Evidence that actions to mitigate contaminant losses from land via Farm Planning can improve water quality

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Executive Summary

This report outlines evidence for the ability of farm environment plans (and specifically those that could form Freshwater Farm Plans) to improve water quality. Surface and groundwater quality is influenced by contaminant losses from the farm, and variation caused by climate and time lags caused by the pathways that contaminants take from farms to waterbodies. Using data from 50 years of national and international studies, **we find clear evidence that after controlling for variation, we can attribute a degree of confidence in water quality improvements to actions designed to mitigate contaminant losses from farms**. The degree of confidence varies, depending on how, when and where data have been collected and is subsequently analysed. As a result, the evidence base spans a wide spectrum, from robust, well-funded farm- and catchment-scale studies that monitor water quality and actions across time and space, to well-intentioned but poorly designed studies that collect limited data at few sites over short periods, making it difficult to attribute observed changes to specific farm actions.

With the national implementation of Freshwater Farm Plans, there is a **clear opportunity to collect the right type and amount of farm-scale data to provide farmers, regulators and the community with confidence that their actions are leading to good water quality outcomes**. To support policy to achieve good water quality this report outlines evidence around:

- Current knowledge of the efficacy of actions to mitigate the loss of contaminants from land to water.
- The current level of implementing mitigation actions across different farming sectors
- Current knowledge of lag times between implementing actions (or changes in land use/intensity) and the detection of contaminants in groundwater and surface waters.
- The information needed to link observed improvements in water quality to the implementation of mitigation actions.
- Emerging science on the probability of detecting reductions in contaminant loss in receiving waters following mitigation efforts.
- Case studies tracing the implementation of mitigation actions through to their likely detection and attribution of water quality improvements in surface and groundwater.

To maximise the likelihood of implementing the right actions to improve water quality we provide a five-step framework. The framework is designed for farmers, industry bodies, regulators and the community to use as part of a collaborative catchment process focused on action. The process begins with the establishment of a water quality target (Step 1), which requires a plan to reduce contaminant losses—typically by a specified percentage of current levels. This report focuses on Steps 2 through 5, assuming that the water quality target has already been set.

The second step sees the land manager identifying actions for their farm plan. These are selected to mitigate the target contaminants, based on the suitability of actions for the farm. The third step ensures actions are implemented in the right place at the right time. This is determined by understanding the current catchment context and risk of contaminant loss, identifying hotspots of risk (e.g., critical source areas) within the farm, and applying suitable actions to maximise outcomes. Actions must be recorded. The fourth step sees appropriate monitoring put in place to connect

what's being done on farms with changes in the receiving water body. The final step assesses the level of confidence that improvements in water quality can be attributed to the implemented actions, after accounting for potential confounding factors such as climate variability and changing production. To ensure the success of each step we make the following recommendations:

Step 2

1. Farm planning tools should prioritise actions to be applied on farm in order from most to least cost-effective.

2. Implementation advice for land managers should also provide guidance on placing actions within critical source areas to maximise cost-effectiveness, speed and longevity of mitigation, co-benefits (including greenhouse gas emissions), and trade-offs.

Step 3

3. Policy should assess the proportion of contaminant concentration, load or yield attributable to farm management, by catchment, and focus actions on this manageable portion. Natural losses can be estimated for some river or groundwater types.
4. Farm planning tools must address both farm management practices (and actions) and inherent biophysical risks, because contaminant loss results from their interaction. Addressing only one factor will not improve catchment water quality or provide useful data.
5. Farm plan maps must identify areas at most risk of contaminant loss (e.g., critical source areas) and capture actions implemented in these areas along with their implementation date.
6. Farm planning systems tools must quantify how actions reduce risk and contaminant loss, to enable comparisons across space and time. Quantification is important if the data will be used to fairly allocate responsibility across farms for improving catchment water quality.
7. Regular auditing must be required by farm plan policy, to confirm that actions have been implemented on-farm to a standard that will improve water quality, and to determine changes in risk and contaminant loss. Records of where and when an action is implemented are critical.
8. Farm planning tools must directly match Good Farming Practice actions to risk assessment for contaminant loss, to ensure these actions contribute to achieving catchment water quality targets. Actions additional to Good Farming Practice guidelines may be required to meet catchment water quality targets.

Step 4

9. Regional councils or catchment groups should assess whether current water sampling frequency in a catchment is sufficient to reliably detect achievement of a water quality target, accounting for natural variability.
10. If the monitoring frequency results in too few samples to determine if detection is likely, councils or catchment groups should consider taking more samples, especially of the top 10% of flows (refer to monitoringfreshwater.co.nz).

11. Councils should maintain some long-term (20+ years) water monitoring sites to statistically control for the influence of a changing climate, lag times, and increasing production.
12. Public expectations about improvement in water quality should be managed with education about time lags. Until more data are available, expectations can be guided by statistical analysis of existing monitoring data, for catchments that have been monitored for long periods (e.g. 20+ years).

Step 5

13. Regional councils should continue to assess the effectiveness of actions using targeted catchment-scale studies in different catchments and under different conditions.
14. Before investing in catchment-scale studies, ensure information in the previous steps has been robustly determined. Studies can only detect the effect of actions where there are data on what actions were implemented, where, when, and their farm-scale effect.
15. Demonstrating farm plan effectiveness across multiple catchments over time requires comprehensive water quality monitoring that collects sufficient data to statistically control for confounding influences such as climate, lag times, and increased production.
16. Proving farm plan effectiveness requires consistent data collection and retention over many years (much like water quality data) to enable the attribution of water quality improvements to specific farm actions. This will require a long-term commitment by government, industry, and farmers to succeed.

By following these five steps—or appropriate variations—it is highly likely that improvements in water quality can be attributed to specific actions. However, this attribution depends critically on having sufficient data collected at the appropriate spatial scales and frequencies. While current water quality and quantity monitoring is generally robust, further optimisation is needed—for example, by balancing long-term sentinel sites with short-term monitoring locations to better detect and reinforce the impact of actions aimed at reducing contaminant loss.

A major barrier to effective attribution has been the limited availability of farm-scale data that quantifies both contaminant losses (e.g., from agricultural production) and the effects of mitigation actions. Our final recommendation is to ensure that farm-scale data in Steps 2 and 3 are collected in a rigorous and consistent manner—for example, through the implementation of Freshwater Farm Plans, following the five-step process outlined in this report—and made accessible to trusted parties for appropriate analysis. Such analysis will enhance our understanding of when, where, and how these actions reduce a farm's water quality footprint and potentially foster greater ownership of off-farm water quality outcomes.

Introduction

The New Zealand Landcare Trust records 223 catchment groups distributed across New Zealand [1]. Eighty seven percent of these groups are taking water quality samples and 73% are doing farm planning, funded over the last 5-6 years through $36M in investment from government [2]. These groups are supplemented or supported by primary sector groups and regional and central government. There is increasing interest in water quality and connecting this to on-farm actions to mitigate contaminant losses, especially since the New Zealand government announced that Freshwater Farm Plans would be a key mechanism to improve water quality [3]. However, while there is good guidance on creating and running a catchment group and encouragement from the primary sectors and government to implement actions, there is little guidance and evidence to link the implementation of actions to improved water quality. With the national implementation of Freshwater Farm Plans (thereafter referred to as farm plans), there is a **clear opportunity to collect the right type and amount of farm-scale data to provide farmers, regulators and the community with confidence that their actions are leading to good water quality outcomes**.

This report outlines evidence for the efficacy of farm plans to improve freshwater quality when used as part of a robust policy process. It is not intended to be a comprehensive technical report, capturing all evidence. Instead, it takes a fresh look at the data and evidence needed to give participants the confidence that improvements in river or groundwater quality can be attributed to the implementation of on-farm actions to mitigate water quality contaminant losses. These data and evidence are framed within a simplified five-step process (**Figure 1**). We outline the evidence for each step and provide clear recommendations for how each step can be sucessfully implemented. We also outline two case studies, one that is primarily groundwater focused and one that is primarily surface water focused.

This report assumes that Step 1 which sets a water quality target has already been set in the receiving river or groundwater, and quantified as a percentage reduction of a concentration (e.g., mg L^{-1}) or load/yield (e.g., kg nitrogen [N] yr^{-1} / kg nitrogen [N] ha^{-1} yr^{-1}) of contaminants like N, phosphorus (P), suspended sediment (SS) or the faecal indicator bacterium – *E. coli* at the farm scale. We make no comment about how this target has been set or allocated to individual farms. The same five-step process is applicable for lake or estuarine catchments.

Implicit in Step 1 is that all barriers [4] to recording farm and catchment scale data have been overcome such as:

- The cost of collecting robust data has been met.
- Data are open and able to be interrogated where applicable.
- Data are compatible across spatial scales.
- The lag time caused by the creation of policy and the implementation of actions has been minimimsed and is known, but biophysical lag between when actions were taken and likely seen in the water is unknown as is the influence of climate on contaminant losses.

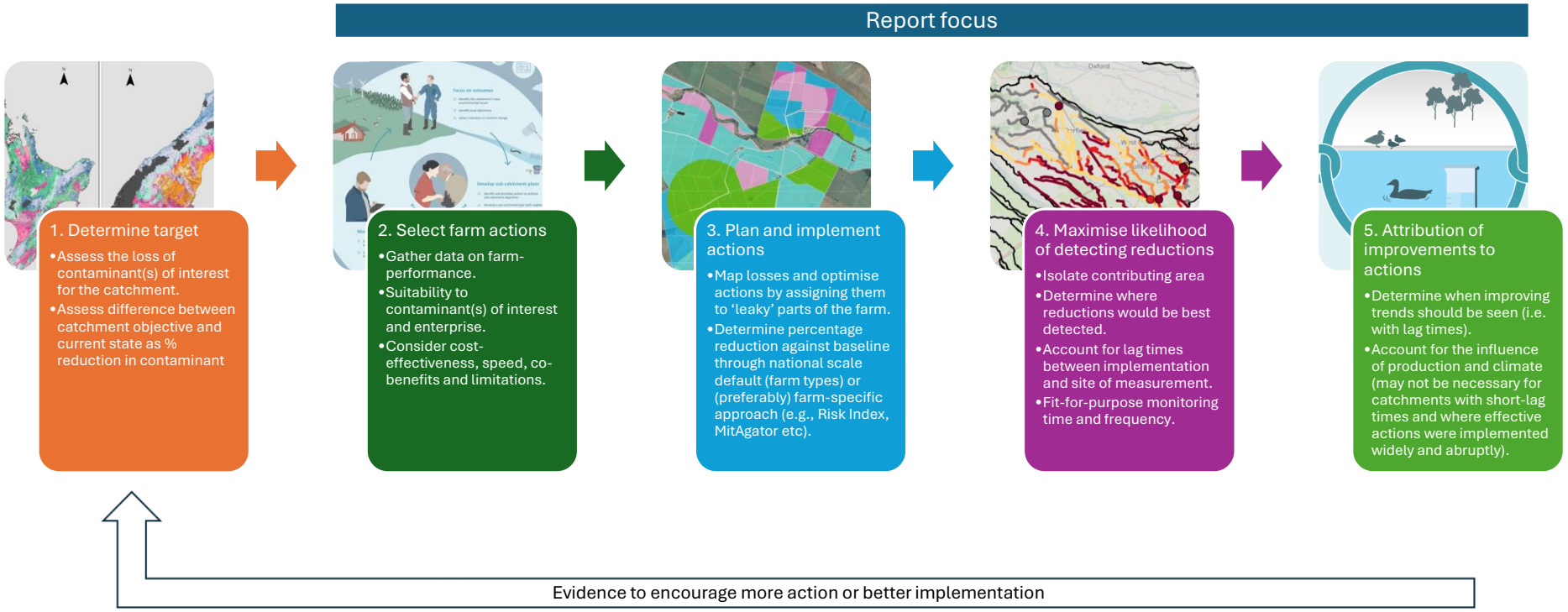


Figure 1. Five-step process for determining the efficacy of Freshwater Farm Plans to improve water quality.

Step 2. Selecting on-farm actions

Recommendations

1. Farm planning tools should prioritise actions to be applied on farm in order from most to least cost-effective.
2. Implementation advice for land managers should also provide guidance on placing actions within critical source areas to maximise cost-effectiveness, speed and longevity of mitigation, co-benefits (including greenhouse gas emissions), and trade-offs.

This section gives a summary of actions available for use in the primary sector to mitigate the loss of N, P, SS and *E. coli* from land to water. Although regulation exists around other contaminants (e.g., pesticides, heavy metals), we focused on these contaminants as they are the most often cited in policy [5], the primary sector [6,7], and by the public [8] as the cause of water quality problems. To provide a consistent approach across all actions, we only include actions that met the following criteria:

1. Data used to define the mean cost and the range of effectiveness for each action were available for three or more observations. Note that the authors' expert judgement has been used to classify the cost or effectiveness of an action where no quantitative data exist.
2. The mode and speed of remediation was known.
3. The action was applicable over a wide geographic range.
4. Where there was high variability in cost, effectiveness or speed of remediation, the reasons for the variability should be considered when assigning the applicability of measures to different farming systems and receiving environments.

Actions were grouped (Table 2) by location within an enterprise and the contaminant they mainly control. Actions were scored based on their cost and effectiveness to reduce each contaminant. Effectiveness was expressed as a percentage reduction in contaminant loads retained or reduced through actions. Cost was expressed in $ per ha (as of 2020) for the target contaminant but normalised to a loss of 1 kg of N, P or SS, and 10^{12} coliform forming units for *E. coli*. Recognising that the cost and effectiveness of actions vary widely depending on their location and over time, actions were classified into low, medium, or high categories. Actions were also categorised and assigned scores according to their likely remediation speed as slow, moderate and fast, which correspond to time periods required to achieve maximum effect of > 1 year, < 1 year, and < 1 season, respectively. These remediation speed categories reflect actions that could be incorporated as part of tactical (day-to-day) or strategic (seasonal to yearly) management decisions. The evidence is based, where possible, on multiple published, peer-reviewed journal articles. Where journal articles were not available, peer-reviewed grey literature has been used as cited in other studies [9-12]. For context, costs are given for establishment and maintenance for some actions in the Horizons region from Low et al. [12].

Factors to consider when choosing actions

Low cost, highly effective and fast actions to minimise contaminant losses are likely to be desired by land managers and investors. However, many other factors should be considered before actions are finalised. Examples of these factors are outlined below.

Applicability

Actions may be applicable to only one class of enterprise or receiving environment. For example, in grazed livestock systems, the use of low-rate dairy shed effluent application to land to mitigate contaminant losses is generally restricted to dairy farms [13].

Placement and timing

Targeting actions to leaky parts of a farm – often small parts of the farm typically referred as critical source areas (or hotspots) [14], greatly increases their cost-effectiveness [15,16]. McDowell [17] showed that across 14 sub-catchments farmed with sheep, red deer, beef or dairy cattle, the cost-effectiveness of actions targeted to critical source areas was 6-7 times higher than untargeted actions. In another example, placing land uses or actions (e.g., forage cropping) in the path of denitrification zones could reduce N load by 20-38% [18,19].

Co-benefits

Some actions reduce the loss, transport or impact of multiple contaminants. For example, many of the actions that target P and SS losses can also decrease *E. coli* loads. Richkus et al. [20] estimated that actions put in place to meet the total maximum daily load in the Chesapeake Bay for P and SS could decrease *E. coli* load by 19-27%. McDowell et al. [21] compared two approaches to mitigating N and P losses from grazed dairy farms in four regions of New Zealand: one to target a single contaminant with the most cost-effective action, and a generalised approach that considered costs, ease of implementation and their effectiveness for multiple contaminants. The targeted approach decreased losses of the target contaminant faster than the generalised approach, but other contaminants were not reduced. Targeting single contaminants could jeopardise water quality and ecosystem health should other contaminants become important downstream [22] or over time.

In addition to reducing the level of contaminant losses to water, it is also commonly assumed that many actions will have a co-benefit of reducing greenhouse gas emissions to air [23,24]. Studies show that when focusing actions on their potential to reduce contaminant losses to water, the co-benefit of the reduction on greenhouse gas emissions is less effective than focusing actions on reducing greenhouse gas emissions with a co-benefit of reducing contaminant losses to water. For example, in modelling the effect of 12 actions to mitigate N and P losses to water, Smith et al. [25] found that four of the actions (all focused on N losses) had the beneficial effect of reducing greenhouse gas emissions, six actions had no effect and two actions (the use of stand-off pads and deferred effluent irrigation; actions 18 and 31 in **Table 2**) increased greenhouse gas emissions. The effectiveness of actions also varies by sector with greater reductions possible in the sheep and beef

sector than the dairy or arable sectors when actions are focused on reducing greenhouse gas emissions (**Figure 1**), but greater reductions possible in the arable and horticultural sectors when focused on reducing contaminant losses to water (**Table 1**). These findings infer that if we are to avoid pollution swapping [26] then the co-benefits between contaminants and for emissions should be carefully considered when selecting suitable actions.

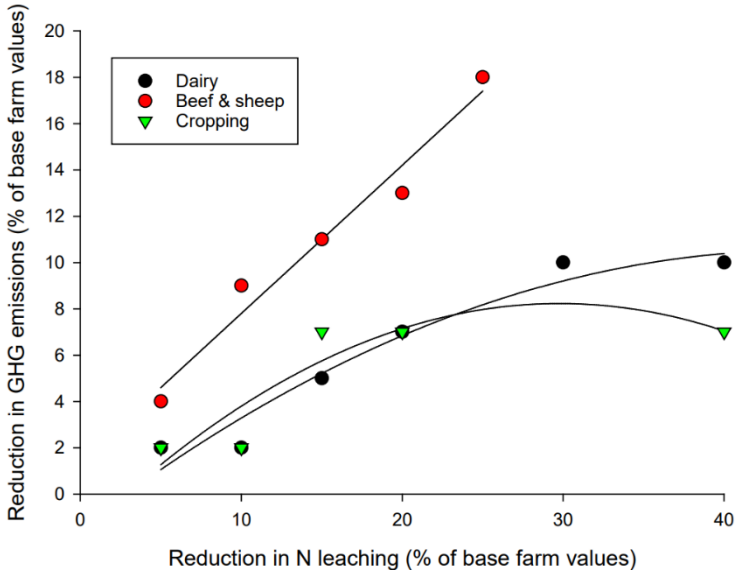


Figure 2. Modelled reductions in greenhouse gas emissions as a function of reductions in nitrogen losses to water when mitigations are chosen with reducing greenhouse gas emissions as the priority and nitrogen losses to water are the co-benefit. Data taken from Shepherd et al. [23].

Table 1. Percentage change resulting from the implementation of actions to mitigate the loss of nitrogen, phosphorus, sediment, *E. coli* and greenhouse gas ($CO_{2\text{-eq}}$). The numbers in parentheses refer to the actions numbered in **Table 2**. Data derived from Daigneault et al. [27].

