Tanner, C.C., McKergow, LA., Goeller, B.C., Woodward, K.B., Sukias, J.P.S., Craggs, R.J. and Matheson, F.E. 2020. The spectrum of edge-of-field to waterway mitigation options for nutrient management in New Zealand's farmed landscapes. In: *Nutrient Management in Farmed Landscapes*. (Eds C.L. Christensen, D.J. Horne and R. Singh). http://flrc.massey.ac.nz/publications.html. Occasional Report No. 33. Farmed Landscapes Research Centre, Massey University, Palmerston North, New Zealand. 25 pages.

THE SPECTRUM OF EDGE-OF-FIELD TO WATERWAY MITIGATION OPTIONS FOR NUTRIENT MANAGEMENT IN NEW ZEALAND'S FARMED LANDSCAPES

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Introduction

The government, regulators, iwi and industry are working together with farmers to address the cumulative environmental impacts of intensive farming practices here in New Zealand, as in many other places around the world. Source control through improved in-field management of diffuse nutrient losses, focussing on input and output budgeting, good fertiliser application and sustainable livestock and cropping practices are the first place to start (McDowell et al. 2013). Protection and enhancement of natural attenuation assets, such as remnant wetlands and vegetated headwaters and gullies, can also provide benefits at low cost. However, in some areas, these mitigation actions will be insufficient to meet desired nutrient discharge limits. We know that all landscapes are not equal in terms of: 1) their propensity to retain or leak nutrients, 2) the efficacy of downstream nutrient attenuation processes or 3) the ecological sensitivity (or resilience) of downstream receiving waters. Many of the interventions required to facilitate farming make things worse by increasing connectivity between land and water and/or compromising natural attenuation processes in the landscape (e.g. draining wetlands and removing bush cover). Here, we briefly overview the spectrum of potential edge-of-field to waterway mitigation options available to complement in-field management of pastoral agriculture and illustrate these with applications from New Zealand and/or Denmark. Options overviewed include mitigations positioned at edge-of-field, edge-of-waterway (riparian), and within surface and subsurface drainage pathways. Larger catchment-scale actions such as re-storing floodplain functioning and bottom-of-catchment wetlands are not addressed here.

Flow pathways

The pathways by which run-off and associated contaminants travel from land to water determine the ratio of base- to stormflow and seasonal flow variability. It determines the locations in the landscape where runoff can be intercepted, the forms of contaminants mobilised, and the suitability and efficacy of mitigation options. Figure 1 illustrates a simplified combination of generic soil types and summarises their flow and contaminant characteristics. Highly permeable soils and subsoils (Type a, Fig. 1) promote infiltration resulting in predominant flows to groundwater, which often re-emerge some distance from their source, making them difficult to intercept and deal with locally. Impermeable subsoils or pans below permeable subsoils create barriers to downward movement of infiltrating water (aquacludes), generating horizontal subsurface flow (interflow) which typically emerges more locally in riparian zones and where there are changes in slope (e.g. toe-slopes), soil texture, water table or flow convergence (e.g. headwaters) (Type b, Fig. 1). Occasional high intensity storms may still generate significant loads of particulates and associated nutrients, particularly P, from this type of soil (e.g. Levine et al. 2020).

Landscapes with impermeable soils (soil types c and d) tend to generate greater quantities of surface run-off, particularly during high intensity storm events. This results in much more variable outflows and mobilises greater quantities of sediments and particulate-associated nutrients. To enable productive pasture and crop growth, sustainable grazing and trafficability, these soil types with restricted infiltration (and also type b) are commonly modified by subsurface tile and mole drainage. This decreases the potential for surface run-off, reducing export of sediments and associated contaminants, but increases the export of dissolved nutrients such as nitrate and orthophosphate (Monaghan et al., 2005; Barkle et al. 2017; Gramlich et al. 2018). Dissolved contaminants discharged in such subsurface drains short-circuiting the attenuation processes that commonly occur during passage through soil and, in particular, riparian zones (McKergow et al., 2016, 2020; Goeller et al. 2019b).

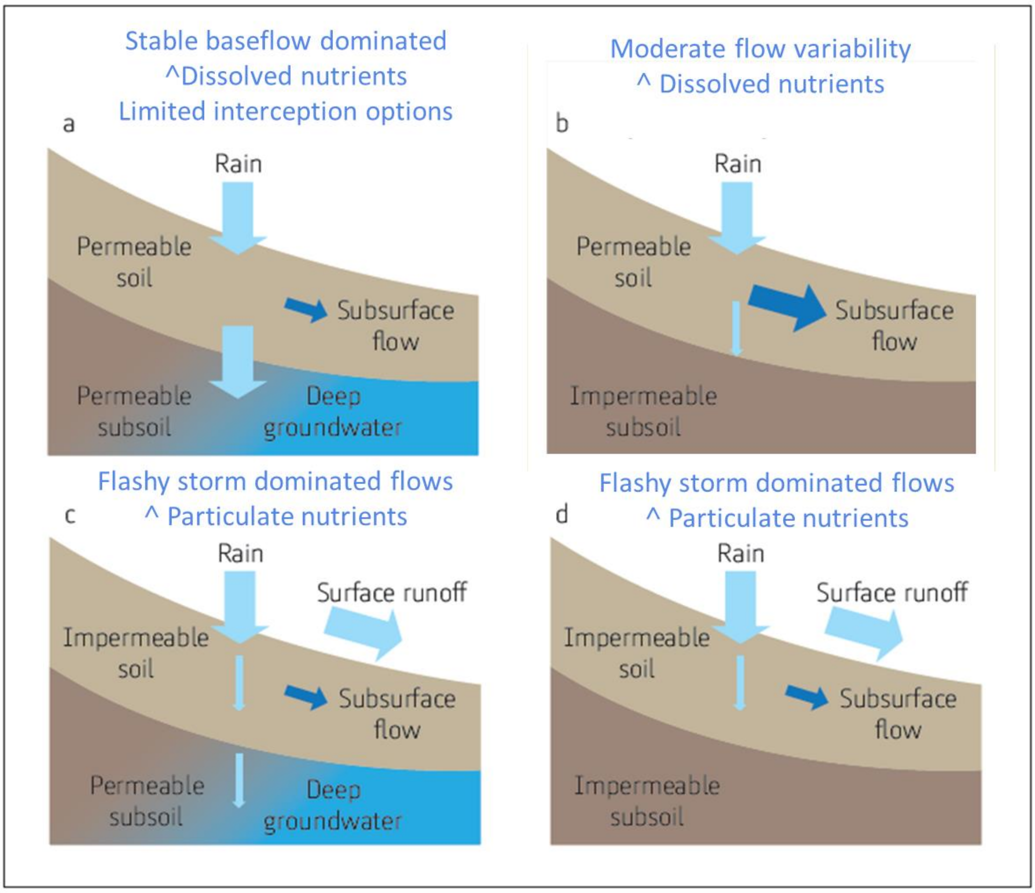

Figure 1: Four basic soil characteristics and the nature of the contaminant flows they generate (adapted from Leibowitz et al. 2018)

Contaminant attenuation processes

A wide range of different contaminant attenuation processes operate as water flows over and through landscapes (Fig. 2). They range from relatively rapid physical processes such as coarse sediment deposition (minutes and hours) through slower biological processes such as plant uptake and microbial denitrification (hours and days). The different options available for mitigation of water quality attempt to reinforce or amplify one or more of these same processes. Obviously, to be effective mitigations must be appropriate to the flow path and contaminant types being targeted and located where they can intercept them. They can only influence the flow and contaminant loads that pass through them and only if they promote appropriate attenuation processes and have sufficient capacity to sustainably receive and manage the flows and loads they receive. For instance, riparian buffers cannot provide effective mitigation where flows containing the contaminant of concern (e.g. leached N) are predominantly transported deep beneath them or bypass them via subsurface drainage (Goeller et al. 2019b, Vidon et al. 2019). Alternatively, if surface run-off during storms is the main flow pathway (e.g. for particulate phosphorus), episodic channelised flow will overwhelm their treatment capacity at the point of interception, while other sections of the riparian buffer will provide minimal benefit (McKergow et al 2020).

