nejadhashemi, A.P., Harrigan, T. and Reinhold, D.M. 2014. Modeling Escherichia coli removal in constructed wetlands under pulse loading. Water Research 50(0), 441-454.

Hruby, C., Soupir, M., Moorman, T., Shelley, M. and Kanwar, R. 2016. Effects of tillage and poultry manure application rates on Salmonella and fecal indicator bacteria concentrations in tiles draining Des Moines Lobe soils. Journal of Environmental Management 171, 60-69.

Hudson, N. and Nagels, J. 2011 Assessing the performance of the Lake Okaro constructed wetland, p. 95, Pastoral 21 Consortium under contract to AgResearch and for the Bay of Plenty Regional Council.

Hughes, A., McKergow, L.A., Tanner, C.C. and Sukias, J. 2013 Influence of livestock grazing on wetland attenuation of diffuse pollutants in agricultural catchments

Hughes, A.O., Tanner, C. C., McKergow, L. A., Sukias, J. P. S 2016. Unrestricted dairy cattle grazing of a pastoral headwater wetland and its effect on water quality. Agricultural Water Management 165, 72-81.

Humphries, B., Qiu, R., Langley, G., Pearson, A. and Weaver, L. 2023 On-site wastewater systems - risks and insights into their function, Wellington.

Jang, C., Gautam, S., Cooke, R.A. and Bhattarai, R. 2018 Development of a process-based subsurface bioreactor model, p. 1, ASABE, St. Joseph, MI.

Kadlec, R.H. and Wallace, S. (2008) Treatment wetlands, CRC press.

Khatiwada, N.R. and Polprasert, C. 1999. Kinetics of fecal coliform removal in constructed wetlands. Water Science and Technology 40(3), 109-116.

Levine, B., Burkitt, L., Horne, D., Tanner, C., Sukias, J., Condron, L. and Paterson, J. 2021. The ability of detainment bunds to decrease sediments transported from pastoral catchments in surface runoff. Hydrological Processes 35(8), e14309.

Lim, T.J.Y., Sargent, R., Henry, R., Fletcher, T.D., Coleman, R.A., McCarthy, D.T. and Lintern, A. 2022. Riparian buffers: Disrupting the transport of *E. coli* from rural catchments to streams. Water Research 222, 118897.

Mardani, S., McDaniel, R., Bleakley, B., H., Hamilton, T.L., Salam, S. and Amegbletor, L. 2020. The effect of woodchip bioreactors on microbial concentration in subsurface drainage water and the associated risk of antibiotic resistance dissemination. Ecological Engineering 143, 100017.

McDowell, R. 2007. Water quality in headwater catchments with deer wallows. Journal of Environmental Quality 36(5), 1377-1382.

McDowell, R., Wilcock, B. and Hamilton, D. 2013 Assessment of strategies to mitigate the impact or loss of contaminants from agricultural land to fresh waters. Report prepared for MfE, AgResearch, New Zealand.

McKergow, L., Hughes, A. and Rutherford, K. 2017 Seepage wetland protection review. Prepared for Dairy NZ, National Institute of Water and Atmospheric Research, Hamilton, New Zealand.

McKergow, L., Matheson, F.E., Goeller, B. and Woodward, B. 2022 Riparian Buffer design guide: Design to meet water quality objectives, NIWA, Hamilton, New Zealand.

McKergow, L., Tanner, C.C. and Monaghan, R.M., Anderson, G 2007 Stocktake of diffuse pollution attenuation tools for New Zealand pastoral farming systems, p. 111, NIWA, Hamilton, New Zealand.

McKergow, L.A., Rutherford, J.C. and Timpany, G.C. 2012. Livestock-Generated Nitrogen Exports from a Pastoral Wetland. Journal of Environmental Quality 41(5), 1681-1689.

McKergow, L.S., J.; Tanner, C 2008. Grass Filter Strips (GFS): On-farm biological mitigation options for nutrient management. Rotorua Farms Research Field Day. Wharenui Station, Rotorua.

Miller, J.J., Curtis, T., Chanasyk, D.S. and Reedyk, S. 2015. Influence of mowing and narrow grass buffer widths on reductions in sediment, nutrients, and bacteria in surface runoff. Canadian Journal of Soil Science 95(2), 139-151.