Land use / Action	Nitrogen loss	Phosphorus loss	Sediment loss	*E. coli* loss	Greenhouse gas emissions
Dairy					
Effluent (19)	–4	–30	0	0	0
Riparian (11)	–56	–66	–75	–60	–2
Fencing (13)	–13	–15	–70	–60	0
Wetland (7)	–10	-45	–65	–55	0
Alum (3)	0	-26	0	0	0
Low water solubility P (23)	0	-10	0	0	0
VRI (26)	–10	0	0	0	0
Res Graz (31)	–36	–30	–40	–10	–10
Sheep and beef					
Riparian (11)	–56	–50	–75	–60	–2
Fencing (13)	–13	–15	–70	–60	0
Wetland (7)	–10	–45	–65	–55	0
Alum (3)	0	–26	0	0	0
Low water solubility P (23)	0	–10	0	0	0
Res Graz (31)	–16	–20	–10	-10	–6
Deer					
Riparian (11)	–51	–50	–82	–60	–2

Fencing (13)	–13	–15	–70	–60	0
Wetland (7)	–10	–45	–65	–55	0
Arable					
Riparian (11)	–51	–50	–75	–60	–1
Wetland (7)	–10	–45	–65	–55	0
Reduced tillage (49)	–2	–25	–25	0	–4
Minimum tillage (43)	–10	–50	–25	0	–20
Catch crop (45)	–60	–25	–10	0	–20
Horticulture					
Riparian (11)	-51	-50	–75	–60	–1
Wetland (7)	-10	-45	–65	–55	0
Limit N App	-4	0	0	0	0
Reduced N fertiliser (21)	-10	0	0	0	–3
Catch crop (45)	-5	-25	–25	0	–10
Reduced tillage (49)	-5	-25	–25	0	–4

Duration

Actions should maintain their efficacy over long periods. For example, the efficacy of wetlands to remove P from flowing waters decreases as wetlands fill with sediment (unless periodically dug out), or anaerobic conditions develop that remove N via denitrification desorb sediment-bound P into the water column [28]. Changes may also occur due to climatic variability, that may induce more frequent and intense storms that deliver greater contaminant yields [29,30]. Depending on the magnitude of changes in efficacy over time, the cost-effectiveness of actions may be compromised. Indicators, such as a significant change in the load of contaminant lost from different land uses should be used as a trigger to determine if additional trials are required to assess whether efficacy changes appreciably under a changed climate.

Uncertainties in cost-effectiveness

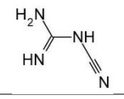
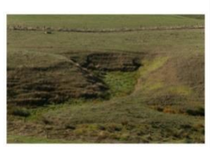
A poor understanding of contaminant sources and pathways and impacts of contaminants on receiving environments can result in actions being placed in the wrong place and used at the wrong time. This increases the uncertainty in the estimates of cost-effectiveness of actions, and therefore uncertainty when recommending actions [31]. To accommodate some uncertainty, but still allow decisions to be made, actions should only be implemented when there is prior evidence of cost-effectiveness under similar conditions. Further accommodation is made by reporting the costs and effectiveness of actions as ranges [32-34], which recognises that despite having similar conditions, the performance of a given action will vary.

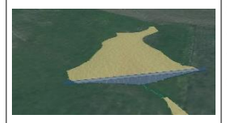
Unintended consequences and trade offs

The unintended consequences associated with implementing actions could outweigh their benefit. For instance, the nitrification inhibitor dicyandiamide (DCD) was used on grazed and fertilised pastures to mitigate the leaching of nitrate-N from New Zealand soils. However, DCD can leach from soils to groundwater and emerge in surface water, where suppression of nitrification can lead to

enriched ammonia concentrations [35]. In meeting the bioenergy goals of the United States, Adusumilli et al. [36] estimated that significant increases in erosion and nutrient loading was likely if pastureland was converted to intensive biomass production for biofuel. To minimise unintended consequences, it is important to increase our knowledge of the potential negative side effects of actions.

Table 2. Master table outline of farm-scale actions to mitigate the loss of contaminants to water (*E. coli* = *Escherichia coli*; N = nitrogen; P = phosphorus; SS = suspended sediment).

Class / Targeted contaminant	Suitable for	Strategy	Description of function	Effectiveness (%)[1]	Relative cost ($/ha/yr)[2]	Response rate[3]	Co-benefits	Reasons for variability	Factors limiting uptake	Example references
Additives										
1. Nitrogen	All pastoral farming enterprises	Diuretic supplementation or additive like salt to dilute urinary N	Diuretics such as common salt generally result in increased water consumption by animals with an associated increase in the spread of urinary N by the animals.	Low	Low	Fast	May also lead to decreased nitrous oxide (greenhouse gas) emissions	Animal avoids eating supplementation or N-modifier loses efficacy over time.	Salt is more appropriate in well-structured soils for long-term use since excess sodium in soil can potentially lead to soil structure degradation.	37
2. Nitrogen	All pastoral farming enterprises	Nitrification inhibitors (Dicyandiamide, DCD) 	Prevents conversion of urinary-N into nitrate allowing for more plant-N uptake and less N leached.	High	Mod	Mod	DCD decreases the emission of the greenhouse gas nitrous oxide by 40 to 80%	The efficacy of DCD depends on how long after the urination event the DCD is applied and once applied, the effectiveness depends on soil type, temperature and moisture.	Currently not permitted for use.	38-40
3. Phosphorus	All pastoral farming enterprises	Applying alum to forage cropland or pasture	P-sorbing aluminium sulphate (alum) sprayed onto a winter forage crop or pasture just after grazing to prevent surface runoff losses of P.	Mod	Mod	Fast	None	May be ineffective in high rainfall environments where it may be washed from the soil.	Few supplies and competing use as a water treatment additive. Perception (false) that adding Al to soils is toxic.	41-43
4. Phosphorus	All pastoral farming enterprises	Red mud (bauxite) to land	P-sorbing and caustic red mud (Al-oxide) incorporated into soil at rate of up to 20 Mg ha^{-1} to prevent P losses in runoff and leaching.	High	Mod	Fast	Alkaline and hence can be used instead of lime.	Increases soil pH which may increase P solubility if outside pH range 5.5-5.9.	Few suppliers. If used for liming effect, grazing animals need to avoid treated area otherwise ingestion may impair rumen function.	44
5. Phosphorus	All farming enterprises	Artificial drainage amendments 	Use of P-sorbing Ca, Al and Fe materials as backfill for artificial drainage systems. Usually accompanied by a gradient of different sized stone chip to aid in the filtering of large particles around the pipe.	High	Low	Fast	Potential to decrease (via filtration) the loss of sediment and faecal bacteria (both unquantified).	Quantities of sorbing materials (e.g. Al, Fe and Ca). Material needs to be coarse enough to maintain good flow and interaction with material	Source may be far away and the cost of transport prohibitive	28,45
Edge of field										
6. Multiple	All farming enterprises	Constructed wetlands  Taken from http://www.stuff.co.nz/business/farming/91113123/owl-farm-wetland-removes-most-nitrates-in-first-water-samples	Modification of landscape features such as depressions and gullies to form wetlands. Slow water movement encourages deposition of suspended sediment and entrained contaminants (e.g. P). Assumes mean annual air temp >12°C and excludes highly permeable soils not able to sustain a wetland.	High [N]; Low [P]; Mod [SS]	Mod [N]; High [P]; Low [SS]	Mod [N]; Fast [P, SS]	Flood attenuation, aesthetic quality, wildlife habitat and biodiversity	Wetland performance depends on intercepting the maximum amount of runoff from the catchment at the right flow rate. For small (~1%), medium (~2%) and large (~4% of catchment) wetlands in the North Island, performance is 25, 36 and 48% reduction of runoff N and 18, 26 and 36% in the South Island.	No suitable areas on farm (i.e. catchment lies outside of farm area); cost; regional regulations.	46
7. Multiple	All farming enterprises	Natural seepage wetlands  Taken from https://www.dairynz.co.nz/media/5787389/making-the-most-of-wt-areas-on-farm.pdf	Natural seepage wetlands at the heads and sides of streams, commonly known as seeps, flushes, valley bottom or riparian wetlands. Wetlands slow water movement through them and encourage the deposition of suspended sediment and entrained contaminants. Seepage of nitrate-rich water through organic soils promotes effective nitrate-N removal via denitrification. Assumes that catchments are approximated by a block.	High [N]; Low [P]; Mod [SS]	Mod [N]; High [P]; High [SS] Maintenance cost $10-13 ha^{-1} yr^{-1}	Fast [All]	Flood attenuation, wildlife habitat and biodiversity	Interception of runoff depends on landscape, hydrogeology and human modification.	Price of permanent fencing >> temporary fencing.	46,47

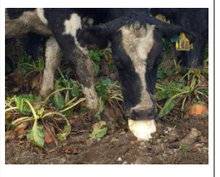
Class / Targeted contaminant	Suitable for	Strategy	Description of function	Effectiveness (%)[1]	Relative cost ($/ha/yr)[2]	Response rate[3]	Co-benefits	Reasons for variability	Factors limiting uptake	Example references
8. Multiple	All farming enterprises and forestry	Sediment traps  Taken from: https://www.dairynz.co.nz/media/254172/5-9_sediment_traps_2012.pdf	Excavation of the stream bed or an ephemeral channel allows coarse sized sediment and associated N and P to settle out during flow. Recommended capacity is 0.5% of catchments <5 ha and 1% for catchments >5 ha.	Low [P]; High [SS]; Low [*E. coli*]	High [P]; Mod [SS]; High [*E. coli*] Cost for establishment $750-1,500 ha^{-1}, maintenance is $75 ha^{-1} yr^{-1}	Fast [P, SS, *E. coli*]	Potential to buffer storm events and therefore potential downstream flooding. May also prevent stock losses if associated with fencing and ephemeral channel.	Ineffective at high flows when most sediment is transported or for finely textured soils. Recommended capacity is 50 m^3 ha^{-1} for catchments <5 ha and double that for catchments >5 ha.	May require resource consent	48-50
9. Multiple	All farming enterprises and forestry	Detainment bunds / retention ponds  Taken from: https://catchmentsolutions.co.nz/detainment-bunds/	Detention bunds coupled with retention ponds allow coarse sized sediment and associated N and P to settle out. Have 120 m^3 of storage volume per ha of contributing catchment, i.e., 1.5% of catchment with a 0.8 m average pond depth.	Mod [P]; High [SS]; Low [*E. coli*]	High [P]; High [SS]; High [*E. coli*]	Fast [P, SS, *E. coli*]	Potential to buffer storm events and therefore potential downstream flooding.	Although design can be modified to maximise removal via settling, traps are ineffective at high flows when most sediment is transported	May require resource consent	51,52
10. Multiple	All farming enterprises and forestry	Grass buffer strips	Vegetated buffer strips (> 5 m width) work to decrease contaminant loss in surface runoff by a combination of filtration, deposition, and improving infiltration	Mod [P]; Mod [SS]; Mod [N]	High [P]; Mod [SS];	Fast [P, SS]	Potential to stabilise stream banks.	Strip can become clogged with sediment and function poorly in areas that are often saturated due to limited infiltration or where surface runoff converges and breaks through strips. N performance ranges: 10-82% 2-5% hillslope length 30-66% >5% hillslope length	Land adjacent to stream may not be available or suitable for a buffer strip. Cannot intercept artificial drainage. Takes land out of production. May require weed control.	49,53,54
11. Multiple	All farming enterprises and forestry	Riparian buffer strips  Taken from Heubeck et al. [55]	Riparian buffer strips with trees and shrubs work to decrease contaminant loss in surface runoff by a combination of filtration, deposition, and improving infiltration. Installed where there is a shallow confining layer <2m below soil surface. Where unbridged crossings exist, they have the potential to be a large source, and covering these crossings with culverts or bridges will likely make important reductions in the loads of N, P and sediment loss from farms.	Mod [P]; Mod [SS]; Mod [N]	High [P]; Mod [SS];	Fast [P, SS]	Potential to stabilise stream banks. Provides stream shading, improved aquatic habitat, carbon supply to streams, landscape aesthetics, recreational and cultural benefits.	Strip can become clogged with sediment and function poorly in areas that are often saturated due to limited infiltration or where surface runoff converges and breaks through strips. Nitrogen performance ranges by soil texture: 40-70% coarser than sandy soil 70-90% sandy loam or finer	Land adjacent to stream may not be available or suitable for a buffer strip. Cannot intercept artificial drainage. Takes land out of production. May require weed control.	56
12. Multiple	All pastoral farming enterprises	Bridging stream crossings	Avoids direct entry of faeces, urine and entrained hoof mud, and substrate disturbance during stock stream crossings.	Low [N]; Low [SS]; Mod [*E. coli*]	Mod [N]; Mod [SS]; Low [*E. coli*]	Fast [All]	Avoiding stock losses in high flows.	Highly dependent of stream length, width and number of crossings per farm and the number of times stock cross them (the closer the bridge is to key infrastructure the more effective they will be) Effectiveness can be enhanced by bunding the edges to direct contaminants back to land.	Too many crossings; regional regulations.	57

Class / Targeted contaminant	Suitable for	Strategy	Description of function	Effectiveness (%)[1]	Relative cost ($/ha/yr)[2]	Response rate[3]	Co-benefits	Reasons for variability	Factors limiting uptake	Example references
13. Multiple	All pastoral farming enterprises	Stream fencing  Taken from http://www.ruralnewsgroup.co.nz/item/12009-new-stock-exclusion-rules-require-greater-flexibility-feds	Preventing livestock access to stream, decreases stream bank damage (and sediment inputs via bank erosion) bed disturbance of sediments (and entrained *E. coli*, N and P) and stops the direct deposition of excreta into streams.	High [P]; Low [SS]; Mod [*E. coli*]	Low [P]; Low [SS]; Low [*E. coli*]	Fast [P, SS, *E. coli*]	Stock safety if banks are steep and streams swift. Bank stabilisation. Can also have wider benefits if coupled with riparian planting (see above).	Gain is dependent on the area of the farm currently unfenced and stream density.	Price of permanent fencing >> temporary fencing. Temporary fencing may require a change in management practice for some farmers. Can take land out of production. Uptake could be enhanced by augmenting fencing with collars (viz. virtual fencing) to direct stock away from streams.	58-60
14. Nitrogen	All farming enterprises	Denitrification beds	Large containers filled with woodchips that intercept drain flow and denitrify nitrate in water to nitrogen gas which is released to the atmosphere. Assumes Denitrification Bed 1 m deep ~1% of catchment area. Assumes that catchments are approximated by a block. Removal range is 0.1-0.8 (mid-point) of 0.5, but we assume artificial drainage captures half of the N leached	High	High	Mod	Might be integrated to support dissolved P removal	High cost when bioreactor was underloaded. True value much more likely to be at lower end when systems properly designed	Appropriate hydrology needed - tile/sub-surface drained land or small surface drains.	61-63
15. Sediment	Cropping	Benched headlands	Constructed level bench that runs across the slope of a field. Suitable for use on cultivated soil where slopes are greater than 3 degrees. These encourage infiltration of water on the bench and reduce the slope length of water pathways.	Mod	Low Cost \$65 ha^{-1}	Fast	Grassed headlands protects from scouring and encourages silt to settle-out before entering drains. Decreased P input (unquantified) to waterways.	Depends on infiltration capacity of soil	Management expertise	64
16. Sediment	Cropping and Forestry	Bunds  Taken from: http://www.hortnz.co.nz/assets/Uploads/Auckland-Waikato-ES-Control-Guidelines-1-1.pdf	Earthen barrier constructed along paddock edge to prevent water flowing onto or from field. Suitable for use on cropping land with slope greater than 3 degrees. Creates ponds of water at bottom of field where sediment settles out.	High	Mod Cost \$130 ha^{-1}	Fast	Increased sustainability of cropping. Decreased P input (unquantified) to waterways.	Depends on infiltration capacity of soil. Recommended to have 50 m^3 ha^{-1} for catchments <5 ha and double that for catchments >5 ha.	Management expertise to ensure structures maximise the time available for sediment to settle.	49,65
17. Sediment	Cropping and Forestry	Silt fence  Taken from: http://esccanterbury.co.nz/wp-content/uploads/2017/02/sc-super-silt-fence.jpg	Material (e.g., geotextile cloth) fastened to a wire fence for filtering out sediment from surface runoff.	High	High Cost \$380 ha^{-1}	Fast	Decreased P input (unquantified) to waterways.	Variability of material and contracting costs	Other actions may be more suitable if capturing losses from large area (e.g., detainment bunds).	49,65
Effluent / fertiliser										

Class / Targeted contaminant	Suitable for	Strategy	Description of function	Effectiveness (%)[1]	Relative cost ($/ha/yr)[2]	Response rate[3]	Co-benefits	Reasons for variability	Factors limiting uptake	Example references
18. Multiple	Dairy	Deferred effluent application and greater effluent pond storage.  Taken from: Retrieved from http://www.ruralnewsgroup.co.nz/dairy-news/dairy-management/treat-poo-as-fert	Effluent is stored in larger ponds allowing it to be applied to land only when both soil moisture deficit and physical conditions allow it to be absorbed/retained in the soil and not lost as runoff	Mod [N]; Mod [P]; Low [*E. coli*]	Mod [N]; Low [P]; Mod [*E. coli*]	Mod [All]	Added water and carbon during summer and decreased (but unquantified) *E. coli* losses. If more nitrogen and phosphorus is adsorbed in the soil it will reduce fertiliser requirements.	Depends on the number of cows, size of pond required, material and suitable location to build a pond. Inaccurate pond size can result in applications during wet periods and N and P losses. Differs with soil types and drainage status.	The requirement for storage is dictated by local climate and if too wet may make practice very expensive and/or require measures to reduce shed water use.	66
19. Multiple	Dairy	Low-rate effluent application to suitability-sized land	Coupling pond storage with low rates of effluent application and larger areas can decrease N, P and *E. coli* loss by spreading effluent wider and minimising the potential for surface runoff and sub-surface losses via preferential flow.	Mod [N]; Mod [P]; Low [*E. coli*]	Mod [N]; Low [P]; Mod [*E. coli*]	Fast [All]	Added water and carbon during summer better matched to pasture growth. Also reduces the potassium being applied to the soil which reduces the risk of developing animal health concerns from too much potassium in the soil.	The requirement for solid separation (using low-rate sprinklers) and degree of existing infrastructure that is already suitable. Differences due to soil types and drainage status.	Cost; availability of suitable land. Increased labour requirements if using sprinklers compared to travelling irrigator.	66-68
20. Multiple	Dairy	Enhanced Pond Systems	Covered Anaerobic Ponds to remove and digest organic suspended solids to methane-rich biogas for energy recovery. High-Rate Algal Ponds remove N and P in harvest algae. Maturation Ponds also remove faecal contaminants as indicated by *E. coli*.	High [N]; Mod [P]; High [*E. coli*]	High [N]; High [P]; High [*E. coli*]	Fast [All]	Energy recovery / production. Fewer greenhouse gas emissions.	Removal efficiency varies seasonally so designed for winter performance specifications and have higher performance in summer.	Requires substantial land area (10 to 40 m2/cow)	69
21. Multiple	All pastoral farming enterprises	Improved N use efficiency	Efficiency gains via i) reduced use of N fertiliser especially in cooler months and for forage crops coming out of long-term pasture; and ii) avoiding excessive N inputs to effluent blocks.	Low	Low	Mod	Decrease emissions of greenhouse gases and an improvement in energy use	The ability to decrease N losses to water depends on (i) the existing level of farm intensity and N loss, and (ii) the management expertise to implement required changes in farm practices. Poor response if fertiliser is applied <6°C.	Expertise is required to maximise harvested feed under a low input farming system, while an increase in per cow production (to allow a decrease in stocking rate) takes time as improved genetics is introduced into herds.	70,71
22. Nitrogen	All farming enterprises	Buffer width for fertiliser application	Implementing a minimum 10m buffer width between the ground application of fertiliser and open water.	Mod	Low*	Fast	Avoids the cost of wasted fertiliser.	Can be ineffective in windy conditions.	GPS in areas where streams are not easily visible.	72
23. Phosphorus	All farming enterprises	Low water-soluble P fertiliser	Less fertiliser-P is lost in runoff due to the low water solubility of products such as reactive phosphate rock resulting in increased P use efficiency.	Mod	Low	Mod	Has a slight liming effect.	Gain compared to highly water-soluble P fertiliser is dependent on time of year that fertiliser is applied. Larger gains are evident where the coincidence of surface runoff soon after application is frequent.	Soil pH < 6.0, rainfall > 800 mm. Also, cannot be used for capital applications and must gradually replace maintenance highly-water soluble P applications at a rate of one-third per annum (i.e. 100% low water -soluble P in year 3)	73-75
24. Phosphorus	All farming enterprises	Optimum soil test P concentration	Matching soil Olsen P concentrations to pasture and forage crop requirements avoids enriched soil P concentrations that are more likely to lose more P in runoff compared to that at an agronomic optimum concentration.	Low	Low	Slow-Fast	None	Gain is dependent on soils being enriched beyond their optimum and method of decrease e.g. natural depletion = slow while cultivation = fast	May require a large amount of sampling (on a georeferenced grid) to control for spatial variation in soil types and P concentrations.	76
Irrigation / drainage										