Mitigations

There is a spectrum of potential mitigation tools available for managing diffuse agricultural losses of nutrients and other contaminants that, depending on landscape characteristics and farming systems, are suitable for different locations on farms and within catchments. In general terms they can be grouped as edge-of-field, edge-of-waterway (riparian), or flow pathway mitigations. The development and testing of these mitigation tools from proof-of-concept through to recognition as an acceptable mitigation practice by farmers, industry and regulatory agencies often has multiple steps involving pilot-, field-, and farm-scale testing. (Fig. 3). There is also an ongoing need to learn from the application of these tools in different situations and by different farmers to refine our understanding and adapt the tools and the guidance provided for their use. Below, we provide a quick overview of a cross-section of the mitigation tools currently available, being investigated or in consideration in New Zealand.

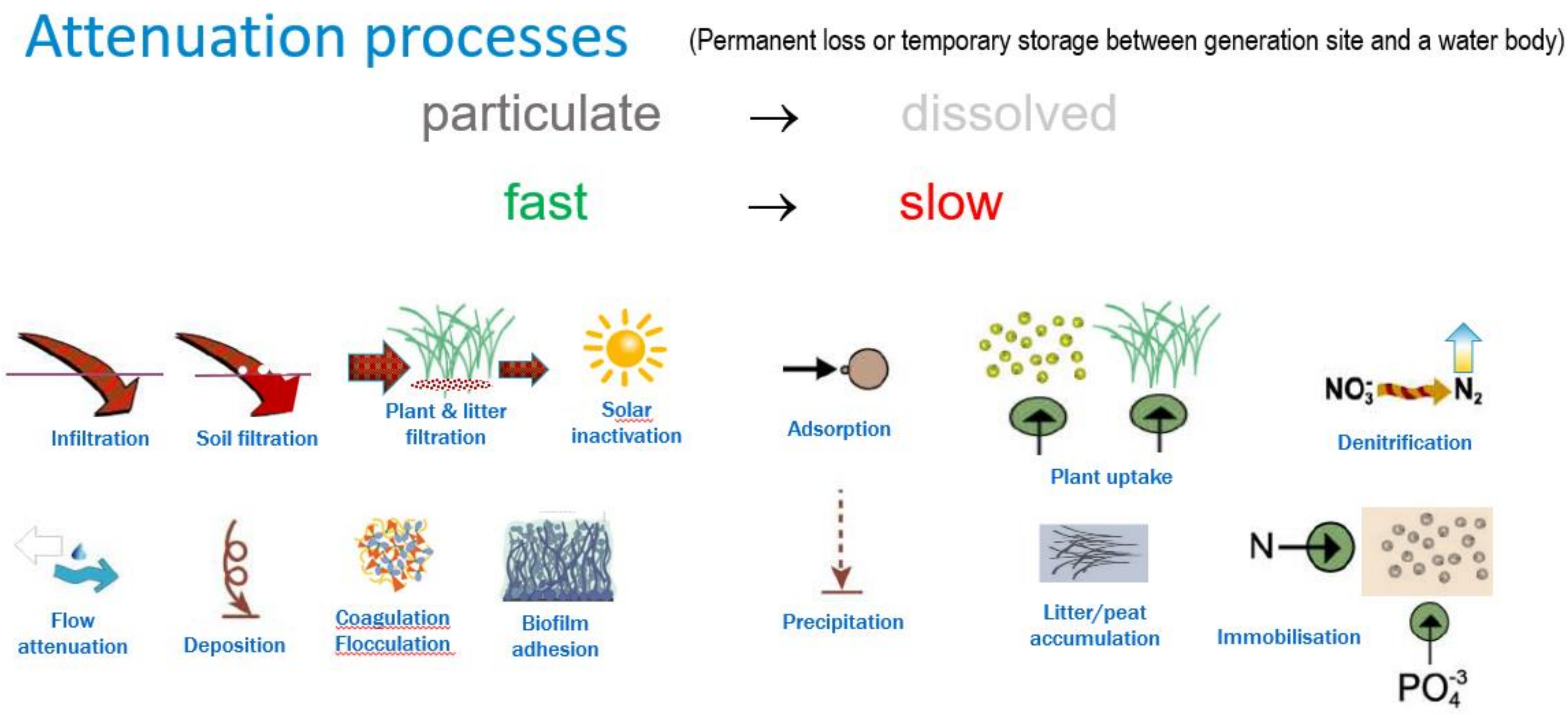

Figure 2: Visual summary of key nutrient attenuation processes that occur as water and contaminants flow through and across landscapes.

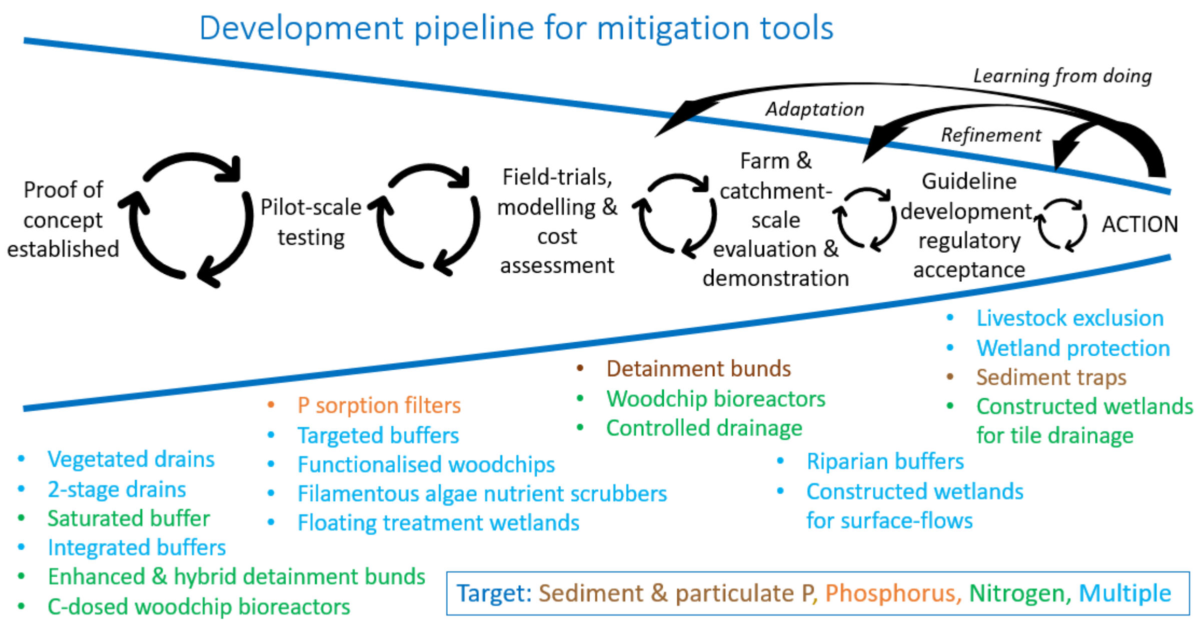

Figure 3: Summary of edge-of-field, -waterway (riparian) and flow pathway mitigation options and their approximate position along the development pipeline in New Zealand. The contaminants potentially targeted are colour-coded as indicated.

Edge-of-field and edge-of-waterway (riparian) mitigations

Riparian buffer zones

The use of riparian buffer zones for water quality improvement in agricultural settings is one of the most common best management practices (BMPs) applied for mitigating agricultural impacts on waterways (Vidon et al. 2019). Riparian buffer zones between productive lands and surface waters range from simple set-back of agricultural activities (a non-cultivated zone) or exclusion of livestock to managed bands of vegetation – comprising grass, shrubs and/or trees (Fig. 4; McKergow et al. 2020). They can reduce contaminant loads in surface runoff and subsurface flows through several processes, including physical retention, biological uptake and biogeochemical processing. These occur above ground, in the root zone of plants and in the subsoil.