Monaghan, R., Laurenson, S., Dalley, D. and Orchiston, T. 2017. Grazing strategies for reducing contaminant losses to water from forage crop fields grazed by cattle during winter. New Zealand Journal of Agricultural Research 60(3), 333-348.

Moore, J.A. 1988. Evaluating coliform concentrations in runoff from various animal waste management systems.

Muirhead, R. 2015. A Farm-Scale Risk-Index for Reducing Fecal Contamination of Surface Waters. Journal of environmental quality 44(1), 248-255.

Muirhead, R., Houlbrooke, D. and Monaghan, R. 2010 Risk assessment of farm dairy effluent irrigation systems: faecal indicator organisms, pp. 61-66.

Muirhead, R.W. and Doole, G. 2017 A Farm-scale E. coli model for Gisborne District Council, p. 38.

Muirhead, R.W., Elliott, A.H. and Monaghan, R.M. 2011. A model framework to assess the effect of dairy farms and wild fowl on microbial water quality during base-flow conditions. Water research 45, 2863-2874.

Munoz-Carpena, M., R., and Parsons, J.E. 2011 VFSMOD-W, vegetated filter strips modelling system, model documentation and user's manual version 6.x, p. 194, Agricultural & Biological Engineering University of Florida 287 Frazier Rogers Hall Gainesville, FL, 32611–0570.

New Zealand Government 2023 National Policy Statement for Freshwater Management 2020 February 2023, New Zealand Government, Wellington, New Zealand.

O'Callaghan, P., Kelly-Quinn, M., Jennings, E., Antunes, P., O'Sullivan, M., Fenton, O. and hUallacháin, D.Ó. 2019. The Environmental Impact of Cattle Access to Watercourses: A Review. Journal of Environmental Quality 48(2), 340-351.

Olilo, C.O., Muia, A.W., Moturi, W.N., Onyando, J.O. and Amber, F.R. 2016a. The current state of knowledge on the interaction of *Escherichia coli* within vegetative filter strips as a sustainable best management practice to reduce fecal pathogen loading into surface waters. Energy, Ecology and Environment 1(4), 248-266.

Olilo, C.O., Onyando, J.O., Moturi, W.N., Muia, A.W., Ombui, P., Shivoga, W.A. and Roegner, A.F. 2016b. Effect of vegetated filter strips on transport and deposition rates of *Escherichia coli* in overland flow in the eastern escarpments of the Mau Forest, Njoro River Watershed, Kenya. Energy, Ecology and Environment 1(3), 157-182.

Olilo, C.O., Onyando, J.O., Moturi, W.N., Muia, A.W., Roegner, A.F., Ogari, Z., Ombui, P.N. and Shivoga, W.A. 2016c. Composition and design of vegetative filter strips instrumental in improving water quality by mass reduction of suspended sediment, nutrients and Escherichia coli in overland flows in eastern escarpment of Mau Forest, Njoro River Watershed, Kenya. Energy, Ecology and Environment 1(6), 386-407.

Oliver, D.M., Porter, K.D., Pachepsky, Y.A., Muirhead, R.W., Reaney, S.M., Coffey, R., Kay, D., Milledge, D.G., Hong, E. and Anthony, S.G. 2016. Predicting microbial water quality with models: over-arching questions for managing risk in agricultural catchments. Science of the Total Environment 544, 39-47.

Orosz-Coghlan, P.A., Rusin, P.A., Karpiscak, M.M. and Gerba, C.P. 2006. Microbial source tracking of Escherichia coli in a constructed wetland. Water Environment Research 78(3), 227-232.

Pachepsky, Y., Allende, A., Boithias, L., Cho, K., Jamieson, R., Hofstra, N. and Molina, M. 2018. Microbial water quality: monitoring and modeling. Journal of environmental quality 47(5), 931-938.

Page, B., Badiou, P. and Steele, O. 2023. Nutrient retention of newly restored wetlands receiving agricultural runoff in a temperate region of North America. Ecological Engineering 195, 107060.