Class / Targeted contaminant	Suitable for	Strategy	Description of function	Effectiveness (%)[1]	Relative cost ($/ha/yr)[2]	Response rate[3]	Co-benefits	Reasons for variability	Factors limiting uptake	Example references
25. Multiple	All farming enterprises and forestry	Cut outs or berms (and amendments) to filter and direct laneway or stockyard runoff away from waterways  Taken from: McDowell et al. [77]	Direct water off laneways, near stockyards or recently cultivated paddocks through a filtration system and away from waterways. Implementation assumes the presence of one laneway (used daily) or stockyard per 30 ha currently discharging into a waterway i.e., the effect of N/P-rich excreta in runoff is diluted by runoff from the rest of the ~30 ha catchment.	High	Low	Fast	Avoids water ponding on laneways or stockyards.	Discharge may occur if flow accumulates from a large impervious area (e.g., laneway)	Unable to create a berm directing water to an area unlikely to enter a stream.	49,77,78
26. Nitrogen	Irrigated land	Precision application of fertiliser and variable rate irrigation	Sensors and automation of irrigation and nutrient inputs optimises crop utilisation at fine scales.	Mod	Low	Fast	Improved farm and herd management; improved crop reliability and quality; conservation of water.	Effect improves with soil heterogeneity, past mismatches between nutrient inputs and requirements and farmer skill.	Insufficient communication and training on benefits. Capital cost of fitting VRI and limited technical support in NZ.	79,80
27. Nitrogen	All farming enterprises	Controlled drainage  Taken from: http://johnstoneng.co.nz/wp-content/uploads/2015/05/Retaining-Walls-and-Drainage.jpg	Delaying drainage to allow for sedimentation and denitrification. Only suitable for cropping areas where soil moisture deficits are strong enough for crops to benefit from increased moisture.	Mod	High[4]	Med	Improved maize yield in autumn.	Water levels must be kept > 20-30 cm from the soil surface to avoid pugging and anaerobic water intersecting with P-rich topsoil resulting in dissolution of P and P-rich drainage.	Only applicable to flat land and cropland (e.g., Maize).	81,82
28. Phosphorus	Irrigated land	Refurbishing and widening flood irrigation bays	Water exiting flood irrigation bays as outwash represents about 20-50% of that applied. Re-contouring irrigation bays, and/or preventing outwash/wipe-off from accessing the stream network decreases P loss.	High	Mod	Mod	More efficient use of flood irrigation water.	Inaccurate level resulting in flow (and outwash) faster than anticipated. Variation in the water supply rates.	A move to spray irrigation is likely to be more cost-effective, however significant capital cost involved	66
29. Phosphorus	Irrigated land	Dams and water recycling	Recycling systems divert irrigation outwash for use in others part of the farm increases nutrient use efficiency.	High	Mod	Mod	More efficient use of flood irrigation water.	Leakage from infrastructure	A move to spray irrigation more cost-effective.	83
30. Sediment	Cropping	Contour drains	Temporary drains that run across the slope of a field and into a permanent drain on the side of the field reduce the slope length of water pathways and thereby reduce the eroding power.	Mod	Low Cost $75 ha^{-1}	Fast	Increased sustainability of cropping. Decreased P input (unquantified) to waterways.	Depends on density of drains. Efficiency decreases with increasing slope.	Management expertise	84
Stock management										

Class / Targeted contaminant	Suitable for	Strategy	Description of function	Effectiveness (%)[1]	Relative cost ($/ha/yr)[2]	Response rate[3]	Co-benefits	Reasons for variability	Factors limiting uptake	Example references
31. Multiple	Dairy	Restricted grazing and off pasture animal confinement systems  Taken from: https://www.sheds.co.nz/tools/blog/wintering-barns-and-dairy-sheds/	Reduced direct deposition by grazing animals and surface erosion is coupled with a stand-off area to reduce losses during heavy rainfall. Includes covered and uncovered stand-off areas, feed pads and composting barns.	High [N]; Mod [P]; Low [SS]	High [N]; High [P]; Low [SS] Cost $1,500-2,000 cow^{-1}	Mod [All]	Decreased soil and pasture damage and N2O emissions.	Costs vary widely due to type of facility, variations in soil type and climate, and on the frequency of use of a restricted grazing strategy. Anecdotal evidence covered stand-off areas/composting barns can improve feed conversion efficiency and if used over dairy cow calving, can improve animal health outcomes.	High capital and operational costs and increased management complexity; potential animal welfare and manure management issues due to the proximity of animals. Must be accompanied by a stand-off area that has no connection to a waterway (e.g. runoff/effluent is captured).	85-87
32. Multiple	Deer	Alternative wallowing	Fencing off existing connected wallows and the creation of a wallow that is not connected to a stream.	Low [N]; High [P]; Mod [SS]	High [N]; Low [P]; Low [SS]	Fast [All]	Allowance for natural behaviour may decrease stress (unquantified).	Poor performance could occur if runoff from alternative wallow reaches stream in large storms.	There must be an area close by that is suitable for an artificial wallow.	88
33. Multiple	Deer	Preventing fence-line pacing	Tree planting to provide shelter and maintaining sufficient feed to avoid stress when, for example, when feed is low or near calving.	Low [P]; Low [SS]	High [P]; High [SS]	Mod [P, SS]	Trees decrease stress and may have anthelmintic properties (if grazed).	Planting, maintenance and effect of tree planting is subject to climatic influences (primarily wind direction).	Supplying sufficient feed to avoid animal stress is dependent on the skill of the farm manager.	89
34. Multiple	All pastoral farming enterprises	Strategic grazing of forage crops in critical source areas (CSAs)  Taken from: from http://www.agresearch.co.nz/news/trial-suggests-winter-management-can-cut-runoff-losses/	Grazing down slope/grazing gullies last in winter to ensure soil damage in these areas was minimized when grazing does occur.	High	Low[4]	Fast [N, P, SS, *E. coli*]	Improved stock health from drier ground.	Effect varies according to the likelihood of runoff-producing rainfall events in spring. Seasonal conditions, e.g. extreme rainfall events require alternate animal management plan	Critical source areas are difficult to get to and move stock. Requires additional labour to shift stock.	90
35. Multiple	All pastoral farming enterprises	Increase rate of finishing, early culling in autumn	Increase rate of finishing or culling (in autumn) to remove stock from the farm faster	Low[4]	Low[4]	Mod	Increase in feed supply to remaining stock	-	Potential lost productivity (especially for dairy production).	91
36. Nitrogen	All pastoral farming enterprises	Including plantain in the diet  Taken from: from http://www.stuff.co.nz/business/farming/agribusiness/74661433/inverary-station-team-runs-the-rule-over-its-farm-performance	Results in lower N concentration in urine than cows grazing perennial ryegrass/white clover pastures. Also thought to inhibit nitrification. Reduces N loss by 1% for every 1% of plantain in diet up to a maximum of 20% plantain. Effect assumes 15% of diet is plantain in a well-kept sward over 7 years.	Low	Low[4] Plantain seed costs around $30 kg^{-1}	Mod	Improved stock health.	Inability to maintain 15% of plantain in sward owing to competition by other species and susceptibility to broadleaf herbicides.	May need to re-sow pastures more frequently.	92-96

Class / Targeted contaminant	Suitable for	Strategy	Description of function	Effectiveness (%)[1]	Relative cost ($/ha/yr)[2]	Response rate[3]	Co-benefits	Reasons for variability	Factors limiting uptake	Example references
37. Nitrogen	All pastoral farming enterprises	Including chicory in diet  Taken from: https://www.dairynz.co.nz/media/cxejzxyg/management-of-choice-chicory-crops.pdf	Reduced nitrate-N leaching owing to scavenging of N from deep taproot. Chicory grows better than ryegrass in summer. Can finish stock quicker over summer/autumn (see #35, above)	Low	Low[4] Chicory seed costs $35 ha^{-1}. Establishment cost $1,000-1,500 ha^{-1}	Mod	Improved stock health (better quality feed than plantain)	Effectiveness varies with climate (especially summer-winter).	Cannot be grazed in winter and can yield less than plantain. Persistence can be an issue, as can managing quality	94,97,98
38. Nitrogen	All farming enterprises	Using winter active crops  Taken from: https://www.dairynz.co.nz/media/4439057/technical-series-june-2016.pdf	Such as an annual ryegrass, Italian ryegrass and some late maturing perennial ryegrasses grow during winter and utilising soil N when leaching is likely. Effect is highly dependent on cultivar. Data shown for cv. Tabu.	Low	Low[4] Cost $450 ha^{-1} plus sowing	Fast	May reduce sediment loss compared to fallow, but could also increase sediment, P and *E. coli* loss if grazing is increased during wet conditions.	Only applicable to flat land that is dry enough to sow winter active pastures - must be resown as usually an annual crop.	That the wrong cultivar is chosen.	99-101
39. Nitrogen	All pastoral farming enterprises	Winter/summer fodder beet  Taken from: https://www.dairynz.co.nz/news/getting-the-best-from-beet/	Urinary-N is dilute (3 g N L^{-1}) compared to other brassica fodder crops or ryegrass (7 g N L^{-1})	Mod	Low Cost $3,000 ha^{-1}	Mod	Can be fed in-situ of lifted and fed on feed pad. High cost of establishment is offset by high yield and ability to put weight on before winter.	Effectiveness is lower if grazed in autumn.	Requires a free-draining soil and careful feed management to avoid issues like acidosis (it is recommended to maintain at least 30% pasture silage/baleage in diet to meet fibre and protein requirements).	102,103
40. Nitrogen	All pastoral farming enterprises	Change animal type	Animal type influences N leaching due to inherent differences in the spread of urinary N (the major source of N loss in grazed pastures). N leaching from sheep and deer is approximately half that from beef cows at the same level of feed intake.	High	Low	Slow	This may also lead to decreased N2O emissions. However, a change to deer may lead to greater sediment and P loss.	Highly variable over time due to changes in relative prices between cattle and sheep meat and the suitability of the landscape.	Changing economic returns matching feed supply and demand, diversification of animal types and financial resilience over time; possibly a need for a mix of animals on a farm; and better farm management skills and farm infrastructure (e.g. extent of fencing).	104-106
41. Nitrogen	All pastoral farming enterprises	Supplementary feeding with low-N feeds	Lowering the protein concentration of supplementary feed to decrease N surplus and N excreted in urine and lost via subsurface flow.	Low	Low	Mod	May also lead to reduced N2O, but greater CO2 production in the production and feeding of the low-N feed sources.	Depends on source and price of feed and the efficiency with which it is fed to animals. Carbon footprint of feed and sustainability of supply chain.	Lack of facilities for feeding out supplementary feed and costs of introducing them; requirement for increased skills in feed utilisation; and increased risk, depending on milk or meat payout and feed prices	70,107,108
42. Nitrogen	All pastoral farming enterprises	Genetic improvement	Factors that affect longevity of animal lifetime acts to reduce N in urine by 6-20%: Effect may parallel protein efficiency low breeding worth = 0.28 g milk solids g^{-1} protein and high breeding worth = 0.30 g milk solids g^{-1} protein	Low	Low	Slow	Potential for lower stock numbers may also decrease other contaminants.	Factors include increase lambing percentages and better fertility in cattle.	Access to high breeding worth stock.	109,110

Class / Targeted contaminant	Suitable for	Strategy	Description of function	Effectiveness (%)[1]	Relative cost ($/ha/yr)[2]	Response rate[3]	Co-benefits	Reasons for variability	Factors limiting uptake	Example references
Cropping and cultivation										
43. Multiple	All farming enterprises	Direct drilling	Avoids soil N mineralisation (hence no benefit to perennial pasture) but prevents soil disturbance, increasing roughness and likelihood of soil loss via erosion compared to conventional tillage.	Mod	Low	Fast	-	Reduction in particulate N assumed to be 60% and particulate N assumed to be 50% of runoff total N.	Must have appropriate machinery and relies on use of glyphosate	111
44. Multiple	All pastoral farming enterprises	Grow maize on effluent block	Allows lower cost maize growth on farm with no fertiliser for at least two years after pasture.	Low[4]	Low[4]	Mod	-	Assumes linear relationship between N / P fertiliser application and leaching / runoff loss.	-	112,113
45. Nitrogen	All farming enterprises	Catch cropping	Typically, short rotation crops with good cool season growth and a deep rooting system that helps to mop up N that would otherwise be leached.	Mod	Mod[4]	Fast	Reduces sediment loss compared to fallow.	Effectiveness is dependent on when crops are sown in relation to grazing/N loading or harvest. Catch crops generally feature in two main systems: 1) summer/early autumn (Mar; 50%) and late autumn (May 10%) cropping (S1), and 2) following winter forage crop grazing depending on the month sown (30, 20 and 10% in Jul, Aug, and Sep, respectively.	Cool weather may impair catch crop growth. wet soils prevent planting/need for specialist drills in heavy and wet soils	103,114,115
46. Sediment	All pastoral farming enterprises	Soil conservation plan to plant trees	Combination of retirement and pole planting on highly erodible land. Introduction of tree roots to soil regolith protects soil on steep slopes from mass movement erosion.	Mod	High	Slow	Decreased P inputs to waterways, improved shelter for animals.	Benefit depends on severity of erosion and the number of years the trees take to grow. Assumes spacing of 10-15 m between trees.	Availability and purchase costs seedlings, planting costs, extended cost of weed and pest management	48,116
47. Sediment	Cropping	Contour cultivation	Cultivation along contours of cropping land with slopes greater than 3 degrees. Reduces the speed and eroding power of runoff water.	Mod	Low	Fast	Increased sustainability of cropping. Decreased P input (unquantified) to waterways.	Depends on infiltration capacity of soil and slope angle	Education	65
48. Sediment	Cropping	Cover crop	Green manure or cover crop after harvesting of main crop is ploughed into the soil stabilising bare soil from erosion and improves water penetration and drainage.	High	Low Cost from $80 ha^{-1}	Mod	Stabilises soil to prevent erosion. Some cover crops can be N-fixing.	Depends on soil structure and previous crop.	May not affect farm-scale losses depending on area.	64,84
49. Sediment	Cropping	Minimum tillage	Direct drilling of seed into stubble or pasture reduces the proportion of time that land is bare and erodible during the growing cycle.	Low	Low	Mod	-	Depends on the amount of time land is fallow	Management expertise. Could increase dissolved P losses.	117,118
50. Sediment	Cropping	Stubble mulching	Stubble is mulched and left on field. Partial ground cover protects soil from erosion.	Low	Mod	Mod	Decreased P input (unquantified) to waterways.	Depends on the amount of partial ground cover	Management expertise and risks for subsequent crops from disease and pest pressured.	(L. R. Basher, 2013)
51. Sediment	Cropping	Wheel track dyking  Taken from: http://www.hortnz.co.nz/assets/Uploads/Auckland-Waikato-ES-Control-Guidelines-1-1.pdf	Series of closely spaced indentations in wheel tracks created by tillage machinery. Slows surface runoff water down and settles suspended sediment.	Mod	Low Cost $35 ha^{-1}	Fast	Decreased P input (unquantified) to waterways.	Depends on the proportion of runoff coming from compacted soil	Specialist knowledge needed. Wheel tracks from spraying should not be ripped as loose soil may make spraying difficult.	65

Class / Targeted contaminant	Suitable for	Strategy	Description of function	Effectiveness (%)[1]	Relative cost ($/ha/yr)[2]	Response rate[3]	Co-benefits	Reasons for variability	Factors limiting uptake	Example references
52. Sediment	Cropping	Wheel track ripping  Taken from: http://www.hortnz.co.nz/assets/Uploads/Auckland-Waikato-ES-Control-Guidelines-1-1.pdf	Ripping of wheel tracks is suitable for use on cropping land after the use of heavy vehicles on cultivated soil. Ripping allows water to percolate into the soil rather than flow down the tracks.	Mod	Low	Fast	Decreased P input (unquantified) to waterways.	Depends on the proportion of runoff coming from compacted soil	Specialist knowledge needed.	65
53. Sediment	Cropping	Wind break crop	Tall crop in paddock providing shelter for neighbouring cultivated paddock from wind erosion.	Low	Mod	Mod	Decreased P input (unquantified) to waterways.	Depends on value of wind break crop	Management expertise	64
54. Sediment	Forestry	Hydro-seeding  Taken from: Forest Owners Association et al. [49]	A mixture of water, seed, fertilisers, organic binders and mulch is sprayed onto the surface to be vegetated. The binders and mulch improve the strike rate and reduce the opportunity for seed to be washed or blown away.	Low[4]	Mod[4]	Mod	None	Grass strike can be within 14 days but may take longer on steeper or low fertility slopes.	Cost and does not control slips.	49
55. Sediment	Forestry	Slash management  Taken from Te Uru Rākau – New Zealand Forest Service and Ministry for Primary Industries [119]	Slash can be used to reduce sediment loss from new construction, exposed soil generated during harvesting operations, and for post-harvest site rehabilitation.	Low[4]	Low[4]	Slow	None	Slash can dissipate the energy of headwater flow but should not be used to dissipate concentrated flow where it can be transported offsite.	Regional council requirements	49,119

[1] kg of nutrient or sediment or coliform forming units retained or immobilised and categorised into low, medium and high = 0-33, 34-66, and 67-100%, respectively.

[2] Cost breakdowns for each quarter were (low, medium, and high): N (<100, 101-366, and >366 $/ha/yr); P (<111, 112-476, and >476 $/ha/yr); Sediment (<81, 82-169, and >169 $/ha/yr); *E. coli* (<129, 130-192, and >193 $/ha/yr). Cost estimated in 2020.

[3] Fast, moderate, slow = within a season, within a year, and longer than a year, respectively.

[4] Effectiveness or cost is estimated owing to a lack of quantitative data.

Step 3. Plan and implement actions

Recommendations
3. Policy should assess the proportion of contaminant concentration, load or yield attributable to farm management, by catchment, and focus actions on this manageable portion. Natural losses can be estimated for some river or groundwater types. 4. Farm planning tools must address both farm management practices (and actions) and inherent biophysical risks, because contaminant loss results from their interaction. Addressing only one factor will not improve catchment water quality or provide useful data. 5. Farm plan maps must identify areas at most risk of contaminant loss (critical source areas) and capture actions implemented in these areas along with their implementation date. 6. Farm planning systems tools must quantify how actions reduce risk and contaminant loss, to enable comparisons across space and time. Quantification is important if the data will be used to fairly allocate responsibility across farms for improving catchment water quality. 7. Regular auditing must be required by farm plan policy, to confirm that actions have been implemented on-farm to a standard that will improve water quality, and to determine changes in risk and contaminant loss. Records of where and when an action is implemented are critical. 8. Farm planning tools must directly match Good Farming Practice actions to risk assessment for contaminant loss, to ensure these actions contribute to achieving catchment water quality targets. Actions additional to Good Farming Practice guidelines) may be required to meet catchment water quality targets.