Although the concept of riparian buffers is well known and they are widely applied for diffuse pollution mitigation across the country, a recent review (McKergow et al. 2020) has highlighted the paucity of New Zealand data available to quantifying their contaminant removal efficacy. Reviews of international data show that performance efficacy varies widely across different landscape types, depending on the flow path being intercepted and the ratio of filter strip and buffer area (Fig. 4) to upslope contributing area, (i.e., buffer area ratio; Polyakov et al 2005; Dosskey et al. 2011). Flow from pastoral landscapes is rarely uniform across hillslopes. Patterns of topography and micro-relief cause convergence into preferential channels which often leads to concentrated flows through relatively small sections of the riparian zone (Fig. 5). Performance can be markedly increased by targeting riparian buffer area to the flow paths where runoff load is greatest (Dosskey at al. 2002). Ways to target application of variable width buffers across different landscapes are the subject of current investigations in New Zealand (e.g., Goeller et al. 2020) and new guidelines are being developed by NIWA and DairyNZ to improve application of riparian buffers and provide estimates of expected contaminant removal performance.

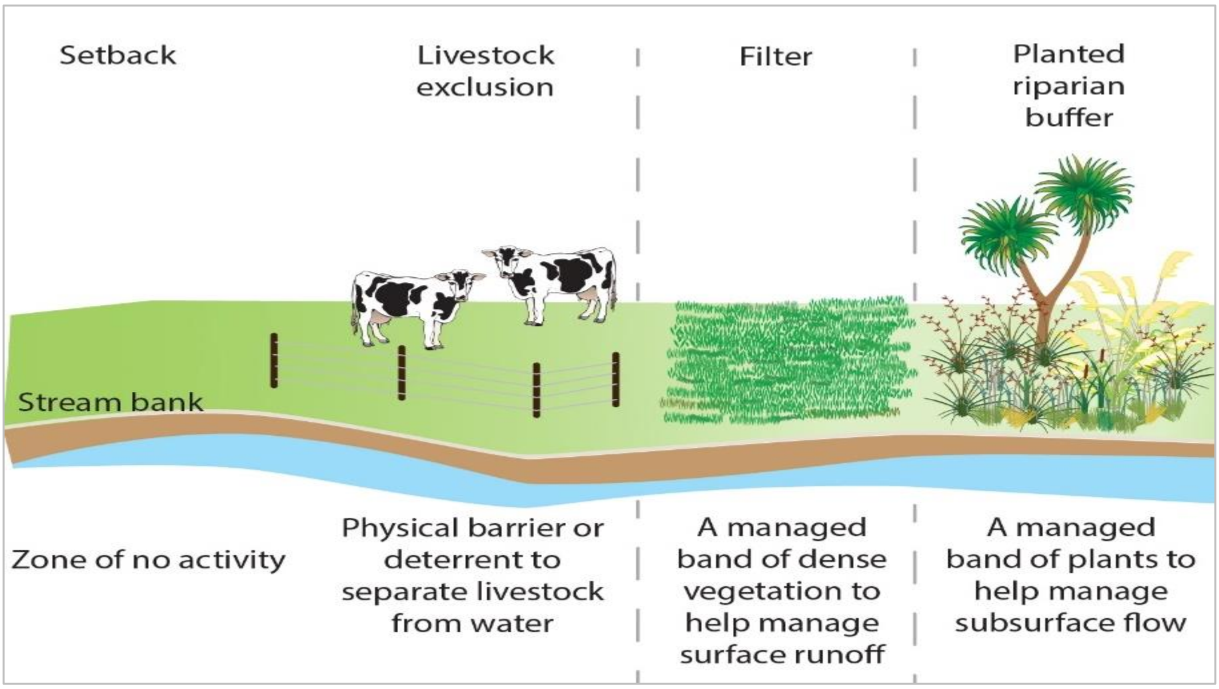

Figure 4: The range of basic riparian buffer types (McKergow et al. 2020)

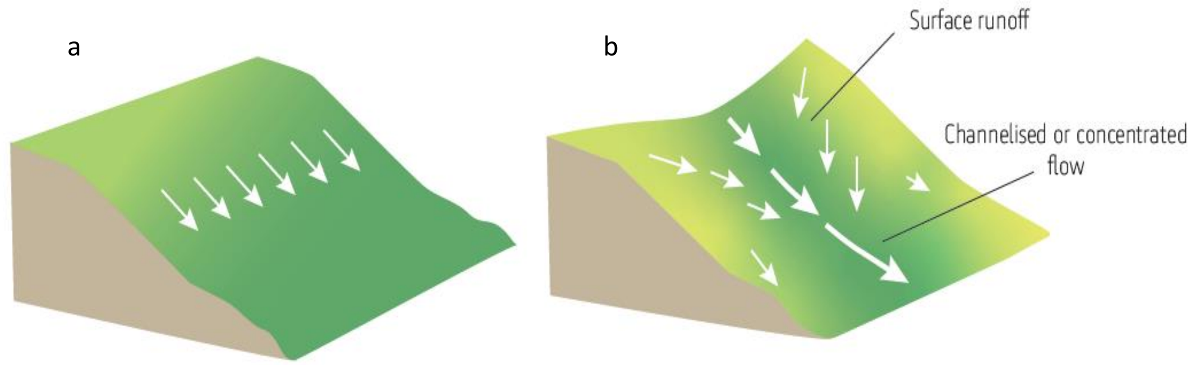

Figure 5: Flow convergence effect on runoff across filter strip face. (a) No flow convergence on a planar slope - runoff flows across the entire filter face at low depth and velocity, (b) flow convergence causes runoff to concentrate and enter a narrow filter face increasing runoff velocity and depth; the filter strip receives little runoff at other locations.

Saturated buffers

Saturated buffers are a relatively recent innovation involving modification of the outlets of field drainage systems. A perforated pipe laid parallel to the waterway laterally redistributes a proportion of drainage flows to percolate through riparian soils (Figs. 6 and 7). Nitrogen removal occurs via denitrification in the saturated soils, similar to that occurring in natural riparian buffers (Jaynes and Isenhart, 2019; Tomer et al. 2020).

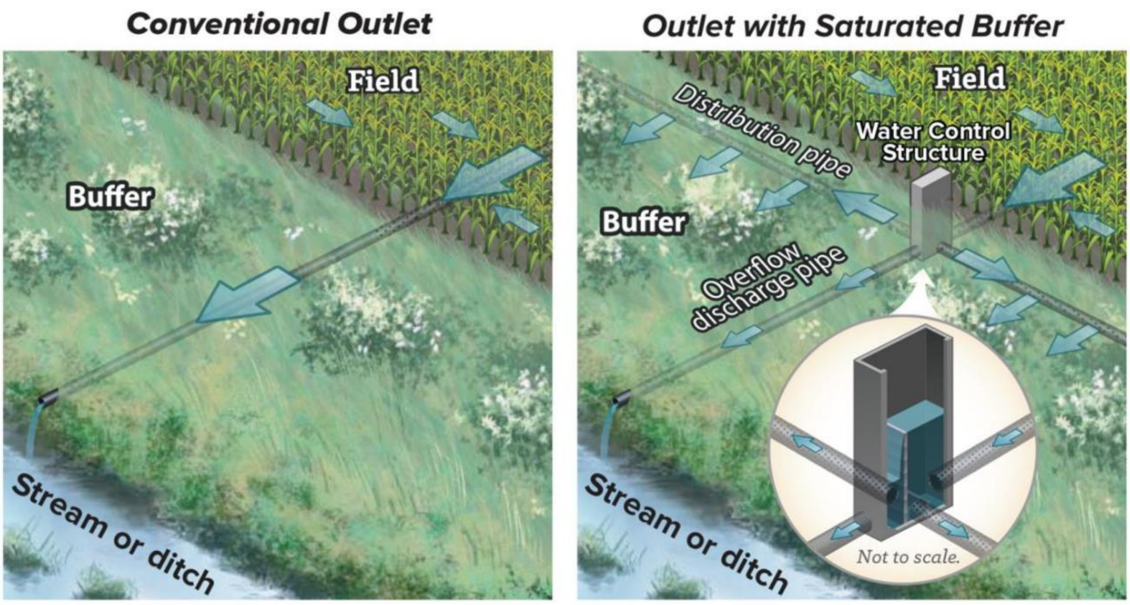

Figure 6. Comparison of a conventional subsurface (tile) drain outlet to a waterway (left) with a saturated buffer outlet, where water is redistributed laterally into riparian soils (from Christianson et al. 2016).