Parajuli, P.B., Mankin, K.R. and Barnes, P.L. 2008. Applicability of targeting vegetative filter strips to abate fecal bacteria and sediment yield using SWAT. Agricultural Water Management 95(10), 1189-1200.

Park, Y.S., Engel, B.A., Shin, Y., Choi, J., Kim, N.-W., Kim, S.-J., Kong, D.S. and Lim, K.J. 2013. Development of Web GIS-Based VFSMOD System with Three Modules for Effective Vegetative Filter Strip Design. Water 5(3), 1194-1210.

Paterson, J., Clark, D. and Levine, B. 2020 Detainment bund PS120: a guideline for on-farm, pasturebased, storm water run-off treatment., Phosphorus Mitigation Project Inc., Rotorua, New Zealand.

Prosser, J.A., R.R. Woods , J. Horswell and Robinson, B.H. 2016. The potential in-situ antimicrobial ability of Myrtaceae plant species on pathogens in soil. Soil Biology & Biochemistry 96, 1-3.

Rambags, F., C.C., T., Stott, R. and L.A., a.S. 2016. Fecal bacteria, bacteriophage, nutrient and organic load reduction in a full-scale denitrifying woodchip bioreactor. Journal of Environmental Quality 45(3), 847-854.

Rambags, F., Tanner, C.C. and Schipper, L.A. 2019a. Denitrification and anammox remove nitrogen in denitrifying bioreactors. Ecological Engineering 138, 38-45.

Rambags, F., Tanner, C.C., Stott, R. and Schipper, L.A. 2019b. Bacteria and virus removal in denitrifying bioreactors: Effects of media type and age. Ecological Engineering 138, 46-53.

Rivas, A., Barkle, G., Stenger, R., Moorhead, B. and Clague, J. 2020. Nitrate removal and secondary effects of a woodchip bioreactor for the treatment of subsurface drainage with dynamic flows under pastoral agriculture. Ecological engineering 148, 105786.

RJ Wilkinson, LA McKergow, RJ Davies-Colley, DJ Ballantine and Young, a.R. 2011. Modelling storm-event E. coli pulses from the Motueka and Sherry Rivers in the South Island, New Zealand. New Zealand Journal of Marine and Freshwater Research 45(3), 369-393.

Schipper, L.A., Robertson, W.D., Gold, A.J., Jaynes, D.B. and Cameron, S.C. 2010. Denitrifying bioreactors—An approach for reducing nitrate loads to receiving waters. Ecological Engineering 36(11), 1532-1543.

Schreiber, C., Rechenburg, A., Rind, E. and Kistemann, T. 2015. The impact of land use on microbial surface water pollution. Int J Hyg Environ Health 218(2), 181-187.

Semadeni-Davies, A., Elliott, S. and Matthews, Y. 2023a Modelling *E. coli* to support implementation of the NPS-FM: Stage 2 technical report, p. 78, Taranaki Regional Council.
Semadeni-Davies, A., Elliott, S. and Matthews, Y. 2023b Regional modelling of *E. coli* to support implementation of the NPS-FM: Stage 2 technical report, p. 107, Horizons Regional Council.
Semadeni-Davies, A., Elliott, S. and Yalden, S. 2023c Calibration of the CLUES *E. coli* model for the Taranaki and Manawatū-Whanganui regions: Stage 1 technical report, p. 45, Horizons and Taranaki Regional Councils.
Shen, P., Deletic, A., Urich, C., Chandrasena, G.I. and McCarthy, D.T. 2018. Stormwater biofilter treatment model for faecal microorganisms. Science of The Total Environment 630, 992-1002.
Smith, L.C. and Muirhead, R.W. 2023. A review of the effectiveness of sediment traps for New Zealand agriculture. New Zealand Journal of Agricultural Research, 1-18.
Soupir, M.L., Hoover, N.L., Moorman, T.B., Law, J.Y. and Bearson, B.L. 2018. Impact of temperature and hydraulic retention time on pathogen and nutrient removal in woodchip bioreactors. Ecological Engineering 112, 153-157.
Soupir, M.L. and Mostaghimi, S. 2011. E*scherichia coli* and Enterococci Attachment to Particles in Runoff from Highly and Sparsely Vegetated Grassland. Water, Air, & Soil Pollution 216(1), 167-178.
Sowah, R., Zhang, H., Radcliffe, D., Bauske, E. and Habteselassie, M.Y. 2014. Evaluating the influence of septic systems and watershed characteristics on stream faecal pollution in suburban watersheds in Georgia, USA. Journal of Applied Microbiology 117(5), 1500-1512.
Stott, R., Sukias, J., Tanner, C. and Paterson, J. (2022) Adaptive