A small fraction of contaminant concentrations and yields lost from agricultural land to water are generated from natural (non-human-influenced) processes like dissolution or erosion-prone geologies. The contribution of these processes has been quantified and termed – “reference conditions”. Concentrations under reference conditions are usually 5-20% but can be much greater in some catchments [120]. The difference between the concentration or yield under business as usual and a water quality target, established in Step 1, is the manageable portion (**Figure 3**). Plans to mitigate contaminant concentrations or yields only pertain to the manageable portion.

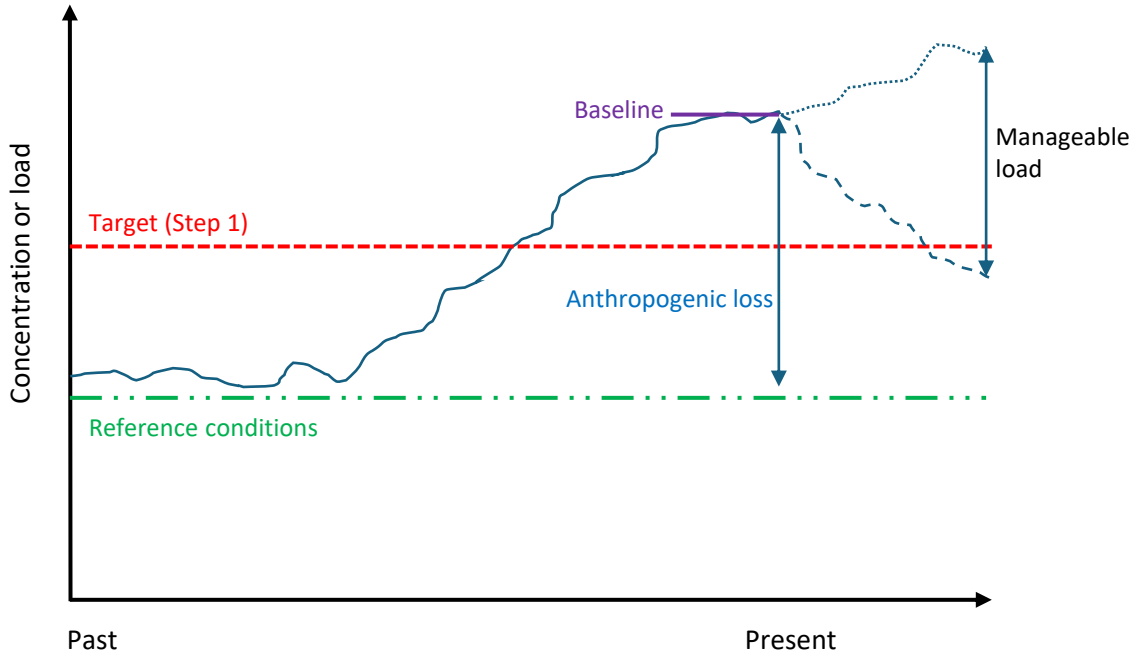


Figure 3. Schematic representation of the increase and contaminant concentration under business-as-usual operation (dotted line) relative to a target and reference conditions, and the point at which a decision is made to mitigate any increases in contaminant concentrations (dashed line) beyond a baseline by implementing actions on the manageable load.

Any plan to reduce losses should set the baseline of current losses and examine the expected effect of actions to decrease losses. Much like there is temporal variation in freshwater concentrations and yields, losses from farms are dependent on the interaction of management and climatic and biophysical conditions. However, unlike rivers or groundwaters which are strongly influenced by climatic and biophysical conditions [121], people can plan management to minimise losses, theoretically reducing variation. Nevertheless, management (or potentially land use) varies from year-to-year, and so too will losses. Hence, it is wise to set a baseline as the mean of multiple years' (e.g., five) management.

After setting a baseline and a target, a spatial farm planning system should be used to robustly quantify the risk of contaminant loss as helps the user decide what, where and when actions should be implemented (see *Planning and data capture at the farm scale*). Before embarking on a detailed analysis of the capacity of all farms in a catchment to reduce contaminant losses, work should gauge the likelihood of reducing losses through catchment averages. Using averages can mean missing losses from environmentally low and high-performing farms but is useful to determine if the manageable load is realistic or too ambitious (e.g., still above reference conditions). For example, work in 2020 [122-124] and then again in 2023/2024 for sheep and beef farms [125] used industry data and statistics to derive average farm types specific to locations around New Zealand. These farm types are considered appropriate to investigate the feasibility or effectiveness of implementing actions to mitigate the manageable load at a catchment scale. We undertook to update these farm types as part of a national-scale case study (page 76) with the most recent mitigation actions acceptable to the primary sector [126-128]. To ensure that the data for the effectiveness of actions were as realistic as possible we also adjusted the effectiveness by new data on the real implementation of actions in the dairy and sheep and beef sectors (no data were available for other sectors). We chose to model N

and P losses as these contaminants have an established method for assessing their risk of loss and mitigation [123].

Planning and data capture at the farm scale

Farm environment plans, including Freshwater Farm Plans, should robustly capture and display information about the risk of contaminant loss and help the user plan actions to decrease risk on the premise that there is a direct link between risk and contaminant loss. Currently, there are several farm environment plan systems offered by commercial companies (at a cost) and by Regional Authorities or industry groups for free. International best practice suggests that to quantify risk, any farm plan should capture the biophysical and management characteristics of the farm and map these to tell the farmer if there are area at high-risk of loss. These high-risk areas account for a disproportionate quantity of loss from a paddock, farm or catchment. Internationally, these high risk areas are called critical source areas [14]. As noted in *Factors to consider when choosing actions*, and in the Resource Management Act [129], targeting critical source areas with actions maximises the cost-effectiveness of actions [17]. If plans do not capture these characteristics, they may not provide a risk of contaminant loss that can be quantified or compared across farms, catchments, regions or to other plans. We have assessed a range of farm environment plans currently being considered for use as Freshwater Farm Plans. Regulation [3,129] dictates that the following elements must be included with a freshwater farm plan (**Table 3**) (with bolded text to group each requirement):

1. *Specified administrative and **map information***
2. *Relevant aspects (to the farm) of the **catchment context**, challenges and values (Catchment context) must be identified and considered in the risk assessment.*
3. *The farms **inherent risks** must be identified and considered in the risk assessment.*
4. *The risks from the **farming/growing** management practices must be identified and considered in relation to the catchment context and inherent risk in the **risk** assessment.*
5. ***Actions** must be put in place to **address** the **risks** identified through the risk assessment to the greatest extent possible while being fair and reasonable.*
6. *Actions must **comply with** any **other** requirements in **regulations** but may be more stringent.*
7. ***Actions** must be **categorised** into regulatory, catchment and other based actions, and be clear and **measurable** (clearly written and timebound).*

All but two of the farm environment plans reviewed did not meet all the elements required by regulation. The two exceptions were Ballance Agri-Nutrients: Ballance Farm Sustainability and the QCONZ Freshwater Farm Plan. Both still require improvement. For instance, the Balance Farm Sustainability plan needs to improve how it assesses risk in a catchment context, while the QCONZ Freshwater Farm Plan needs to improve its quantification of risk. The most common problem amongst the farm plan systems, was the poor quantification of risk (instead focusing on advocating for Good Management Practice). The problem is caused by a disconnect between an assessment of risk and freshwater contaminant concentrations or loads. This voids a crucial criterion, recognised by a recent report from the Parliamentary Commissioner for the Environment [130] that any model should

estimate the variable or interest (e.g., kg N ha^{-1}) directly and not via a proxy (risk). This problem is not fixed by moving from Good Management Practice [131] or Good Farming Practice [132] to a risk-based approach **unless** risk is linked to (i.e., can be translated into) a quantitative assessment of contaminant loss. We argue that to be cost-effective and quick, mitigation actions should be prioritised and placed in areas most at risk of contaminant loss via an objective tool/model. We support the risk-based approach, but only if work is undertaken to translate or align risk to contaminant losses so that reductions align to the, for example, percentage reduction required. Not doing so will lead to Freshwater Farm Plans that are not transferrable between locations or land uses. For instance, lowering risk based on farm actions without considering inherent risk (e.g., Tiaki) could lead to two farmers having the same risk score by adjusting actions to address risk despite farming a deep or shallow soil that is inherently low or high risk of leaching nitrate-N. Furthermore, if converted to pork production that spreads P-rich manures, an emphasis on N losses may be erroneous without catchment context if algal growth is limited by P. It is also worth noting that minimising the risk of loss may not result in good water quality if the level of reductions are too small.

Table 3. Summary of the fit of industry farm environment plans to the elements required in a Freshwater Farm Plan (heading relate to bold text in numbered bullet points above). Note that element 6 (compliance with other regulation) is excluded as this is often regionally dependent. Material for the summary was sourced from internal reviews by the Ministry of the Environment, except for QCONZ, which was released after internal review were complete, but judged by the report's authors.

Farm environment plan	1. Maps information	2. Catchment context	3. Inherent risks	4. Farming/growing risk	5. Actions address risk	7. Actions categorised (prioritised) and measurable	Main improvement required to meet regulation
Ballance Agri-Nutrients: Ballance Farm Sustainability [133]	Yes	Yes	Yes	Yes	Partial consideration	Partial consideration	Better documentation to outline how risk is quantified.
Beef and Lamb NZ: Environment Plans [127]	Yes	No	Partial consideration	Partial consideration	Yes	Yes	Needs to move beyond GMP to a quantitative risk-based approach
BioGro [134]	No	No	No	No	No	No	Improvement in all areas needed
Fonterra: Tiaki [135]	Yes	Yes	No	Partial consideration	Partial consideration	No	Needs to consider inherent risk and move beyond GMP to a quantitative risk-based approach
Foundation for Arable Research Farm Environment Plan template [136]	Yes	Yes	Partial consideration	Partial consideration	Partial consideration	No	Needs to move beyond GMP to a quantitative risk-based approach and prioritise and match actions to risk
NZ Farm Assurance Programme Plus [137]	Yes	No	Yes	Partial consideration	Partial consideration	Yes	Needs to move beyond GMP to a quantitative risk-based approach
NAGAP Environmental Management System Add-on [138]	Yes	Yes	Partial consideration	Partial consideration	Partial consideration	No	Needs to move beyond GMP to a quantitative risk-based approach and prioritise and match actions to risk
NZPork: Farm Environment Plan [139]	Yes	No	Yes	Yes	Partial consideration	Yes	Needs to move beyond GMP to a quantitative risk-based approach
QCONZ: Freshwater Farm Plan [140]	Yes	Yes	Yes	Yes	Yes	Partial consideration	Needs to quantify improvements from actions
Ravensdown: Farm Environment Plan [141]	Yes	No	Partial consideration	Yes	Partial consideration	No	Needs to move from GMP to a quantitative risk-based approach and prioritise and match actions to risk

Two examples of objective tools to assess risk as part of a farm plan are MitAgator [142] from Ballance Agri-Nutrients which manages N, P, SS and *E. coli* and the Risk Index Tool [143] from the Ministry for the Environment, which currently only manages N loss. Example maps for each are shown below (**Figure 4**). Both have been calibrated and tested against observations of contaminant loss from experimental trials, farmlet or small catchment studies, and have estimates of uncertainty and sensitivity to inputs and variables controlling risk of loss. The underpinning algorithms for both tools are open source. The exception is MitAgator's risk map for N which relies on elements of Overseer, meaning that it is only suitable for estimating N losses from intensive agriculture on flat land [144]. Both tools provide an estimate of the cost-effectiveness of actions to mitigate risk. MitAgator applies these using an optimisation process allowing the user to input either a desired percentage reduction to achieve or investment available to spend on actions to reduce losses and output a list of suitable actions ordered by cost-effectiveness.

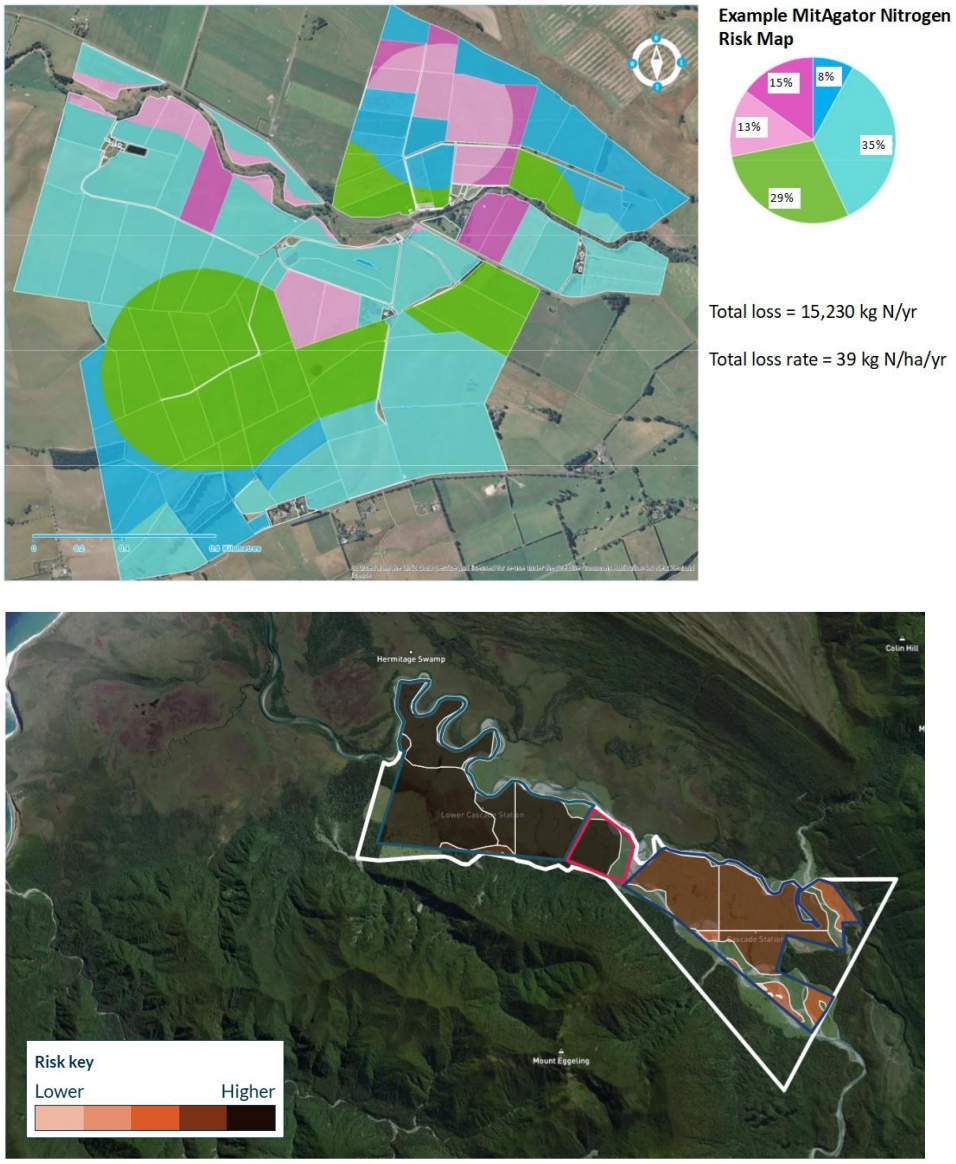


Figure 4. Example maps of the risk of nitrogen loss from farm using MitAgator (top) and the Risk Index (bottom). Note that risk in the index is given by colour-coded class but is also given to the user as a numerical risk by polygons of common slope, soil type and climate by management interactions.

The Freshwater Farm Plan system also requires an initial certification that actions are fit for purpose, followed by on-going audits of their implementation, and periodic re-certification to gauge if actions are still fit for purpose. Auditing can be problematic for attributing actions as the cause of water quality improvements (**Step 5**). For example, good information is obtained from an audit that assesses if all actions have been implemented to 'standard' [1] if accompanied by a quantitative assessment of risk to indicate, for example, a percentage reduction in contaminant loss. This reduction, and iterations thereof, can be used over time to attribute actions to water quality improvements. Under many audit schemes this receives an A grade [145] and are not required to be audited again for three-four years. However, audits receiving B, C or D grades indicate lower levels of implementation and non-compliance with the plan. These are audited more frequently, but do not inform us about attribution unless accompanied by a quantitative reassessment of the magnitude of risk reduction. It is also possible that audits do not occur at a frequency compatible with attribution, where for example, actions that cause change within a year are only picked up every four years, and that non-compliance elsewhere dilutes the catchment water quality improvement. To improve catchment water quality, we must know who is reducing their losses, where and by when. The Freshwater Farm Plan system is an appropriate way of aggregating and accounting for those changes.

Step 4. Maximise likelihood of detecting change

Recommendations
9. Regional councils or catchment groups should assess whether current water sampling frequency in a catchment is sufficient to reliably detect achievement of a water quality target, accounting for natural variability. 10. If the monitoring frequency results in too few samples to determine if detection is likely, councils or catchment groups should consider taking more samples, especially of the top 10% of flows (see monitoringfreshwater.co.nz). 11. Councils should maintain some long-term (20+ years) water monitoring sites to statistically control for the influence of a changing climate, lag times, and increasing production. 12. Public expectations about improvement in water quality should be managed with education about time lags. Until more data are available, expectations can be guided by statistical analysis of existing monitoring data, for catchments that have been monitored for long enough.

The purpose of monitoring freshwater quality is to establish the current state and any trend in observations to determine if concentrations (or other units of measurement) achieve targets or are

[1] Note that we assume all auditing systems apply the principles in Step 2 and that there is no variation in audit assessments.

improving or degrading. Clearly, not only is this information used to set targets and the need for farm actions to mitigate contaminant losses, but it is also central to showing (with a degree of confidence) that actions are reducing/changing contaminant concentrations in an acceptable period.

There are three factors that influence our ability to detect changes over time. The first is that we know that runoff and drainage from a farm is routed to the point of detection. Processing like denitrification may occur during routing. A commentary on the tools (viz. models) available to route and process contaminants is outside the scope of this report. It is an active area of research indicating that the level of confidence in these tools is not optimal (at least from a science-perspective) [146,147].

The second and third factors relate to the number and frequency of sampling to capture (and discount) variation, for example, from the changing influence of actions, increases in losses via production or climate, and the lag time between actions being implemented and the runoff or drainage reaching the point of detection.

Robust monitoring regimes rely on capturing enough samples, over a long-enough period, to increase the likelihood (power) of detecting that a change is real [148]. Statistical power is the probability that the expected result (e.g., a change in contaminant concentration) is real. It is common to use a binary threshold for statistical power of ≥0.80 as the level at which the likelihood of an effect (one-sided test) is real [149], but this is influenced by the critical level (α), sample size, the desired level of effect (e.g., percentage reduction in concentration) and variability in the data. If the critical level is set at $P<0.05$, the standard deviation of the contaminant concentrations over a period can be used to calculate either the level of reduction detectable with a specified number of samples or the number of samples required to meet a specified level of reduction. This information helps regulators and farmers avoid missing the opportunity to detect change and mistakenly concluding that change with a low level of confidence equates to no change. The data are also useful to forecast the time it may take to detect the change and the cost of detection since our main method of increasing power is to increase the number of samples taken [150].