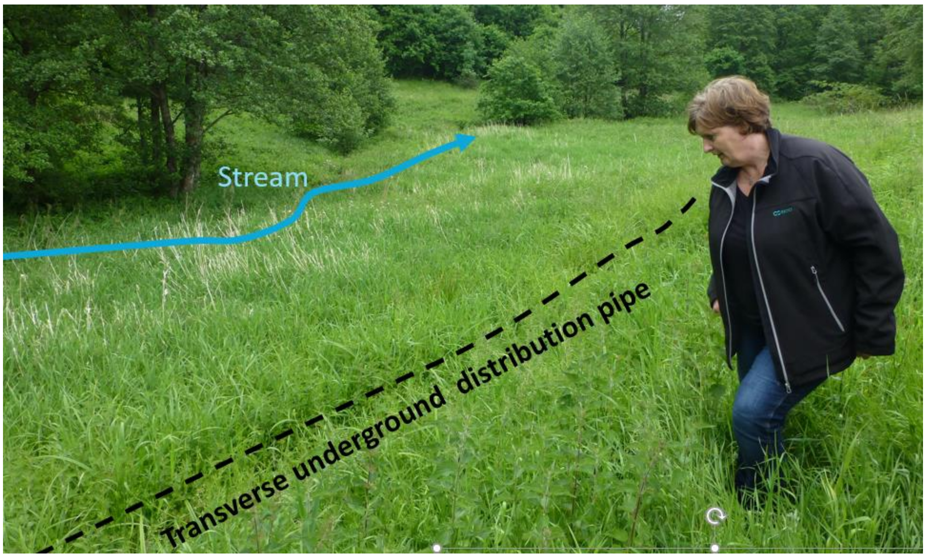

Figure 7: Dr Charlotte Kjaergaard (SEGES) inspecting the location of an established saturated buffer being trialled in Denmark (Photo: Chris Tanner, July 2019).

Denitrification walls intercepting shallow groundwater paths
Addition of slow-release carbon sources such as sawdust or woodchips can promote nitrate removal through microbial denitrification under anoxic conditions. The efficacy of carbon-rich walls or beds intercepting shallow groundwater paths (Fig. 8) was initiated here in New Zealand (Schipper and Vojvodic-Vukovic, 2001) and has also been widely investigated internationally (Schipper et al. 2010). A denitrification wall is also being tested in an alluvial gravel aquifer near Kaipoi in the Silverstream catchment, Canterbury (Burbery et al. 2019). Options to create similar denitrification zones along tile drains have also been investigated (Jaynes et al. 2008).

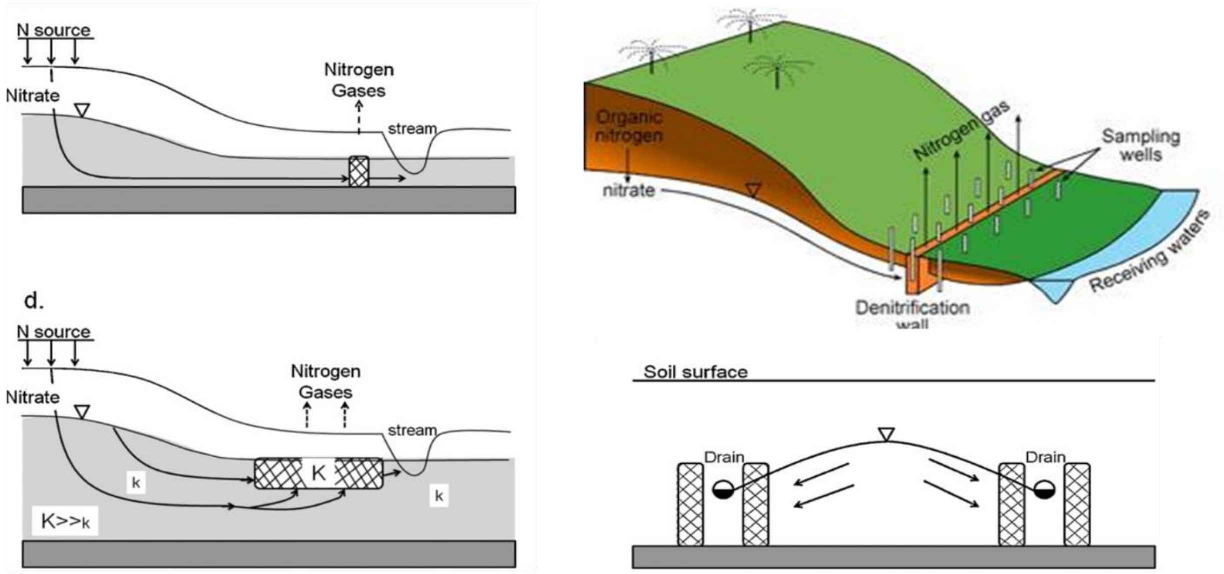

Figure 8: Various permutation of denitrification walls and beds (from Schipper et al. 2010 and Schipper and Vojvodic-Vukovic 2001).

Integrated buffers
Integrated buffer zones combine a pond, where flow can be detained and soil particles present in drain water or surface runoff can settle, and a planted subsurface flow infiltration zone to promote denitrification and phosphorus sorption (Fig. 9; Zak et al. 2018). Water tolerant trees such as alder and willow and emergent wetland plants have been trialled in the infiltration zone (Fig. 10).

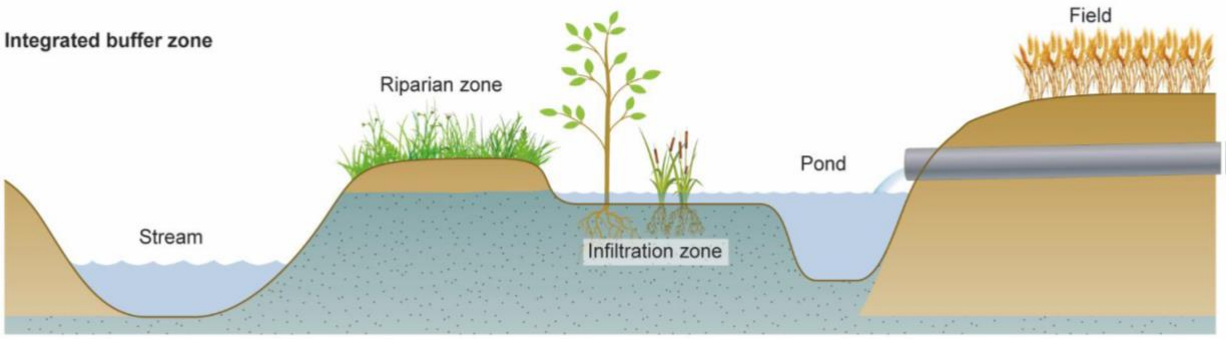

Figure 9: Integrated buffer zone intercepting subsurface drainage (from Zak 2018).


Figure 10: An Integrated Buffer Zone being trialled in Denmark (Photo: Chris Tanner, July 2019).

Flow pathway mitigations

Contaminant attenuation can also be achieved after diffuse run-off coalesces and flows along drainage ditches or low-order stream channels, or is intercepted by subsurface tile drains. The advantages of targeting these flows is that run-off that was previously widely dispersed is now concentrated in a channel (a diffuse-point source; Neal and Jarvie 2005) and will likely represent a substantial proportion of the contaminant yield to downstream waterways. The challenge, however, is that now you have an episodic, fast-moving and typically very flashy flux of water and contaminants, which can overwhelm the treatment capacity of mitigation tools. Surface flows

Re-engineered drainage ditches

Agricultural drainage ditches are generally designed and managed solely to get rid of excess water during wet periods of the year. Plant growth in drainage ditches is thus considered antithetical to their efficient functioning, because it reduces their hydraulic conductivity and traps sediments in their base (requiring frequent mechanical "cleaning out") which would otherwise flush downstream. Turning this on its head, vegetation-enhanced retention of sediments (with the potential for recycling back onto surrounding land), buffering of flow, attenuation of contaminants and associated biodiversity and habitat enhancements, can provide wider environmental benefits (Dollinger et al. 2015).