Strategies for Future Farming. Christensen, C.L., Horne, D.J. and R., S. (eds), Farmed Landscapes Research Centre, Massey University. Occasional Report No. 34, Palmerston North, New Zealand.
Stott, R., Sukias, J.P.S., McKergow, L.A., Davies-Colley, R.J. and Tanner, C.C. 2023. Net export of E. coli from a constructed wetland treating agricultural drainage. Ecological Engineering 194, 107023.
Stott, R., Tondera, K., Blecken G, T. and C., S. (2018) In Ecotechnologies for the Treatment of variable stormwater and wastewater flows. Tondera, K., G-T., B., Chazarenc, F. and Tanner, C.C. (eds), pp. 57-74, SpringerBriefs in Water Science and Technology.
Sukias, J. and Heubeck, S. 2020 Performance of a constructed wetland receiving drainage water on a Lichfield dairy farm: 2019 drainage year, p. 95, DairyNZ.
Tanner, C., C.,, Depree, C., V.,, Sukias, J., P, S.,, Wright-Stow, A., E.,, Burger, D., F., and Goeller, B., C. 2022 Constructed Wetland Practioner Guide: Design and performance estimates, p. 40, DairyNZ/NIWA, Hamilton, New Zealand, DairyNZ Ltd.
Tanner, C., Sukias, J. and Burger, D. 2015 Realising the value of remnant wetlands as farm attenuation assets. Currie, L.D. and Burkitt, L.L. (eds), Fertilizer and Lime Research Centre, Massey University.
Tanner, C.C., Tomer, M.D. and Goeller, B.C. 2023a. A framework for applying interceptive mitigations for diffuse agricultural pollution. New Zealand Journal of Agricultural Research, 1-22.
Tanner, C.C., Tomer, M.D., Goeller, B.C. and Matheson, F.E. (2023b) Diverse solutions for efficient land, water and nutrient use. Christensen, C.L., Horne, D.J. and R., S. (eds), Farmed Landscapes Research Centre, Massey University. Occasional Report No. 35, Palmerston North, New Zealand.
Taranaki Regional Council 2022 Stratford District Council Stratford WWTP Monitoring Programme Annual Report 2021-2022, Stratford, Taranaki.
Tate, K.W., Atwill, E.R., Bartolome, J.W. and Nader, G. 2006. Significant Escherichia coli attenuation by vegetative buffers on annual grasslands. J Environ Qual 35(3), 795-805.
Tilman, L., Plevan, A. and Conrad, P. 2011 Effectiveness of best management practices for bacteria removal Developed for the Upper Mississippi River bacteria TMDL, Emmons & Olivier Resources, Inc, St Paul, Minnesota.

Trewartha, M. 2023 On-site wastewater management risk assessment tool, Blenheim, New Zealand.
Uuemaa, E., Palliser, C.C., Hughes, A.O. and Tanner, C.C. 2018. Effectiveness of a natural headwater wetland for reducing agricultural nitrogen loads. Water 10(3), 287.
Wheatley, R.M. and Poole, P.S. 2018. Mechanisms of bacterial attachment to roots. FEMS microbiology reviews 42(4), 448-461.
Wu, S., Carvalho, P.N., Müller, J.A., Manoj, V.R. and Dong, R. 2016. Sanitation in constructed wetlands: A review on the removal of human pathogens and fecal indicators. Science of The Total Environment 541, 8-22.
Yu, C., Duan, P., Yu, Z. and Gao, B. 2019. Experimental and model investigations of vegetative filter strips for contaminant removal: A review. Ecological Engineering 126, 25-36.