Internationally, and in New Zealand, streams and rivers are commonly sampled monthly [151]. A power analysis of monthly sampling of 13 catchments in Canada estimated that detecting a 40% reduction in total phosphorus [152] would take 8-50 years of data [148]. In New Zealand, an analysis was conducted of the power to detect the change from median values at 2022 to national bottom lines set in the National Policy Statement for Freshwater Management [153] for clarity, toxicity for NH_4-N and NO_3-N, and primary contact in bathing areas for *E. coli*, and a threshold between C and D attribute states that relates to an enhanced impact on water quality through periphyton growth [154] for dissolved reactive P. The results for 753 stream and river sites sampled from 2004 to 2019 indicated that there was a low level of confidence in reductions to bottom lines for an average of 21% of site by contaminant combinations in 5 years, reducing to 12% over 20 years because more samples were taken (**Table 4**) [155]. Interestingly, using the 30% mean reduction calculated as achievable after the implementation of actions to the farm types by 2035 [124], a power analysis indicated that 39% of sites would not detect a 30% reduction after 20 years and 94% of site by contaminant combinations would not detect this reduction (with confidence) after 5 years (**Table 4**) [155]. This is because 30% tends to be a smaller, and hence less detectable change than that required to meet national bottom lines [156].

Table 4. Mean percentage of sites (n = 753) not able to confidently detect changes given a monthly sampling monitoring regime to meet bottom lines or a 30% reduction by 2035 for freshwater quality by contaminant in 5 and 20 years.

Contaminant	5 years	20 years
National bottom lines		
Clarity	17	13
Dissolved reactive P	14	9
E. coli (median)	40	22
E. coli (95th percentile)	49	24
Ammoniacal N	1	1
Nitrate N	2	1
30% by 2035		
Clarity	100	62
Dissolved reactive P	91	0
E. coli (median)	100	100
Ammoniacal N	82	17
Nitrate N	97	56
Total N	85	1
Total P	100	37

Similar analyses of statistical power were conducted from groundwater (n = 948) and lake (n = 148) sites across New Zealand, except here the sampling frequency was commonly quarterly for groundwater and bimonthly for lakes. For groundwater, the percentage of sites able to detect a change from current concentrations to a national bottom line of 2.4 mg nitrate-N L^{-1} in 5 and 30 years was 5 and 85%, respectively [157]. For lakes, 15, 3 and 4% could not detect change to a national bottom line for chlorophyl-a, total N and total P, respectively within 5 years, reducing to 8, 1, and 1%, respectively over 20 years [158].

The policy implications of these findings for monitored streams and rivers and groundwaters are that:

1. More samples would need to be taken at existing sites to detect changes, but this was estimated to increase cost by 4-5 times (2024 costs) for streams and rivers and 2-3 times for groundwaters (excluding the effect of lags which make 40% of sites unresponsive after 30 years – see below).
2. Additional work could look to account for the influence of climate, thereby reducing variation and the number of samples required to detect change. However, this may not be possible and still require more data over a long period (more on this is covered in Step 5).
3. Sites could be changed to areas that are more likely to exhibit change because they may have been moved closer to where change is occurring.
4. The percentage change required as a threshold policy target is changed.

The third factor influencing our ability to detect change is lag time. We define lag time as the time taken for changes in contaminant loss from the topsoil to be reflected in the contaminant load delivered to a waterbody. This is consistent with other definitions of a biophysical lag time [159], and excludes any lag caused by policy and implementation. Despite action to mitigate contaminant losses from agricultural land, long lag times for past contaminant inputs can sustain or even increase future

contaminant loads in groundwaters, rivers and lakes [160]. Clearly, understanding the magnitude of lag times within and between catchments is essential to knowing how much, where and when actions to mitigate contaminant loss from land will be best placed and take effect [161]. Remaining ignorant of these effects can raise expectations that water quality will improve within unrealistic time frames, which compromises the confidence of landowners to invest in mitigation actions [162].

Most studies of lag times have examined nitrate-N. Internationally, lag times for nitrate-N have been calculated at 12-34 years in agricultural catchments of the Great Lakes and 7-22 years in mixed forest/agricultural catchments in Germany. Lag times vary depending on stream location and season as either shallow, shorter lag times (e.g., through riparian soils and tile drains) or deeper, longer lag times (e.g., aquifer-driven) flow-paths contribute the nitrate load. A study in New Zealand assessed the difference between likely nitrate-N lost calculated from 1990-2017 (modelled via Overseer and agricultural census data for dairy and sheep and beef land) and the load calculated in 77 sites from the National the National River Water Quality Network (NRWQN) run by the National Institute of Water and Atmospheric Research and Regional Councils [163]. Lag times between these nitrate-N losses were determined for 37 catchments but could not be determined in other catchments because they exhibited little change in nitrate-N losses or loads. Lag times in nitrate-N loads varied from 1.5 to 11.5 years according to catchment size, slope and agricultural land use. For nine catchments where isotope and modelling data were available, the mean transit time for surface water at baseflow (excluding spring-fed streams) was on average 3.3 years younger, but never older, than the mean lag time.

More recent data has quantified the age of water across the NRWQN isolating the mean young water fraction (proportion of annual water <2.3 ± 0.8 months) as about 25% [164] - implying that even in these large catchments (mean area 2,640 km^2), most water is under a year old (**Table 5**). The implication of these data is that with a dominance of younger water and a median lag time across the 37 catchments for nitrate-N of 5.4 years, lag times for nitrate-N are unlikely to influence the detection of change in most catchments <2,640 km^2 because data are available for much longer periods (e.g., at least 15 years). However, the same implication cannot be extended to other contaminants like sediment or sediment-bound N and P which following erosion events can settle out in stream channels often taking many decades to pass through river systems [165,166].

Table 5. Biophysical characteristics and co-ordinates of sites within the National River Water Quality Network with the fraction of young water (F_{yw}; and confidence intervals determined by machine learning) from Dudley et al. [164] and mean lag time for nitrate-N from McDowell et al. [163].

Site	Site name	Easting	Northing	Catchment area (km^2)	F_{yw}	F_{yw} low	F_{yw} high	Mean lag time (yrs)[a]
AK1	Hoteo @ Gubbs	2646600	6534000	270	0.22	0.13	0.34	Impacted
AK2	Rangitopuni @ Walkers	2654900	6494000	82	0.28	0.14	0.51	2.3
AX1	Clutha @ Luggate Br.	2215500	5601900	4453	0.08	0.04	0.12	MDC
AX2	Kawarau @ Chards	2184400	5569800	4302	0.16	0.11	0.21	MDC
AX3	Shotover @ Bowens Peak	2172200	5571000	1079	0.26	0.01	0.54	MDC
AX4	Clutha @ Millers Flat	2230200	5498700	16548	0.16	0.11	0.22	-
CH1	Hurunui @ Mandamus	2472500	5824000	1060	0.22	0.12	0.33	MDC
CH2	Hurunui @ SH1 Br.	2517900	5812100	2525	0.21	0.14	0.29	4.3
CH3	Waimakariri @ Gorge	2433100	5760500	2387	0.24	0.14	0.36	MDC

CH4	Waimakariri above old HW Br.	2481000	5754400	3076	0.28	0.15	0.41	4.0
DN1	Taieri @ Tiroiti	2295900	5546600	2423	na[b]	na	na	3.3
DN2	Monowai below Gates	2085300	5475100	258	0.53	0.32	0.80	6.3
DN3	Sutton @ SH87	2283200	5508400	151	na	na	na	-
DN4	Taieri @ Outram	2295800	5481000	4716	0.19	0.14	0.24	Impacted
DN5	Clutha @ Balclutha	2259000	5436200	20582	0.11	0.01	0.21	2.0
DN6	Mataura @ Seaward Down	2186600	5416000	5139	na	na	na	-
DN7	Mataura @ Parawa	2163500	5507300	787	na	na	na	5.5
DN8	Oreti @ Lumsden	2154100	5489200	1139	na	na	na	4.0
DN9	Oreti @ Riverton HW Br.	2145400	5420800	2150	0.19	0.07	0.31	3.0
DN10	Waiau @ Tuatapere	2099400	5439800	8098	na	na	na	MDC
GS1	Waipoa @ Kanakanaia	2935900	6292300	1571	0.20	0.13	0.28	6.0
GS2	Waikohu @ No. 1 Br.	2908300	6299700	30.5	0.09	0.03	0.20	Impacted
GS3	Motu @ Waitangirua	2914700	6323300	293	0.15	0.08	0.23	-
GS4	Motu @ Houpoto	2917600	6360500	1376	0.12	0.05	0.18	-
GY1	Buller @ Te Kuha	2402000	5929400	6309	0.26	0.14	0.38	Impacted
GY2	Grey @ Dobson	2370000	5860100	3827	0.33	0.22	0.46	4.5
GY3	Grey @ Waipuna	2410000	5871800	642	0.29	0.18	0.41	MDC
GY4	Haast @ Roaring Billy	2212900	5689500	1027	0.20	0.05	0.37	MDC
HM1	Waipa @ Otewa	2715600	6323400	304	0.10	0.06	0.15	MDC
HM2	Waipa @ Whatawhata	2699700	6376000	2822	0.09	0.05	0.14	4.5
HM3	Waikato @ Hamilton Traffic Br.	2711800	6376400	8431	na	na	na	7.0
HM4	Waikato @ Rangiriri	2698900	6416700	12429	na	na	na	7.5
HM5	Waihou @ Te Aroha	2749400	6402600	1127	na	na	na	-
HM6	Ohinemuri @ Karangahake	2750600	6417200	305	0.10	0.04	0.17	-
HV1	Makaroro @ Burnt Br.	2792800	6148900	121	na	na	na	9.4
HV2	Tukituki @ Red Br.	2846600	6158100	2438	0.15	0.09	0.21	-
HV3	Ngaruroro @ Chesterhope	2842500	6171500	2001	0.18	0.11	0.26	12.0
HV4	Ngaruroro @ Kuripapango	2796900	6197400	384	0.15	0.06	0.25	MDC
HV5	Mohaka @ Raupunga	2867200	6228500	2371	na	na	na	-
HV6	Mohaka @ Glenfalls	2824000	6218800	1040	0.12	0.04	0.20	-
NN1	Motueka @ Woodstock	2495300	5995200	1750	0.21	0.13	0.30	-
NN2	Motueka @ Gorge	2502800	5952600	166	0.21	0.12	0.33	MDC
NN3	Wairau @ Dip Flat	2503600	5923400	521	0.15	0.06	0.26	MDC
NN4	Wairau @ Tuamarina	2589600	5973800	3494	na	na	na	2.8
NN5	Buller @ Longford	2459000	5937800	1404	0.15	0.09	0.22	MDC
RO1	Tarawera @ Lake outlet	2816900	6329700	414	0.10	0.06	0.16	3.0
RO2	Tarawera @ Awakaponga	2841200	6355700	914	0.05	0.02	0.09	1.0
RO3	Rangitaiki @ Murapara	2832700	6298400	1144	0.05	0.02	0.09	-
RO4	Whirinaki @ Galatea	2836800	6295300	509	0.06	0.01	0.10	MDC
RO5	Rangitaiki @ Te Teko	2843600	6344400	2818	0.05	0.02	0.09	-
RO6	Waikato @ Reids Farm	2777800	6276300	3305	0.07	0.04	0.10	5.0
TK1	Opihi @ Waipopo	2376200	5658600	2373	0.06	0.00	0.21	7.0
TK2	Opihi @ Rockwood	2345400	5669000	406	0.14	0.01	0.31	1.8
TK3	Opuha @ Skipton Br.	2348200	5679000	458	0.24	0.02	0.50	4.0
TK4	Waitaki @ Kurow	2308000	5608800	9741	0.18	0.09	0.27	MDC
TK5	Hakatakamea above MH Br.	2311200	5606200	896	0.22	0.12	0.34	-
TK6	Waitaki @ SH1 Br.	2360000	5585000	11909	0.15	0.07	0.23	5.6

TU1	Whanganui @ Te Maire	2699800	6249000	2139	0.12	0.03	0.21	Impacted
TU2	Tongariro @ Turangi	2753700	6241700	786	0.07	0.00	0.14	MDC
WA1	Waitara @ Bertrand Rd	2618700	6238900	1114	0.20	0.08	0.33	-
WA2	Manganui @ SH3	2618900	6213000	19	0.09	0.00	0.32	7.5
WA3	Waingongoro @ SH45	2614000	6180300	216	na	na	na	-
WA4	Whanganui @ Paetawa	2693500	6155400	6567	0.14	0.04	0.23	6.0
WA5	Rangitikei @ Mangaweka	2750400	6151200	2689	0.17	0.10	0.25	8.0
WA6	Rangitikei @ Kakariki	2718300	6117200	3451	na	na	na	-
WA7	Manawatu @ Weber Rd	2774700	6102600	716	0.17	0.11	0.24	9.5
WA8	Manawatu @ Teachers Coll.	2732800	6089300	3916	na	na	na	-
WA9	Manawatu @ Opiki Br.	2719500	6082700	4846	na	na	na	-
WH1	Waipapa @ Forest Ranger	2573400	6658400	122	0.12	0.03	0.26	MDC
WH2	Waitangi @ Wakelins	2606100	6657700	307	0.31	0.14	0.53	-
WH3	Mangakahia @ Titoki Br.	2605900	6607100	809	0.23	0.14	0.34	2.5
WH4	Wairu @ Purua	2614900	6615900	546	0.31	0.20	0.44	3.5
WN1	Hutt @ Boulcott	2671200	5999200	605	na	na	na	-
WN2	Hutt @ Kaitoke	2694200	6015000	87	0.17	0.09	0.29	MDC
WN3	Ruamahanga @ Waihenga	2714600	5998400	2368	na	na	na	-
WN4	Ruamahanga @ Wardells	2734700	6019200	640	na	na	na	5.5
WN5	Ruramahanga @ SH2	2729900	6046100	78	0.28	0.14	0.48	MDC

[a] Impacted, MDC and – refer to sites that where lag times could not be determined owing to being impacted by hydropower schemes, and under minimally disturbed conditions or agricultural conditions that exhibited no change in nitrate-N loss or load from 1990-2017.

[b] no data were available.

Like rivers and streams, shallow groundwater is often young, and in freely draining, light soils, filtration or sorption process can be ineffective at removing P, *E. coli* and nitrate-N meaning that responses to land use can be detected within a few months or years. For instance, Pang et al. [167] measured water velocities and the movement of *Bacillus subtilis* at 30 to 85 m day^{-1}. The saturation and compromise of P sorption sites in pore walls by dairy shed effluent was also noted in a freely-draining Canterbury soil resulting in an 8% year-on-year increase in dissolved P concentrations in leachate [168]. The potential enrichment of topsoil and leaching of P via uniform rate irrigation has been attributed to an increase in groundwater P in a few sites [169]. However, it has also been noted that when irrigation management changed to variable rate, leaching of P to depth was mitigated by 70% within a year [80].

Contamination of deeper groundwater is almost exclusively by mobile nitrate-N as other contaminants tend to be filtered out or sorbed on the way. Lag times to deep (e.g., > 100 m deep) groundwater can be decades or hundreds of years. This may be of concern if the aquifer is used for potable drinking water and nitrate-N concentrations are increasing or during period of prolonged low flow where older groundwater makes up a greater proportion of flow and/or spring-fed streams. The majority of age tracing data, at least in Canterbury, is biased towards deeper groundwater and resulted in mean transit times that are >50 years for half of the 600 samples taken [170]. However, Scott et al. [170] also note many cases where transit times and nitrate-N lag times are only a few years (**Table 6**). Lag times in shallow aquifers in Southland are also commonly <5 years [171]. Indeed, an analysis of 288 samples across New Zealand found mean lag time for nitrate-N of 30 years with 117 of those samples <10 years (**Figure 5**) [157].

Table 6. Case studies of nitrate-N lag times between shallow groundwater. The lag time is listed as the time in months for an increase in rainfall or grazing / effluent application to cause a steady state response. Data from Scott et al. [170].

Management / location	Soil drainage characteristics	Well screen depth (m)	Nitrate-N response time (months)
Response to rainfall			
Balmoral, Hurunui	Well drained shallow	12-22	0.1
Pleasant point, Timaru	Moderately-well drained	4.75[a]	0.5
Rolleston, Selwyn	Well drained	33-34	2
Yaldhurst, Christchurch	Well drained	27-33	1-2
West Melton, Selwyn	Well drained	40	4
Dorie, Ashburton	Moderately-well drained	30	1-2
Swannanoa;	Moderately-well drained	16-20	3
Ikawai, Timaru	Imperfectly drained	4.6	1
Pareora, Timaru	Imperfectly drained	5-8	1
Seadown, Timaru	Imperfectly drained	10	2
Seadown, Timaru	Poorly drained	6-11	1.2
Grazing intensification / effluent applications			
Balmoral, Hurunui	Well drained shallow	12-22	36
Hekaeo-Hinds, Ashburton	Well drained shallow	2-15	60
South Canterbury	Well drained	6-17	36

[a] Single value equals well depth as no screen depth given.

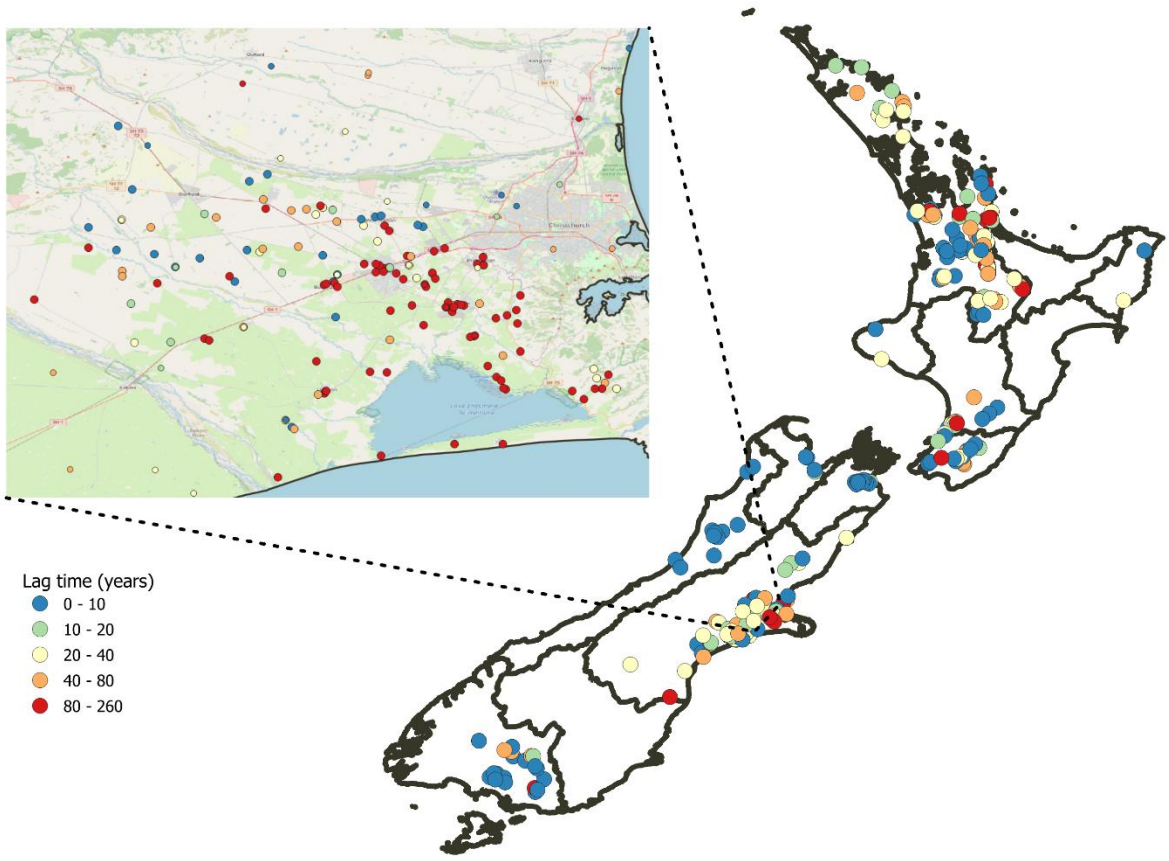


Figure 5. Estimated lag times (years) for nitrate-N at 288 groundwater sites across New Zealand. The expansion shows lag times for the lower Selwyn District (relevant to the Case Study).