Vegetated drainage ditches

Vegetated drainage channels (Fig. 11) have been found to be promote higher rates of denitrification (e.g. Pierobon et al. 2013; Soana et al. 2019; Taylor et al. 2015) and attenuation of a wide range of other contaminants (Nsenga Kumwimba et al. 2018; Vymazal and Březinová 2018). Accommodating vegetation in drainage channels while maintaining drainage function is likely to require different drain designs. One option is to have wider drains that can pass the same flow of water in the presence of vegetation.

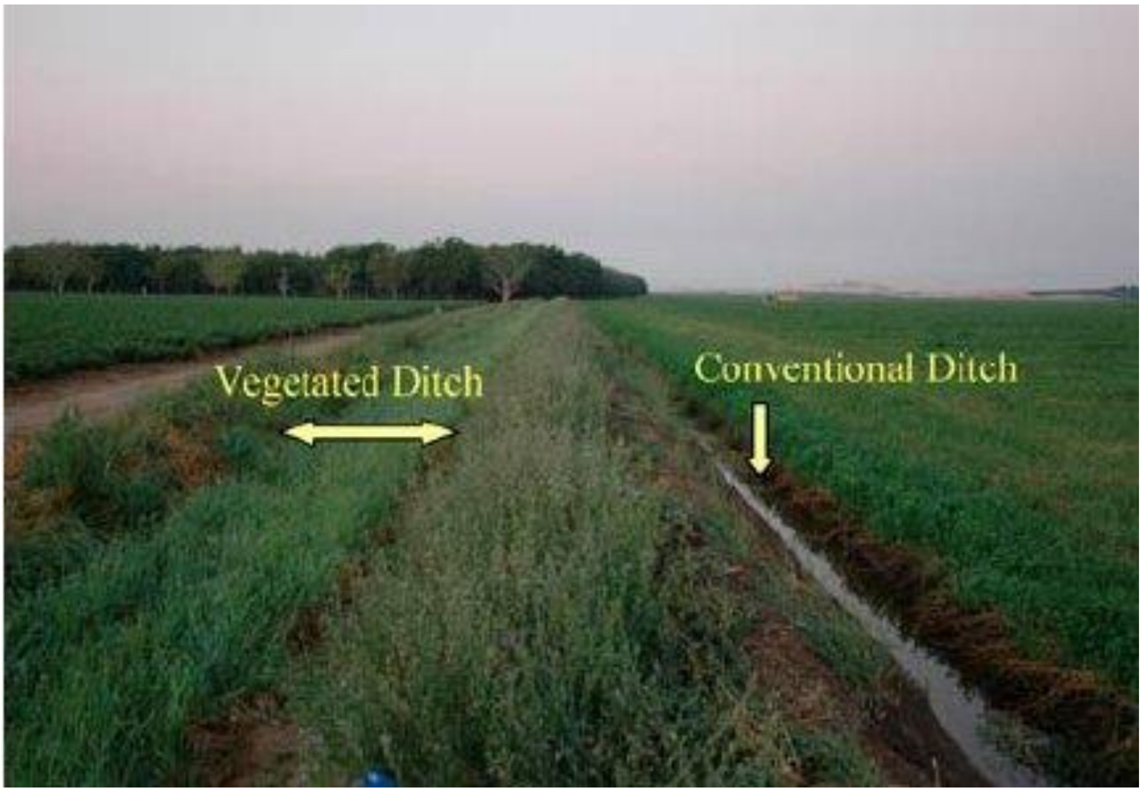

Figure 11: A vegetated compared to a conventional non-vegetated one (Gill et al. 2008)

Two-stage ditches

Two-stage ditches incorporate floodplain benches on either side of the drain that become inundated during high flow events (Fig. 12). The benches become vegetated over time and can intercept tile-drain inflows entering from surrounding fields (Kalcic et al. 2018; Nsenga Kumwimba et al. 2018; Vidon et al. 2019). They offer a potentially practical approach to incorporate the benefits of herbaceous vegetation whilst maintaining drain flow capacity and stability. Mahl et al (2015) undertook a multi-site evaluation of two-stage ditches and found significant reductions in turbidity and dissolved P. Although they measured apparent enhancement of N-removal capacity in the drain sediments, they did not find significant reductions in measured nitrate-N loads in the water column. A preliminary assessment of the potential of two-stage ditches in New Zealand has been undertaken by Febria and Harding (2018).

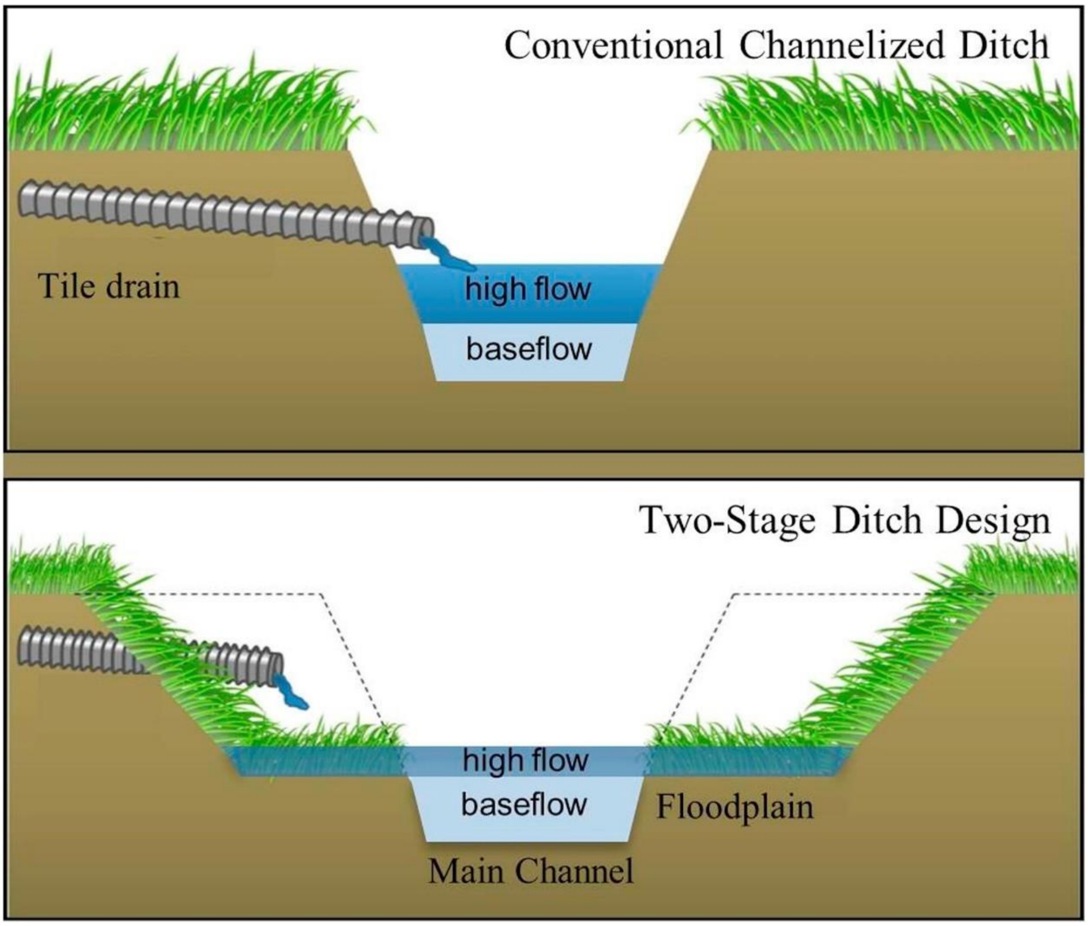

Figure 12: Comparison of a conventional drainage ditch and a two-stage ditch (from Christopher et al. 2017).