When combined with the power analysis, lag times had little effect on detecting change in shallow wells, but owing to a dominance of deeper wells in the national network, the ability to detect change to the national bottom line decreased by half, from 85 to 41% with 30 years of bimonthly sampling [157]. This implies that at either the current frequency of sampling is too low to detect change or, more likely, that the groundwater is too old to detect recent nitrate-N losses.

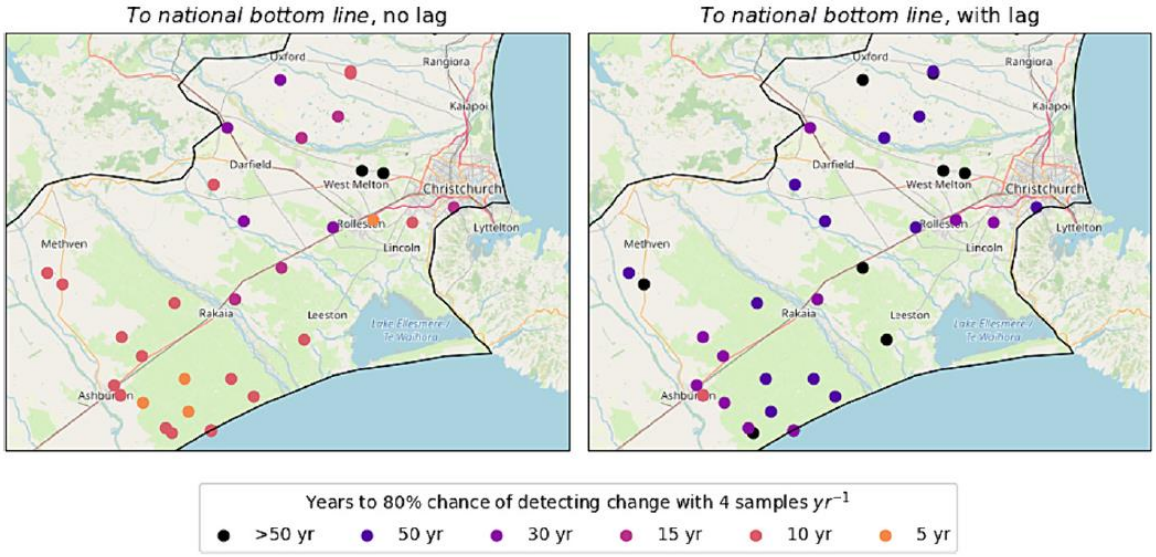


Figure 6. Map showing the effect of incorporating lag time (years) on the likelihood of detecting a change to 2.4 mg nitrate-N L^{-1} ('national bottom line'). Note the stark contrast in detection times for wells with deeper (older) groundwater east of Ashburton.

There is considerable scope to change current monitoring systems to better detect changes in water quality. Decisions about monitoring sites and frequency of sampling are based on many factors such as cost, the need to detect short-term responses to management and mitigation actions and to have long-term sites that may detect slow change. The desire to have feedback about the effects of mitigation actions is evident from the abundance of catchment groups (n > 200) established around New Zealand that are also undertaking monitoring. Sampling locations for most catchment groups have been informed by an array of expert opinion. However, a tool is now available to help inform where to best sample to detect change. The tool (www.monitoringfreshwater.co.nz) allows a user to select multiple use cases such as:

1. Choose a catchment and see the effect of different monitoring frequencies over time on the likelihood of detecting a user-derived level of change in contaminant levels at existing monitoring sites.
2. Choose a catchment and set a desired detectable magnitude of change for a target contaminant and see the likelihood of detecting change at different parts of the stream network.

3. Choose a catchment and see the proportion of catchment loads (at monitored sites) attributable to high flows (top 10% of flows)

The third scenario recognises that high flows account for most catchment contaminant loads. If a target has been set as a load or yield and allocated to areas within the catchment (e.g., a 30% reduction in load for all farms), information about the proportion of the load coming from high flows informs the need for more sampling at high flows to reduce uncertainty. The frequency of sampling is a key factor that determines the accuracy of contaminant yield estimates. An 'accurate' yield is one that is as close as possible to the 'true' yield. Although continuous records of contaminant concentrations and hence the true yield do not exist, high-frequency sampling by contaminant sensors, often only a few minutes apart, is seen as the closest proxy for continuous measurement and therefore can be combined with high-frequency river flow measurements to give accurate estimates of the 'true' yield. High-frequency contaminant monitoring is becoming more common especially for nitrate-N and turbidity (with total P and *E. coli* becoming cheaper), with the data used to inform our understanding of the interactions between catchment and stream processes such as nutrient cycling, stream metabolism and the effects of different lithologies, soil types, and flow paths on contaminant transfers [172]. Bieroza et al. [172] also point out that high-frequency sampling can be used to detect changes in load that infrequent sampling may miss. For example, Shore et al. [173] showed that the underprediction of nutrient loads was exacerbated if high flows coincided with, or were soon preceded by, an application of dairy slurry containing N and P. In another study, Wilkins et al. [174] examined the nitrate-N response to forestry harvesting using a high frequency UV sampler and found that 80% of the load occurred during a quarter of the year that discrete sampling couldn't accurately characterise.

A recent study [175] assessed the contribution of high-flows (top 10 percent) to total yields of nitrate-N, total N, total P, and *E. coli* calculated from monthly data over 15 years (310-325 sites) in New Zealand (**Figure 7**), and at 24 sites (**Table 7**) with high-frequency (30-min) nitrate-N and total P (via regression to turbidity). The top 10% of flows contributed 51-74% of annual contaminant yields at long-term sites and 48% of nitrate-N and 63% of total P in the high-frequency sites. Mean uncertainties in annual yields were also estimated from different frequencies of monitoring data (2, 6, 12, and 24 hourly, weekly and monthly) compared to the true yield, calculated from high-frequency records for all flows and just for high flows. Uncertainties from monthly data for all flows were 29% for nitrate-N and 52% for total P (**Table 8**). The corresponding uncertainties for high flows were 54 and 58%. These magnitudes of uncertainty are similar to or greater than many load reduction targets. Based on the data from these 24 sites, there is a need to sample more frequently, especially at high flows if load estimates, and progress towards targets, are to be determined accurately. Daily sampling will be needed to reduce uncertainty to <10%. Additional modelling suggests that this especially relevant for catchments with a high proportion of agricultural land use where there is large variation in, or actions in place to reduce, contaminant losses.

Table 7. High-frequency sites name, length of data record, total yield, high flow yield and the percentage of total yield associated with high flows (high flow / total yield) of nitrate-N and total P for each high-frequency data site.

Contaminant / Site name (size, km^2)	Years of record	Total yield ($kg\ ha^{-1}\ yr^{-1}$)	High flow yield ($kg\ ha^{-1}\ yr^{-1}$)	High flow / total yield (%)

Nitrate-N				
1 Kakanui River at McCones (676)	6.8	1.1	0.7	61
2 Kakanui River at Gemmels (294)	1.1	1.1	0.5	46
3 Mill Creek (41)	3.0	7.8	5.2	66
4 Shag River at Craig Road (430)	5.9	0.9	0.7	70
5 Shag River at Shakey Bridge (505)	2.9	1.0	0.7	73
6 Kaiapoi River (6)	3.7	11.2	1.8	16
7 Hurunui River (2513)	4.2	2.7	0.6	24
8 Windermere drain (52)	4.2	6.0	1.8	31
9 Aparima River (1259)	7.3	10.0	4.5	45
Total P				
10 Mangaeho River (415)	1.2	0.8	0.6	68
11 Mangati Stream (2)	4.1	1.1	0.5	47
12 Tangahoe River (280)	0.4	0.9	0.5	56
13 Waingongoro River (226)	1.5	0.6	0.3	48
14 Wakapuaka River (43)	4.9	0.6	0.2	39
15 Horokiri Stream (29)	7.5	0.8	0.6	75
16 Pauatahanui Stream (39)	7.5	2.3	1.8	81
17 Porirua Stream (33)	7.1	2.6	2.1	81
18 Manawatu River (714)	7.9	1.2	1.0	79
19 Mangahao River (280)	3.1	0.5	0.3	66
20 Mangatainoka River (403)	8.4	0.7	0.5	70
21 Ohura River (670)	11.4	0.9	0.5	58
22 Pohangina River (487)	8.7	1.1	0.7	66
23 Rangitikei River (2685)	8.5	0.4	0.3	62
24 Tiraumea River (761)	7.9	5.4	2.4	45

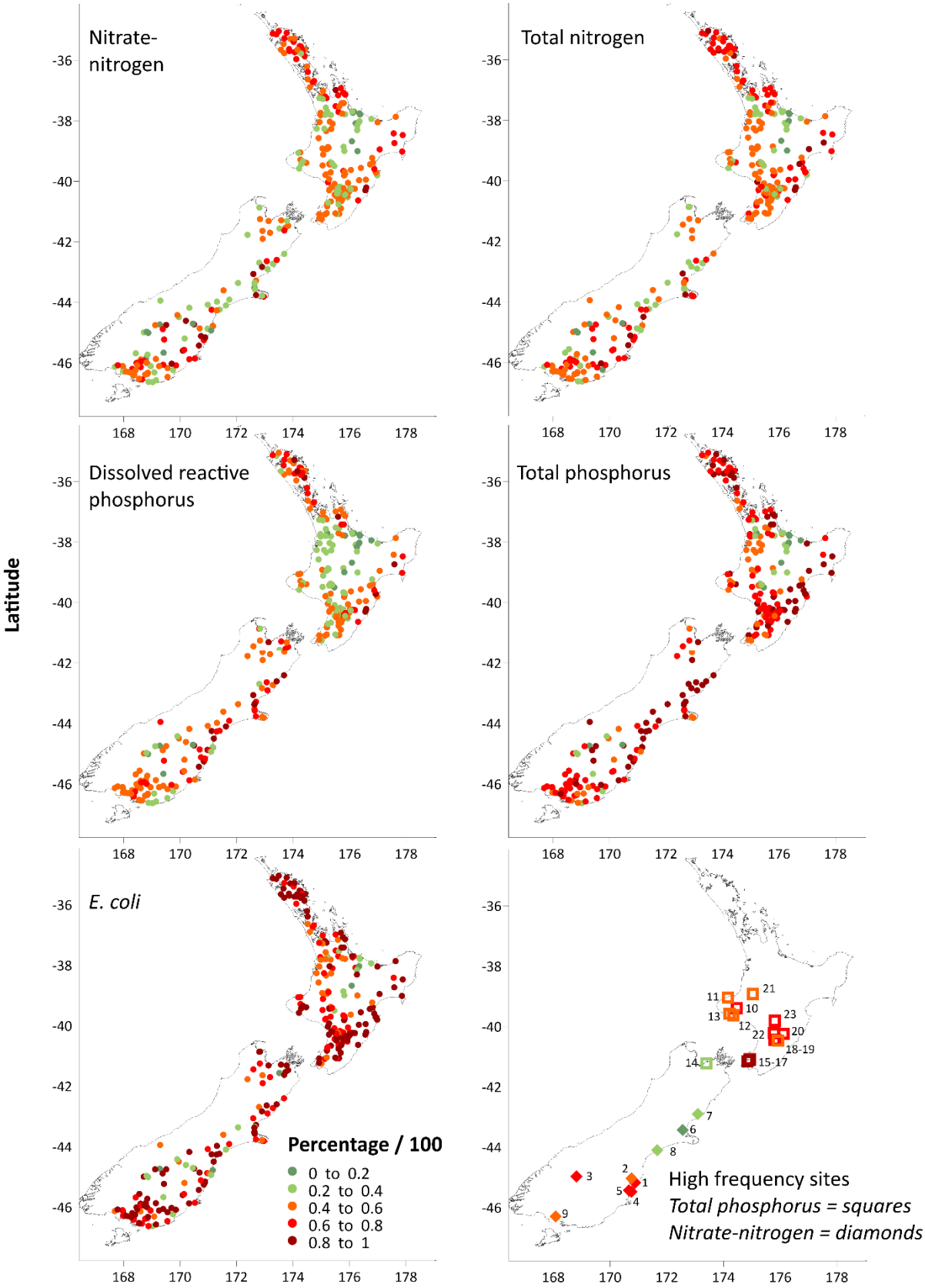


Figure 7. The influence of high flows on nitrate-nitrogen and total phosphorus yields. Total contaminant yields associated with high flows, at monthly sampled and high-frequency (bottom right map) sites. Numbers in the bottom right map refer to high-frequency sites listed in Table 7.

Table 8. Uncertainty caused by sampling frequency. Mean number of samples and the absolute mean percentage difference (± 95% confidence interval) in yield (estimated yield based on lower sampling frequency vs true yield calculated from high-frequency data) for high flows and total (all flows) for nitrate-N and total P.

Contaminant yield and sampling frequency	Mean number of samples	Absolute mean (%) difference (± 95% CI) for high flow yields	Absolute mean (%) difference (± 95% CI) for all flow yields
Nitrate-N (mean true yield = 4.6 kg ha^{-1}; mean yield for high flows = 1.8 kg ha^{-1})			
2-hourly	4179	3.7 (0.9, 6.5)	2.1 (0.3, 3.9)
6-hourly	1394	2.0 (0.7, 3.2)	0.8 (0.2, 1.4)
12-hourly	697	5.1 (1.4, 8.7)	2.3 (0.5, 4.2)
Daily	348	9.8 (3.0, 16.6)	5.7 (1.8, 6.9)
Weekly	50	17.2 (8.7, 25.7)	11.7 (4.6, 18.8)
Monthly	11	53.7 (23.0, 84.4)	28.7 (14.3, 43.1)
Total P (mean true yield = 1.6 kg ha^{-1}; mean yield for high flows = 1.0 kg ha^{-1})			
2-hourly	3481	0.7 (0.3, 1.5)	1.2 (0.7, 1.7)
6-hourly	1195	1.8 (1.0, 2.5)	3.8 (1.9, 5.7)
12-hourly	598	4.3 (1.7, 7.0)	4.8 (3.2, 6.3)
Daily	299	8.3 (3.0, 13.5)	7.3 (3.8, 10.9)
Weekly	43	28.4 (19.7, 37.1)	18.8 (14.9, 22.8)
Monthly	10	57.7 (43.2, 72.2)	52.4 (36.3, 68.7)

Step 5. Attribution of improvements to actions

Recommendations

13. Regional councils should continue to assess the effectiveness of actions using targeted catchment-scale studies in different catchments and under different conditions.
14. Before investing in catchment-scale studies, ensure information in the previous steps has been robustly determined. Studies can only detect the effect of actions where there are data on what actions were implemented, where, when, and their farm-scale effect.
15. Demonstrating farm plan effectiveness across multiple catchments over time requires comprehensive water quality monitoring that collects sufficient data to statistically control for confounding influences such as climate, lag times, and increased production.
16. Proving farm plan effectiveness requires consistent data collection and retention over many years (much like water quality data) to enable the attribution of water quality improvements to specific farm actions. This will require a long-term commitment by government, industry, and farmers to succeed.

International evidence shows that monitoring and recording each step is crucial to determining why water quality may not be changing as desired. A lack of a standardised approach is often cited in regional and global assessments as a key factor in failing to attribute water quality improvements to

actions [4]. This can be for several reasons, like using the wrong actions. For instance, in the European Union, payment is commonly made for 'desirable' agricultural practices, but water quality improvements have been to slow to negligible because those payments may not be optimally aligned to reducing contaminant losses [176]. Poor training of personnel has been cited as a cause of a failure of mandatory and voluntary schemes to implement the transport of manure (and nutrients) from areas of excess to areas of deficit in the United States [177]. Even where good knowledge and training enables actions to be in what is theoretically the right place, large initiatives such as the United States Department of Agriculture's Conservation Effects Assessment Project have estimated reductions in contaminant loss, but lacked evidence to show that conservation practices improve water quality at a catchment scale [178]; a problem that is now being addressed with the establishment of 23 catchment-scale studies [179].

The attribution of water quality changes (ideally improvements) requires all steps in the process (**Figure 1**) to be done within an appropriate design. Essentially, there are two designs (**Table 9**), the first is a targeted design whereby the response of water quality is monitored in relation to where actions are known to have been implemented. This design is often used to establish the effect of an action in a specific region but can yield erroneous results unless designed to account for differences in biophysical conditions, changes in production (increasing losses) and the effects of climate. The second approach is a distributed design where water quality sampling occurs over a range of sites along with the collection of other data like changing climate and production (influencing contaminant losses). The hope of a distributed design is that an appropriate statistical analysis will remove the influence of climate and production to show reductions attributable to actions. As shown in the previous section, the frequency of sampling depends on the magnitude of contaminant reductions relative to variation of contaminant losses in samples (**Figure 8**). This variation is caused by different biophysical conditions or changes in climatic conditions. Although robust farm environment plan data are needed irrespective of the monitoring programme, some programmes are more likely to be able to attribute water quality improvements to the implementation of actions. Here, we give examples of targeted and distributed designs to attribute improvements in water quality to the implementation of actions.

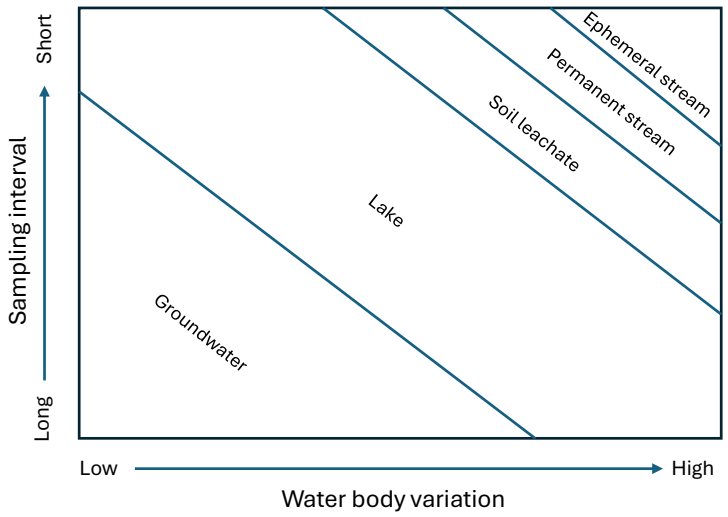
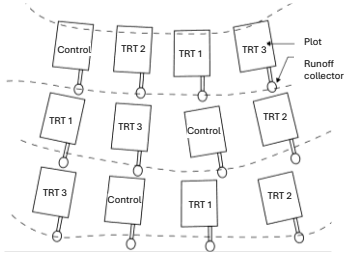
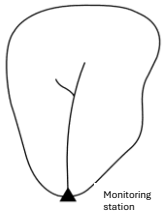
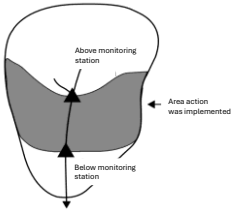
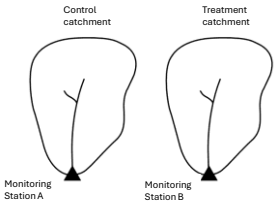
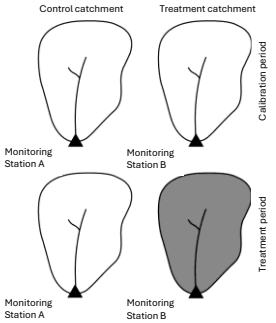


Figure 8. Relationship between the water quality sampling interval (viz. frequency) and variation in water quality and quantity in different water bodies. Diagram modified from [180].