Low-grade weirs

Low-grade weirs are low, check-dam structures designed to be placed within drainage ditches to detain flow and enhance nutrient removal (Fig. 13; Kröger et al., 2012). Recent studies have highlighted benefits including increased hydraulic residence time and reductions in nutrient concentrations and sediment loads (Nsenga Kumwimba et al. 2018). Impacts on the high-flow conveyance characteristics of drains with low-grade weirs in Mississippi suggest the impact is minimal (Prince Czarnecki et al 2014).

Figure 13: A low-grade weir in Mississippi (from Prince Czarnecki et al 2014).

Peak run-off control

A step up from low-grade weirs is peak run-off control, which involves a series of higher-level bunds constructed in drainage ditches to temporarily detain and slow outflows (Fig, 14). By holding back the runoff, a proportion of the sediments and associated contaminants suspended in the water can settle out as storm flows are slowly released. An additional benefit is the reduction of stream power downstream, which reduces potential stream bank erosion and mobilisation of sediments already settled on the stream bed. Peak run-off control structures are being trialled in the Waituna Lagoon catchment in Southland (Living Water, 2020). The basic concept of peak run-off control can be extended and modified in a range of ways; e.g. detention bunds (see below).

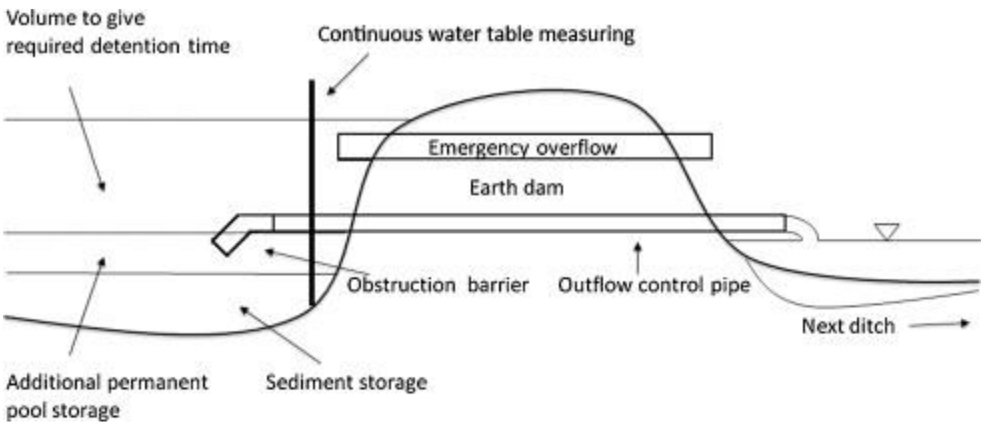

Figure 14: Schematic of peak run-off structure used to control discharge rates from drainage ditches (from Marttila et al. 2010).

Detention Bunds

A detention bund is a low earth berm (typically 1.5 - 2 m high) constructed across an ephemeral storm water flow path to temporarily detain storm water run-off for water quality objectives (Figs 15 and 16). The ponded water may partially infiltrate and then the remainder be released after a short period; 3 days is the recommended maximum period of detention, otherwise there is potential to harm pasture growth after inundation (Paterson et al. 2020). Clarke et al. (2013) studied 3 detention bunds in the Lake Rotorua catchment and found significant reductions in total suspended sediment concentrations of outflow waterafter ponding events. Reductions in particulate nutrients were also recorded during ponding events, although dissolved nutrient concentrations in the outflowing water typically did not decrease. Sediment retained in the ponded areas during this study was enriched with P relative to the benthic sediments of the stream and lake below.

Levine et al, (2020) undertook a comprehensive 12-month study of 37 ponding events in 2 detention bunds in the Lake Rotorua catchment. The detention bunds reduced annual discharge yields reaching connected waterways by 43% to 63% overall (including enhanced downstream infiltration when ponded water was released a few days after intense rainfall events). Similar reductions in suspended sediment, and total phosphorus and nitrogen were recorded due mainly to the decreased volume of runoff discharged. A guideline document has recently been produced based on the Rotorua studies (Paterson et al, 2020).

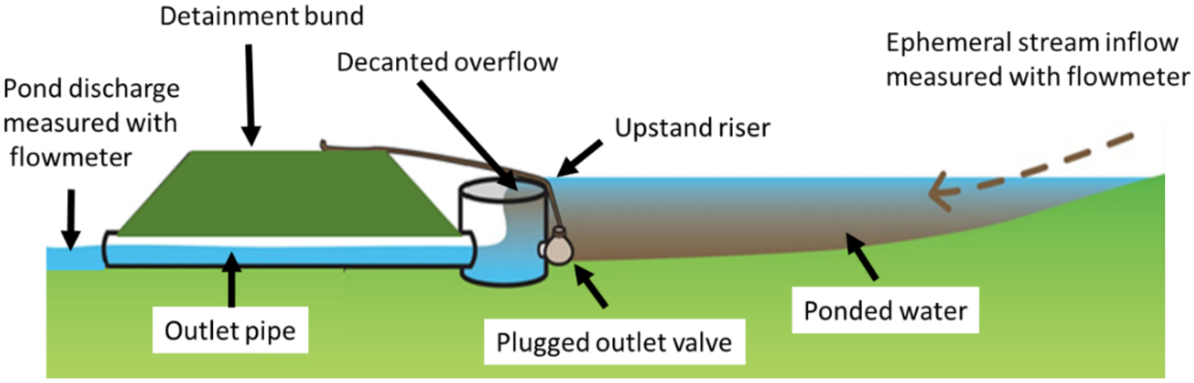

Figure 15: Schematic of Detainment Bund showing the ephemeral stream inflow, ponding behind a detainment bund and the upstand riser and discharge pipe features. (from Levine et al. 2020)

Figure 16: Ponded drainage temporarily impounded behind a detainment bund created from a causeway road (from Clarke et al. 2013).

Constructed wetlands

Surface-flow (sometimes known as free-water-surface) wetlands are the most appropriate type of constructed wetland for treating agricultural runoff[1]. Their ability to remove sediments and nutrients, especially nitrate-N, from diffuse agricultural runoff is well established (Crumpton et al. 2020; Kadlec 2012; Kadlec and Wallace 2009; O'Geen et al. 2010). Surface-flow constructed wetlands generally comprise channels or a series of vegetated shallow impoundments and operate similarly to natural swamps and marshes. As the name suggests, water flows across the surface of the wetland soil through beds of emergent aquatic plants such as sedges and bulrushes (see Figure 17). Their simplicity, robustness under highly variable flow conditions, and ability to cope with sediment loads make them widely applicable across a range of farm types, landscapes and flow pathways.

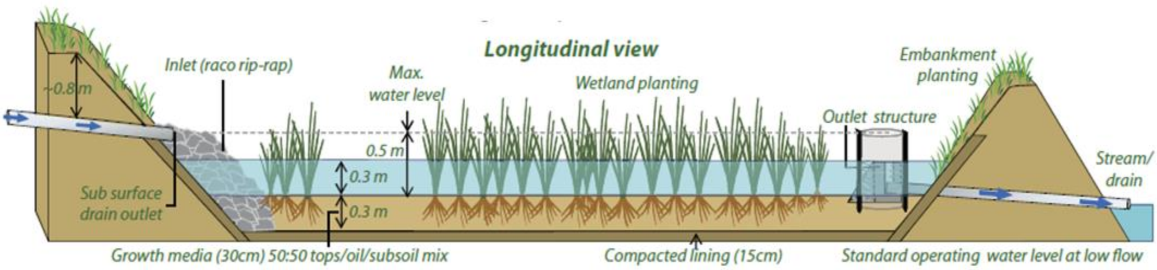

Figure 17: Longitudinal cross-section of a surface-flow constructed wetland for treating farm run-off. (Tanner et al. 2010)

Appropriately sized constructed wetlands are capable of reducing sediment, nitrogen, particulate phosphorus and faecal microbe loads, with performance related to their size relative to their contributing catchment. Testing of constructed wetland treatment of subsurface pastoral tile drainage has been carried out in New Zealand (Tanner and Sukias, 2011) and translated into practical guidelines (Tanner et al. 2010). Recently, the local and international performance of field-scale surface-flow wetlands has been reviewed (Woodward et al., 2020) and preliminary guidance provided for wetland design and expected treatment efficacy across a wider range of flow pathways (Tanner et al. 2020). Figure 18 shows, for example, expected TN reduction for different wetland sizes relative to contributing catchment for warm and cool areas of New Zealand New Zealand.

[1] The other major types of wetland are: 1) Sub-surface flow wetlands, where water flows horizontally or vertically through porous sand or gravel beds vegetated with emergent wetland plants (i.e. below the surface); and 2) Floating treatment wetlands, where water flows through the roots of emergent wetland plants supported on floating mats. These wetlands are generally more expensive to construct and less suited to highly variable diffuse agricultural flows and contaminant loads. Sub-surface flow wetlands in particular are vulnerable to clogging by suspended sediments, while floating treatment wetlands require relatively expensive floating platforms and do not cope well with extended dry periods during the year.

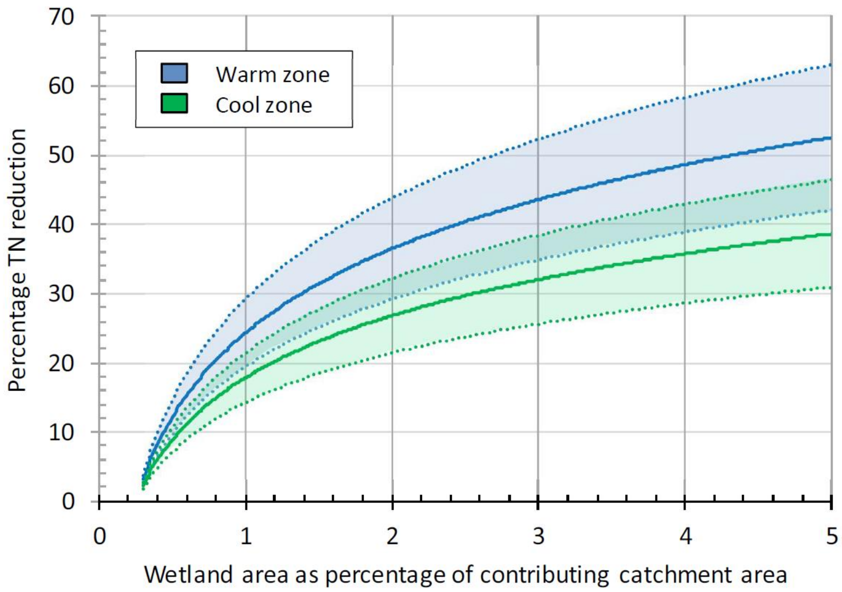

Figure 18: Long-term median annual TN reduction performance expectations for appropriately constructed wetlands receiving run-off and drainage from pastoral farmland for warm (median annual temperature >12°C) and cool (median annual temperature 8-12°C) climatic zones. Solid lines show expected medians; shaded areas show the expected variability in long-term median performance (Tanner et al. 2020).

A range of wetland trials are currently being developed by NIWA and DairyNZ in partnership with regional councils to fill wetland efficacy knowledge gaps across a wider range of New Zealand farming conditions (Fig. 19).

Figure 19: Two examples of constructed wetlands treating diffuse agricultural run-off in Waikato. Top: Baldwin’s wetland, Putaruru before full vegetation establishment, and bottom: Owl Farm wetland, Cambridge. (Photos: Chris Tanner)

Filamentous Algal Turf Scrubbers

A novel new approach for nutrient removal by periphytic algae using shallow flow-ways (Fig. 20) is being developed for New Zealand conditions (Sutherland and Craggs 2017). A range of filamentous species are being tested for uptake of nutrients from diffuse agricultural run-off. The biomass needs to be harvested regularly to maintain uptake. Nutrients retained in the algal biomass can be reused as livestock fodder or slow-release organic fertilisers and soil conditioners. There is also some potential in the future for the extraction of novel biological products. Fine sediments may also be retained in the biomass, while high daytime solar exposure, elevated pH and super-saturated dissolved oxygen concentrations can promote efficient disinfection of faecal microbes.

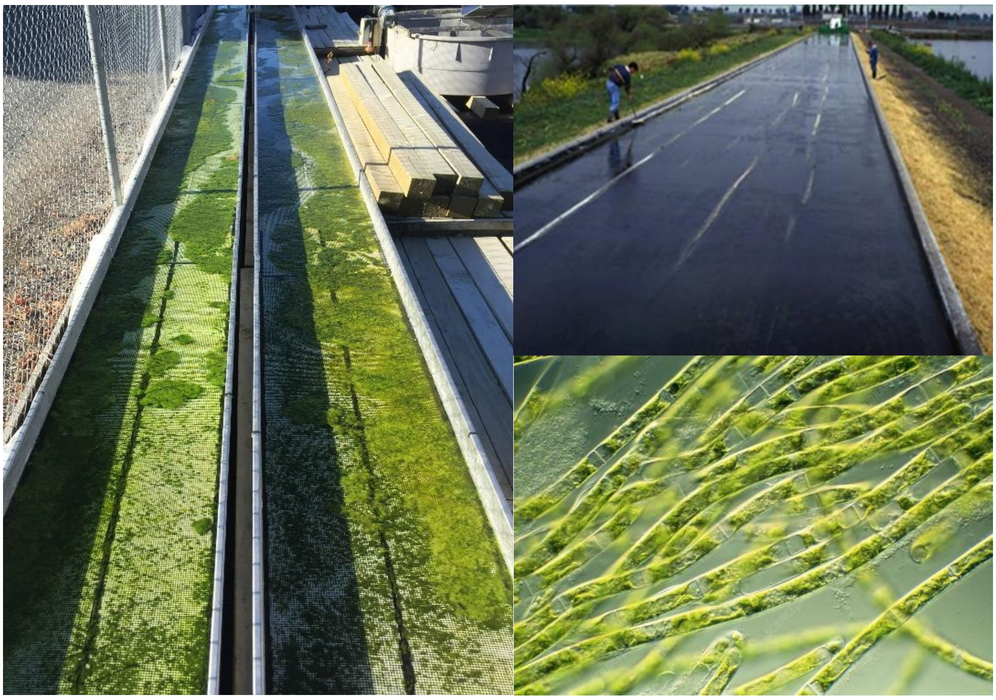

Figure 20: Filamentous Algal nutrient scrubbers being trialled at pilot scale in Waikato (left) and field scale in California, USA (top right). Close-up of filamentous algal (bottom right). (Photos: Rupert Craggs)

Re-engineered sub-surface drainage

The major effect of subsurface artificial drainage is to alter the route of water movement from surface run-off to subsurface drain-flow. This lowers the water-table, increasing infiltration rates and reducing the potential for surface run-off and particulate associated contaminants. However, it increases export of dissolved nutrients such as nitrate and orthophosphate (Monaghan, 2005; Tomer et al. 2003)

Controlled drainage

Controlled drainage is used in many places around the world as a beneficial management practice to decrease loads and concentrations of nutrients and pesticides transported to receiving waters (Skaggs et al. 2012; Carstensen et al. 2019). Basically it involves installing a water table control structure, either within subsurface drains or in surface drains to which they discharge (Fig. 21; see also peak run-off control above) The aim is to restrict drainage to only the excess water that would damage crop or pasture growth, or limit grazing or farm equipment access to paddocks.

There are few reported studies of controlled drainage in normal grazed pastures. Ballantine and Tanner (2010) reviewed the potential for use of controlled drainage in pastoral New Zealand agriculture. Controlled drainage appears to be most relevant for relatively flat land with an impermeable clay layer at about 1-3 m depth below the surface where significant areas of land drainage can be controlled from a single point without excessive loss to deep groundwater. Many existing tile-drained areas in New Zealand are not well set-up to enable retrofitting of controlled drainage.

Detailed studies have generally shown that reductions in nutrient loads are predominantly due to reduced drainage flows rather than significant reductions in nutrient concentrations (e.g. due to elevated denitrification in saturated subsoils). This brings up the question of how much of these apparent reductions in nutrient load still find their way to ground and surface waters, and what overall level of reduction is achieved at catchment scale (Carstensen et al. 2019).