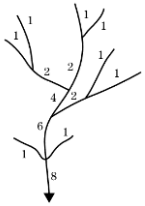
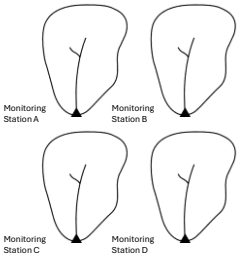
Table 9. The advantages and disadvantages of common water quality attribution designs and the role of the five-step process for farm environment plan data to show (with a degree of confidence) that actions are improving water quality. Diagrams modified from [180].

Method of sampling for attribution	Advantages	Disadvantages	Use in five-step process
Targeted design			
Treatment plots 	Event-based sampling of contaminant losses and response to treatments relative to a control	Measured effect at the plot scale is attenuated at larger scales leading to an over-estimation of the catchment scale impact of the action	The most common method for establishing if a mitigation action is effective, but not suitable for widespread use owing to expense
Single site / catchment 	Simple design to monitor changes before and after the implementation of actions	Effect of actions often swamped by the interactions of climate on catchment hydrology	Only suitable for detecting large changes caused by actions and requires a long period of sampling to account for climate effects. Does not produce knowledge of how general the effect may be across different environments.
Above and below action (grey)	Useful for isolating critical source areas and is less confounded by the influence of climate on hydrology	Below samples are not independent of above samples and may be influenced by biophysical processes and not the action in the area implemented	Tends to be the most common method used by catchment groups (along with single sites), but must be paired with good record keeping, and a high level of confidence that the catchment is biophysically uniform to gauge the effect of implementing actions



Two catchments 	Two catchments can be easily identified and chosen	Analysis of the effect will be highly confounded by different biophysical and climate effects	Design should always be avoided because catchments always differ in their biophysical characteristics and their response to both mitigation actions and climate variability. This means that the analysis of effects will be confounded and may lead to misleading conclusions
Paired catchments 	Relationships derived during a period of calibration can remove differences in biophysical conditions and hydrology, allowing the treatment (i.e., action) effect to be better determined during second, treatment phase	Water quality response during the treatment phase may be influenced by climatic effects that are different to the calibration period. The design is costly.	Robust design to determine the effect of an action at a catchment scale, but still prone to climate effects.

Distributed design

Synoptic survey  	Grab sampling at sites known to accumulate contaminants.	Snapshot dependent on the conditions at the time of sampling	Only suitable to detect poor practice at times of high risk of contaminant loss (e.g., non-compliant effluent discharge)
Multiple catchments  	Can work across catchments with heterogeneous biophysical conditions and implementation, if the degree of implementation is measured in each catchment. The analysis will produce an overall "regional average" effect size (average across the range of environments sampled. May also be able to remove climate effects if sampled for long enough.	Must have a wide-enough gradient in, for example, management to establish a statistically robust effect. Expensive to run as monitoring programme must be run for > 10 (and preferably 20) years. Also prone to confounding changes (trends) over this time.	To account for the effect of confounding factors such as changing loss rates either through increasing intensification or the implementation of actions, appropriate water quality sampling must be accompanied by robust record keeping or where and when actions are implemented.

Studies using a targeted design

There is a moderate body of evidence to show that targeted designs using the five-step process with a good record of the implementation of on-farm mitigation actions has been associated with water quality improvement. Most have combined the recording of the timing and placement of actions with monitoring at either single or multiple catchment sites.

In a review of 25 studies across 24 countries examining the water quality response to on-farm action, Melland et al. [181] found improvements in 17 studies commensurate with the implementation of practices including landscape engineering (use of sediment traps and bunds to retain runoff), better crop management (including variable rate irrigation and lower rates of N and P fertiliser) and reductions in stocking rates or conversions to lower intensity land uses. However, it took 4-20 years to confidently detect the improvements (usually with monthly monitoring). It was important to note that all these catchments were small (1-100 km^2) meaning that there is likely to be a substantial proportion of the catchment where actions were implemented.

New Zealand has a long history of farm planning beginning with the development of the Land Use Capability system in the 1940s and Soil Conservation and Catchment Boards [182]. However, while there is much evidence to support specific mitigation actions via treatment plots, less evidence is available to attribute the implementation of actions to water quality improvement at a catchment scale. Some notable exceptions include:

- After decades of erosion, farm planning in 1990's in the Wither Hills (Marlborough) targeted remedial works, in-stream channel structures and planting which reduced erosion by 60-80% [183]. Planning and erosion control has continued to this day [184]. An *ex-post* analysis of the internal rate of return of planting in the Wither Hills (Marlborough) was ~8% resulting from less erosion, increased pasture growth and a doubling of stocking rates [185].
- Riparian planting conducted as part of farm planning for the Kaituna Catchment Control Scheme in the 1980s. Sampling after planting showed an 85% decrease in particulate-P (the difference between total P and dissolved P) and a 40% decrease particulate-N concentrations compared to before planting [186].
- Likewise, the need for riparian planting in the 1990s and subsequent planting of 13,000 km of streams in the Taranaki ring plain (through farm environment plans) has been attributed (though measurement and modelling) to an 85% improvement in invertebrate metrics and the reduction of *E. coli* concentrations at all sites (n=11) monitored [187].
- Finally, beginning in 2001, five dairy-dominated catchments (from 598 to 2,480 ha) were studied. From 2001 to 2010, comprehensive extension advice was given to farmers consisting of workshops, and measurements made of stream water quality, flow, farm practices (via surveys and modelling) to identify the magnitude of contaminant losses and provide solutions to address site-specific contaminant losses (**Figure 9**). From 2011-2020, only water quality monitoring and farm practice surveys and modelling of losses were continued. Of the water quality contaminants (including dissolved reactive P, total P, ammoniacal-N, nitrate-N, suspended sediment and *E. coli*), 83% of water quality trend directions were either improving (n=16) or showed little confidence in change (n=9) from 2001-2010. Over the 20-year dataset, 20 out of 30 contaminant-catchment combinations

(67%) were improving, but nine were degrading, dominated by nitrate-N (n = 4), dissolved reactive P (n = 2) and *E. coli* (n = 2). Importantly, abrupt and sustained decreases in contaminant concentrations, were correlated with the timing of recorded on-farm practice changes despite increases in some contaminant losses from rising production. These practices were mainly associated with transition from direct discharge of farm dairy shed effluent to waterways to land application, and the capture of effluent from off-paddock facilities (like stand off or feed pads). However, the increase in production overwhelmed the effect of actions resulting in an increase in nitrate-N concentrations caused by transitioning from flood to spray irrigation and a commensurate increase in cow numbers and nitrate-N leaching. These were the first catchment-scale data in New Zealand to indicate that extension advice and on-farm practice change have helped to improve overall water quality over time.

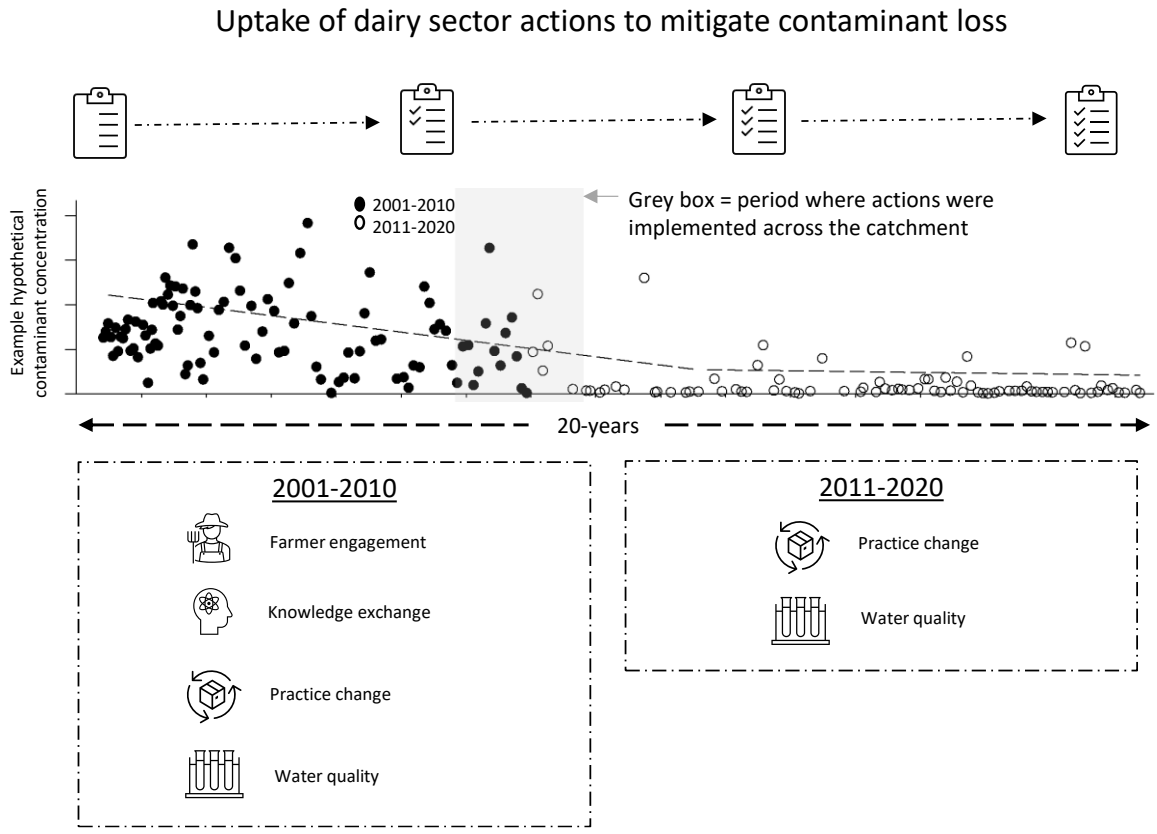


Figure 9. Timeline of advice and monitoring in the five dairy catchments that led to the success detection and attribution of water quality improvements to on farm actions to mitigate contaminant loss.

Factors that made these findings possible in the 5 dairy catchments were partly designed and partly serendipitous. For example, monitoring was done at a frequency that over time was able to detect change: i.e., statistical power for each catchment was >80% for the magnitude of change seen over 20 years and for abrupt changes (detected as step changes in water quality). These changes were supported by a strong dairy-farming signal (often 80% of land use) and the small catchment size which led to short groundwater lag times (< one year). Furthermore, while investigations identified where contaminants were coming from and the most cost-effective actions to mitigate contaminant

loss, the recording of farm practices and modelled farm-scale contaminant losses was also done at the right frequency to be reveal abrupt changes in water quality. However, it is unlikely that delaying the frequency of collecting farm survey data and modelling losses to longer than every three years would have the same success. In short, these catchments could be used as examples of how to monitor for change.

Studies using a distributed design

At large scales it can be hard to attribute change to the implementation of mitigation actions. This is caused by stronger land use and climatic (leading to hydrologic) variation than variation in contaminant concentration from actions. The scale at which the effect of actions in **Table 2** was established (and based on a limited number of studies) tended to be small dominated by low order streams (Strahler order 1 or 2), while the mean order for monitored New Zealand sites is ~6. Understandably, climate and land use will vary greatly from the headwaters to a 6th order river, meaning that the likelihood of all land uses using the same mitigation action (at the same time) is slim. However, efforts have been made to disentangle climate from land use and land management signals. For instance, Scarsbrook et al. [188] then Snelder et al. [121] examined trends in national river monitoring sites over 20 years period. They found that *between* site variation in contaminant concentrations was influenced most by the Southern Oscillation Index (a measure of climate variability). Snelder et al. [121] then found between site variation was attributable to stocking intensity for pastoral land uses and the area in plantation forest. This implies that to attribute water quality change to mitigation actions requires good knowledge about how all contributing factors change over time and that the effect of actions is best identified where the signal from actions is strong (i.e. either a small to medium sized catchment, or the monitoring site is close to the action).

How we know that river water quality is affected by climate variation

When assessing the effectiveness of farm plan policy to improve water quality, climate variation is a confounding factor because it may amplify or counteract the effects of the policy on water quality changes [121,188-190]. Climate effects on river water quality are visible in plots of rolling trend assessments [190]. These plots are constructed by first performing trend analyses for combinations of sites and variables that have a fixed assessment period, for example 10 years. For each site and variable, trends are assessed for windows in time with a fixed duration and ending in a specified 'end year'. For each site, windows are incremented by one year (i.e., a "rolling window") from the beginning of the monitoring data time series. For each window, the site trend is expressed as the confidence the trend was decreasing, which is denoted C_d. Confidence varies between zero and one with high and low values (categorised as per **Table 10**) indicating high and low confidence, respectively. Because trend direction can only be increasing or decreasing, low confidence that the trend was decreasing is equivalent to high confidence the trend was increasing.

Table 10. Categories used to convey the level of confidence that a trend was decreasing. The confidence categories are used by the Intergovernmental Panel on Climate Change (IPCC; [191]).

Categorical level of confidence trend was decreasing	Value of C_d
Virtually certain	0.99 – 1.00
Extremely likely	0.95 – 0.99
Very likely	0.90 – 0.95
Likely	0.67 – 0.90
As likely as not	0.33 – 0.67
Unlikely	0.10 – 0.33
Very unlikely	0.05 – 0.10
Extremely unlikely	0.01 – 0.05
Exceptionally unlikely	0.00 – 0.01

Figure 10 shows the results of a rolling windows analysis for five water quality variables (visual clarity (CLAR), dissolved reactive phosphorus (DRP), nitrate-nitrite-N (NNN), total suspended solids (TSS) and turbidity (TURB)) at up to 158 river monitoring sites in Canterbury for 5- and 10- and 15-year periods. In **Figure 10**, each panel represents the rolling trends for period ending in the 'End year' shown on the x-axis for a duration (column) and variable (row). The bars shown for each year indicate the proportions of sites having the nine categorical levels of confidence in direction (**Table 10**). **Figure 10** hints at periodic fluctuations in the proportions of sites assigned to the confidence categories for all variables. The amplitudes of these fluctuations decrease as the duration of the trend period increases.

Analyses like those shown in **Figure 10** have produced the same results in other regions and nationally [121,190]. All these studies indicate that there are synchronous water quality changes occurring implying that large scale processes are forcing trends across sites at the same time. The obvious processes are climatic because these occur over large spatial scales (i.e., national to global scales) and exhibit strong inter-annual variation. An example of a large-scale process that causes inter-annual variation in climatic conditions is the El Niño-Southern Oscillation (ENSO) cycle. ENSO cycles tend to be quasi-periodic with irregular phases that occur at timescales ranging from two to seven years [192]. More detailed studies have shown that indicators of climate variation, such as the Southern Oscillation Index (SOI) and monthly rainfall, can partly explain variation over time in trend strength and direction at individual and for aggregated sites [121,189]. Taken together with international literature, there is little doubt that climatic variation influences water quality changes (and trends). The component of a water quality change (or trend) that is associated with climatic variation is a confounding factor that is ideally accounted for in any analysis of data that aims to assess policy effectiveness.

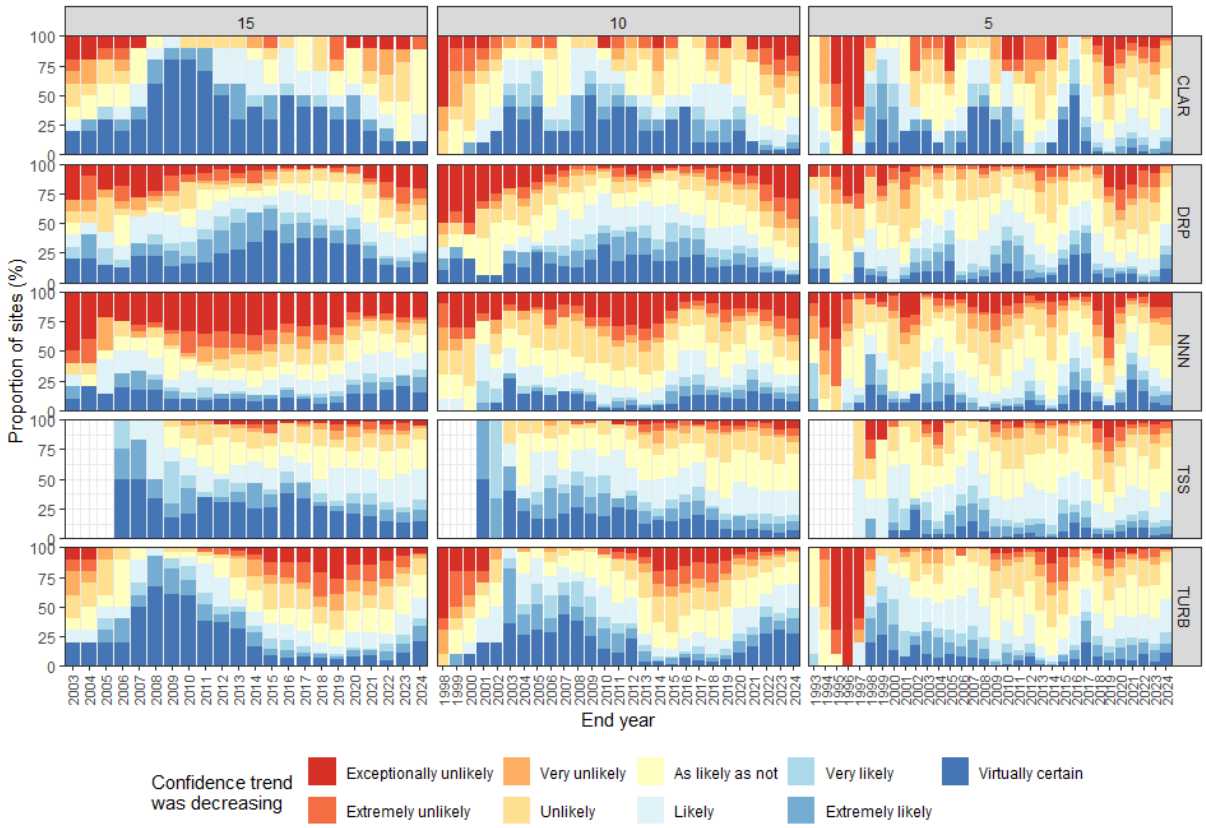


Figure 10. Summary plot representing the proportion of sites in each confidence category defined in Table 1 for up to 158 river water quality monitoring sites in Canterbury. The bars show the proportion of sites belonging to each confidence category for rolling windows of 15-, 10- and 5-years duration (columns) and each variable (row). It is noted that the number of sites represented by each bar is not constant and generally increased with time.

Controlling water quality trends for climate effects

Because a component of water quality change (or trend) is associated with climatic variation, analyses that aim to assess policy effectiveness will ideally account for, or control for, its' effect, i.e., hold a potentially confounding variable constant or adjusting for its influence, ensuring that any observed effects are due to the variable of interest rather than the confounding variable. An important point to note is that the influence of inter-annual climate variation on water quality trends decreases as the trend period duration increases (**Figure 10**). This is because at longer timescales, short-term climatic variations merge into background variability. This is evident in **Figure 10** where the the amplitudes of the fluctuations in water quality trends are largest for the shorter duration.