Controlled drainage and other methods to buffer run-off flow peaks (e.g. low-grade weirs and peak runoff controls) also have potential to be used in combination with other mitigations to increase residence times and concomitant treatment efficiency.

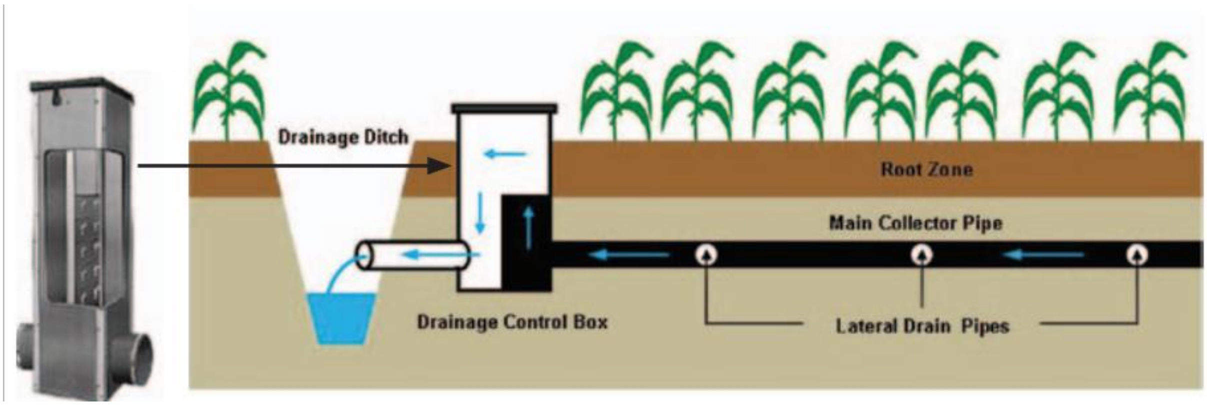

Figure 21: Concept of adjustable outlet weirs to control subsurface drainage water levels (Skaggs et al. 2012)

Woodchip bioreactor filters

Beds of carbon-rich woodchips (Figs. 22, 23, 24) have been shown to effectively remove nitrate-N from a range of wastewaters and agricultural drainage (Schipper et al. 2010; Christianson et al. 2012, 2016) and also provide reductions in ammonium (Rambags et al. 2019a) and faecal microbes (Rambags et al. 2019b). In New Zealand, nitrate removal from subsurface drainage has been tested in Waituna, Southland (Hudson et al. 2019.), Hinds, Canterbury (Goeller et al 2019a), Tatuanui, Waikato (Rivas et al. 2020) and Windy Farm, Wairarapa (Pratt, 2020).

Current woodchip bioreactor trials in the Waikato by NIWA, University of Waikato and Lincoln Agritech are looking at ways to accelerate treatment during high-flow events by carbon-dosings and functionalise woodchips to enhance phosphate and nitrate adsorption, and instream removal is being tested in Barkers Creek, South Canterbury (Burbery et al., 2020).

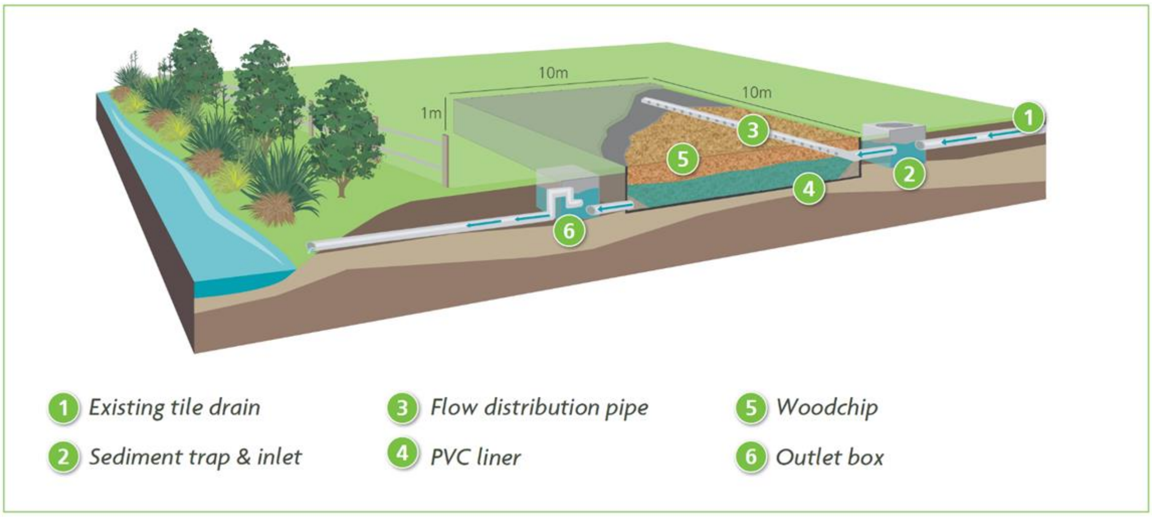

Figure 22: Schematic of a the "Nitrate Catcher" woodchip bioreactor (DairyNZ/Living Water/NIWA, 2015)

Figure 23. Grass-covered woodchip bioreactor treating tile drainage at Tatuanui, Waikato (Rivas et al. 2020).


Figure 24: Woodchip bioreactors treating tile drainage at Waituna, Southland (left; photo C. Tanner) and stream flow at Barkers Creek, South Canterbury (right; photo RadioNZ 2019).

Other porous reactive filter materials

A wide range of reactive media have potential to sequester phosphorus, metals, pesticides and micro-contaminants (e.g. endocrine disrupting chemicals) by absorption, adsorption, precipitation or transformation to different forms. These can be used as filter materials in porous beds, as substrates in constructed wetlands and adjuvants to bioreactor media (Ballantine and Tanner 2010, Haynes 2015, McDowell et al. 2008). For P-adsorption, grain size distribution, pH, specific surface area and the Al, Fe and Ca ions present are particularly important properties affecting performance (Cui et al. 2008). Porous media are susceptible to clogging with fine sediments so are most suitable for treatment of subsurface drainage. Given that reactive media have a finite performance capacity, their cost and local availability, as well as affinity and capacity, have a major impact on their practicality and cost-effectiveness (Ballantine and Tanner 2010, McDowell et al. 2013).

Summary

A range of potential edge-of-field, edge-of-waterway (riparian) and flow pathway options for mitigation of diffuse agricultural run-off are at varying stages of testing and readiness for application in New Zealand. Many of the options have the potential to reduce multiple contaminants and provide additional benefits (e.g. biodiversity). They are also amenable to being “stacked” (used in combination) with other mitigations to achieve desired water quality improvements (Goeller et al. 2020). Riparian buffers, constructed wetlands and detention bunds are relatively mature mitigation options, with improved guidelines currently in development. Care is needed to ensure that mitigations such as these do not result in unintended impacts or pollution swapping (Fenton et al. 2014, Rivas et al. 2020;) There is a lack of basic information on the flow and contaminant load characteristics of small waterway flows, in particular the importance of different flow pathways in different landscapes; the degree of aggregation and settleability of suspended solids in diffuse run-off; and the susceptibility of retained sediments to P loss. There is also a need for more on-farm demonstration and learning-by-doing to build the interest and capability of farmers, contractors, rural professionals and regulators.

Acknowledgements
This research has been supported by the Ministry for Business, Innovation and Employment, and NIWAs Strategic Science Investment Funding. We thank our many fellow researchers, land managers and farmers from New Zealand and around the world who have shared their knowledge and experience with us. Compiling this review of mitigation options was inspired by site visits hosted by SEGES and Aarhus University in Denmark in the company of Drs Ranvir Singh, Caroline Read and other Kiwis. We especially thank Drs Brian Kronvang, Charlotte Kjaergaard, Carl Christian Hoffmann, Mette Vodder Carstensen and Flemming Gertz for sharing their expertise. We are grateful to FLRC at Massey University for inviting this presentation and supporting the senior authors travel and attendance at the workshop.

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