Ideally, we would have good information about the effects of actions shortly after implementation (e.g., after 5 years). However, because water quality trends are often strongly influenced by climate at timescales of 5 to 10 years, they cannot be regarded solely as indicators of the influence of human activities. Ideally, the confounding influence of climate variability is removed from water quality trend assessments so that the trend can be regarded purely as a signal of the anthropogenic response of catchment.

It would be ideal if trends could be adjusted to control for the effect of climate. Covariate adjustment in statistics involves using measured values of variables (covariates) that might otherwise

confound the outcome of interest. Indicators of climate variation, such as the SOI can partly explain between-site variation in water quality trends and have been suggested as relevant covariates [193]. However, although climate indicators provide evidence that trends are partly driven by climate variability, the amount of variation that is explained, and the inconsistency in explanation between sites, means that they cannot be used to estimate the purely anthropogenic response of catchments [189]. This is because temporal variation in water quality is a complicated outcome of mixtures of spatially and temporally variable catchment processes such as contaminant generation, storage and transport [194-196]. These processes cannot be well described by simple climate indicators.

An alternative approach to controlling for the effect of climate variation on water quality is taken by the modelling approach called Weighted Regression on Time, Discharge and Season (WRTDS; [197]). WRTDS links river water quality data with hydrological records comprising mean daily flows. Because hydrological variation is more directly associated with climate variation than water quality, WRTDS can control the influence of year-to-year variations in discharge and produce trend assessments that represent the water quality change in the absence of climate variation. This means the model can describe two types of trends. First, the 'raw trend' is an assessment of the actual change in the value of the water quality variable over time, which is analogous to a traditional water quality trend assessment. Second, WRTDS quantifies the 'flow-normalised trend', which represents an assessment of the anthropogenic component of the trend (i.e., that is due to human activity without the influence of climate).

It is therefore recommended that, as much as possible, flow-normalised WRTDS trends are used to account for the confounding effect of climate when using river water quality data to assess policy effectiveness. Flow-normalised trends could be used in analyses of water quality responses to management actions at the level of sites [198] or across multiple sites [193]. However, it is emphasized that WRTDS is a model, and it cannot entirely remove the confounding influence of climatic variation. In addition, the WRTDS model is more complicated and requires that water quality monitoring sites are associated with continuous flow records. This recommendation applies to all the designs mentioned above. The only possible exception is the above and below design. However, even in the design the assumption of homogeneity of response to climate variation at the upper and lower sites is not often justifiable.

Rigorous attribution

Establishing whether a policy has been successful in improving water quality involves determining if that policy has caused, or contributed to, an observed change (i.e., percentage decrease or trend over time) in a water quality contaminant. This is referred to scientifically as 'attribution'. Observed changes in water quality are generally caused by multiple drivers and are the aggregate of many actions taking place across a catchment. This means that the effect of policy is likely to be confounded by other drivers, which complicates analyses that attempt to assess policy effectiveness and reinforce the use of on-farm actions.

Rigorous attribution involves quantifying the relationships between water quality changes and the drivers [199-201]. These analyses aim to isolate the contribution of the policy to the water quality changes and therefore the other drivers are treated as confounding factors. Weaker alternatives to rigorous attribution include qualitative reasoning, references to previous studies and simple

speculation. When attribution is weak, there is a risk that conclusions about the effectiveness of a policy will be wrong.

Because rigorous attribution involves quantitative analysis, it requires data describing changes in both the water quality and the drivers. A key reason that rigorous attribution is rare for assessing policy effectiveness is that suitable data characterising spatiotemporal variation in actions (and the effect on contaminant losses) like those contained in farm environment plans are scarce and fragmented. Monitoring that includes measurements of all drivers (i.e., including the implementation of actions and confounding variables) is therefore required if policy effectiveness is going to be rigorously assessed.

To help understand the importance of rigorous attribution and the effects of a confounding factor, a synthetic simulation (i.e., made up data) scenario is presented below. The scenario represents a situation where a regional policy aiming to decrease contaminant discharges from a target land use by the implementation of actions from land use has been implemented for 10 years. A water quality variable representing the concentration of a contaminant has been monitored in the catchments of 60 water quality monitoring sites across the region. The water quality change at each monitoring site has been quantified as the flow-normalised confidence the 10-year concentration trend was decreasing (i.e., C_d). This implies that the confounding effect of climate has been controlled, and the trend can be understood as the anthropogenic component of the water quality change. Because the water quality variable is a contaminant concentration, values of C_d >0.5 indicate water quality improvement and values of C_d <0.5 indicate water quality degradation.

However, let us also assume that data have been collected on the implementation of actions in the catchment of each monitoring site, and that that there have been changes that the land manager has made that have affected production on the targeted land use. An increase in production may occur in response to market conditions and be achieved through increased fertiliser use and stocking rates but may also lead to increases in contaminant losses. The change in production in the target land use will be a confounding variable in the assessment of the effectiveness of the policy. However, assume that data has also been collected that describes the changes in production.

A synthetic dataset was created to represent this scenario by generating a random variable representing the proportion of the target land use in each of the 60 catchments ranging from zero to one. A second random variable was generated to represent the proportion of the target land use where the policy of implementing actions was followed that ranges from 0.1 to 1. A third variable, denoted ‘Intervention’, was generated as the proportion of the catchment occupied by the target land use where the policy of implementing actions was followed. Intervention has a positive effect on C_d for each site (i.e., improves water quality) that is proportional to the amount of implementation in the upstream catchment. The change in production is proportional to the proportion of the catchment occupied by the target land use and is denoted as the variable ‘Production’. For this simulation, the magnitude of the effect of the Production is twice that of Intervention. The synthetic dataset also includes some random noise that represents other drivers of water quality change that are not measured.

Figure 11 is a plot of the value of C_d for each catchment against the variable Intervention. **Figure 11** indicates a negative association between C_d and Intervention. This suggests that the policy of implementing actions was ineffective because catchments with the greatest intervention were associated with the most degradation (i.e., lowest C_d values). However, in general, it would be

unrealistic to assume that the analysis presented in **Figure 11** indicates the true effectiveness of the policy because there would likely have been other changes in the catchment over time. For the synthetic dataset presented here, we know the relationship represented by **Figure 11** is misleading because Intervention has a positive effect on Cd and the effect of Production has not been accounted for.

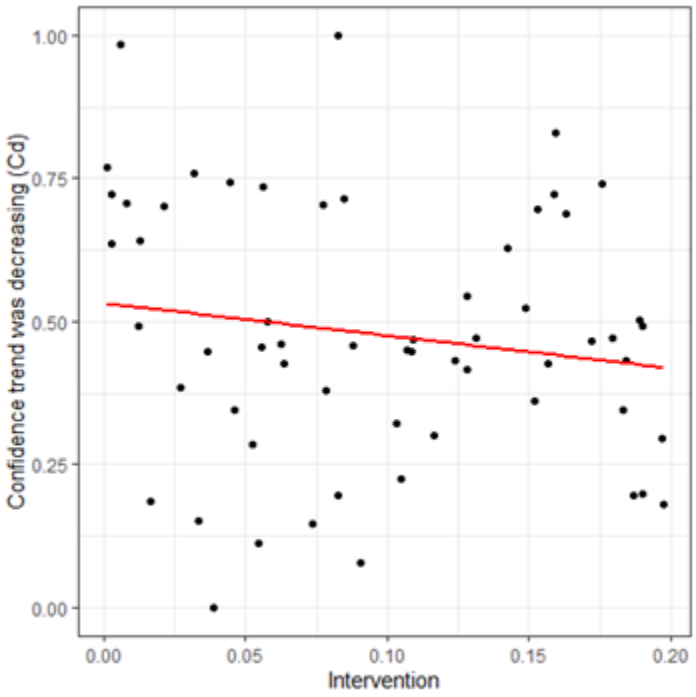


Figure 11. Simulated results of the Scenario 1 showing site values of Cd versus Actions.

A more rigorous analysis of the synthetic dataset is to fit a linear model that represents Cd as the combined effect of Production and Implementation. This model is shown as the two-dimensional plot shown in **Figure 12**. The surface shown in **Figure 12** represents C_d and decreases with increasing Production and increases with increasing Intervention, which is consistent with the synthetic data. Thus, when the model accounts for the combined effect of both Production and Intervention, the results indicate that Intervention is positively related to C_d (i.e., improving water quality). Additionally, Production is negatively related to C_d and its contribution is approximately twice that of Intervention.

Had we only measured the implementation of actions (**Figure 11**), we would have concluded that the policy was ineffective. However, when our analysis was more rigorous and included the confounding effect of increasing Production (**Figure 12**), we concluded the policy had been effective but was often counteracted by the effects of production changes. In other words, not accounting for the confounding effect of Production would have resulted in incorrect conclusions about the effectiveness of the policy to implement actions. Note that, depending on the circumstances, it is quite plausible that an assessment could incorrectly conclude that a policy was effective, when in fact it was not. An example of this for different farm types was modelled comparing the likely national contaminant loads of N and P in 2015 if the practices of 1995 had been employed [202]. The implementation of actions was estimated to have reduced P losses by 25% but had not reduced N

losses owing to an increase in production and N losses. However, after taking production into account, losses without actions would have been 45 and 98% greater for N and P, respectively.

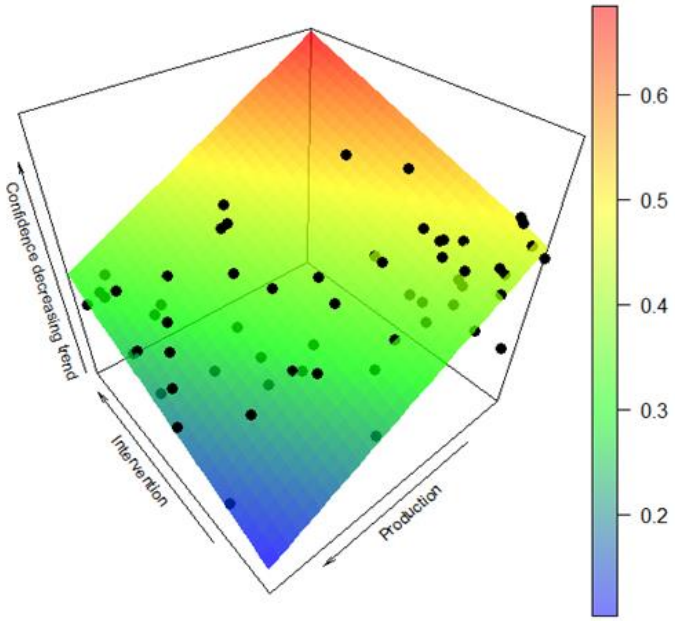


Figure 12. Three-dimensional representation of a linear model expressing change in C_d as a function of Production and Intervention. The black points represent the simulated monitoring site C_d values and the coloured surface represents the fitted model. The surface colours indicate the fitted Cd values from blue, indicating low confidence trend was decreasing (which is equivalent to high confidence trend was increasing) to red values, indicating high confidence trend was decreasing.

Data gaps that need to be filled to improve the likelihood of attributing water quality improvement to actions on farm

During the writing of this report, it became clear that several data gaps remain to improve our confidence in attributing water quality improvements to farm actions. Several gaps are addressed in the two following case studies or have been included in the recommendations for each step in our 5 step process. However, for clarity and brevity we outline data gaps in order of priority:

1. Regularly updated (e.g., every three years at least) and audited farm scale data on contaminant losses as loads or yields.
2. Regularly updated (e.g., every three years at least) and audited farm scale data on what, where and when mitigation actions were implemented or plan to be implemented.
3. Regularly updated (e.g., every three years at least) and audited risk map of contaminant losses from areas of the farm. Note that risk must be quantitative and include biophysical and management risks.
4. Water quality sampling at the correct frequency and in the right place to maximise the likelihood of detecting change. This may take the form of monitoring sites that are

representative of many other catchments. Simultaneous intensive monitoring of both water quality and farm losses and actions will maximise the chance of detecting change.

5. Data on the lag time between contaminant loss from the farm and the receiving waterbody. With recent advances in the assessment of nitrate-N time lags, new data should focus on other contaminants.
6. Data for the performance of mitigation actions in different locations and over the long-term, recognising that most actions are only tested for, at most, three years, but are expected to last 15-20 years.
7. Data linking and confirming the flow path contaminants take from the site of loss to the receiving waterbody. This includes data on transformations during flow leading to contaminant enrichment or dilution.
8. Data on catchment climate variability and land surface recharge to inform appropriate statistical analyses.
9. Targets that can be expressed as, for example, a percentage reduction in existing contaminant losses in known locations.

Case Study (groundwater): Determining the likelihood of on-farm actions in the Central Plains Water scheme to reduce nitrate-N concentrations in nearby groundwater wells by 30%

This case study outlines evidence from a large irrigation scheme (Central Plains Water) that has managed and audited farm environment plans (Steps 2 and 3). We focus on assessing when the percentage reduction in nitrate-N recorded in these plans would be most likely detected in nearby wells. We use data on farm environment plans to show evidence for Steps 2 and 3. For Step 4 we assume the following:

1. That there was no nitrate removal during the transport path (e.g., all groundwater is oxic and nitrate behaves as a conservative tracer)
2. That the age distribution of water in the well is static and does not change over time.

This case study focuses on maximising the likelihood of detecting change. It does not (yet) attribute changes solely to the implementation of actions because change could be caused by actions or other factors (e.g., climate [121]). However, it does identify those wells that are highly likely to have been influenced by actions because they have short lag times and strong changes following the quick implementation of actions. Furthermore, because our power calculations are conservative and based on data that are influenced by variation from sampling, biophysical processes, climate and land management (and actions), our estimates of the time taken to detect change will be longer than if variation from these sources had been removed. This implies that attribution of change to actions will improve over time. An example is given in **Figure 13** for a nitrate-N concentration in a theoretical groundwater bore showing the reduction in variation, and hence samples need to detect change as, variation is accounted for and removed.

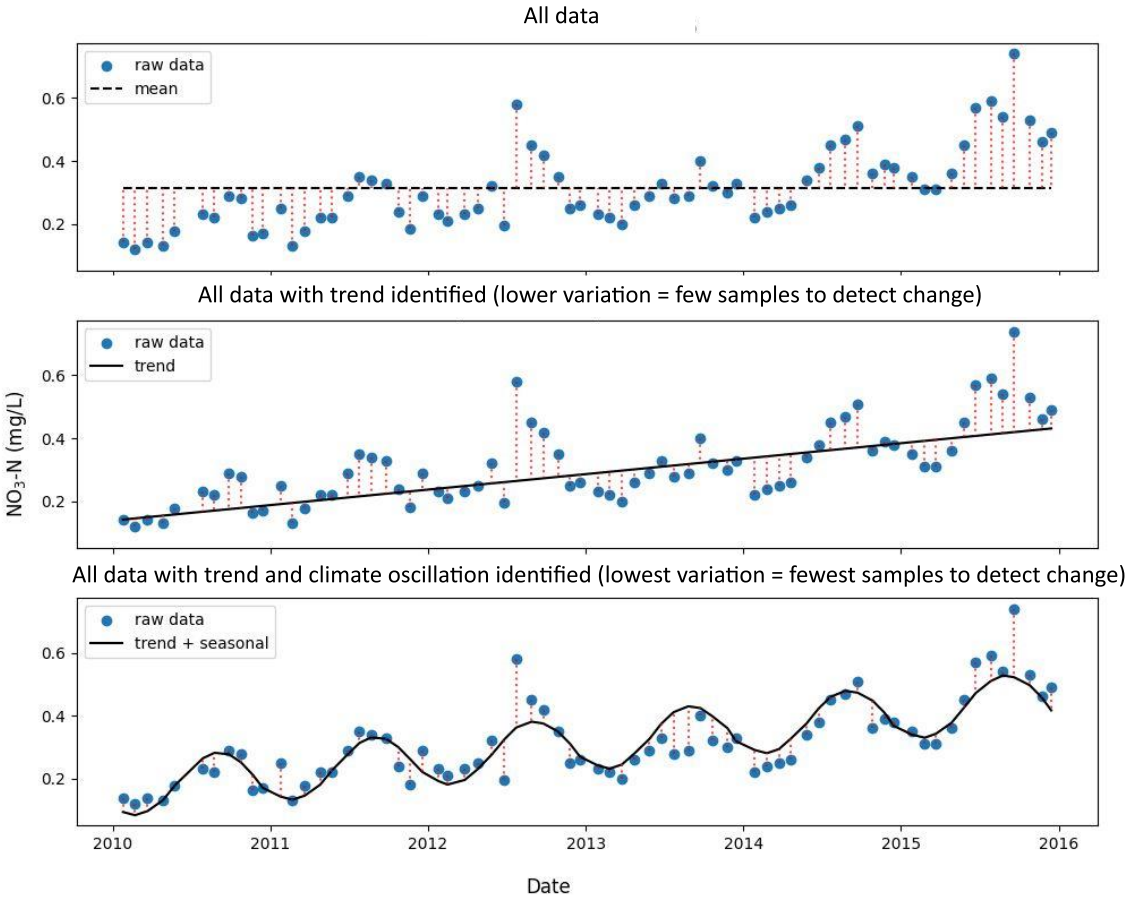


Figure 13. Plot showing the variation (dashed red lines) from a modelled relationship over time with no trend detected, a trend detected, and a trend and climatic influence detected. Less variation means fewer samples are required to detect change.

Because it can take many years (even decades) to collect sufficient data to maximise the likelihood of detecting change, eventually the time taken to collect samples to prove change is real may coincide with having sufficient data to also remove the influence of climatic factors, thereby focusing any change on the implementation of actions. Having long-term groundwater quality sites is therefore crucial in attributing the implementation of actions as the most likely cause of water quality improvement.

Introduction

The Canterbury Regional Council has been using a consistent farm environment planning system for the implementation of the Land and Water Regional Plan [203]. Since 2016, 5,825 farm environment plans have been independently audited, Canterbury-wide. The number of audits was initially dominated by ISO-accredited plans assessed of irrigation scheme shareholders, but now equally split between irrigation scheme shareholders and individuals [145]. Auditors assess compliance with the implementation of industry-agreed good management practices (as of 2015, but recognising that these may change) [131] and apply an A to D grade (**Table 11**).

Table 11. Frequency of auditing of farm environment plans by independent auditors for compliance with the Canterbury Regional Land and Water Plan. Data from Canterbury Regional Council [145].

Submission	Grade A (Compliant and achieving GMP)	Grade B (Compliant and on track to achieve GMP by the next audit)	Grade C (Non-compliant and not achieving GMP)	Grade D (Serious non-compliance and not achieving GMP)
Individual farms	3 years	2 years	1 year	6 months
Farms connected to ECan accredited audit programmes (industry programmes)	Dependent on approved ISO accredited programme timeframe	Dependent on approved ISO accredited programme timeframe	1 year	6 months
Farms connected to an Irrigation scheme, Principal Water Supplier or Hurunui Waiau Collective	4 years	2 years	1 year	6 months
Change in management or significant change in farm systems	1 year	1 year	< 1 year	6 months

The Central Plains Water (CPW) scheme irrigates 45,000 hectares of farmland between the Rakaia and Waimakariri Rivers. The first stage was constructed in 2014/15, with stage 2 and 3 coming on-line in 2017/18. As part of the consent to construct and operate the scheme, Central Plains Water Limited (CWPL) was required to install 20 groundwater bores (**Figure 14**), with quarterly sampling for nitrate-N (and a range of other contaminants, not analysed here) to occur two years prior to the scheme operating. An additional 29 river sites were to be sampled monthly for the same range of contaminants. The data from these sites have been combined with data from farm environment plans to demonstrate the five-step process (**Figure 1**).

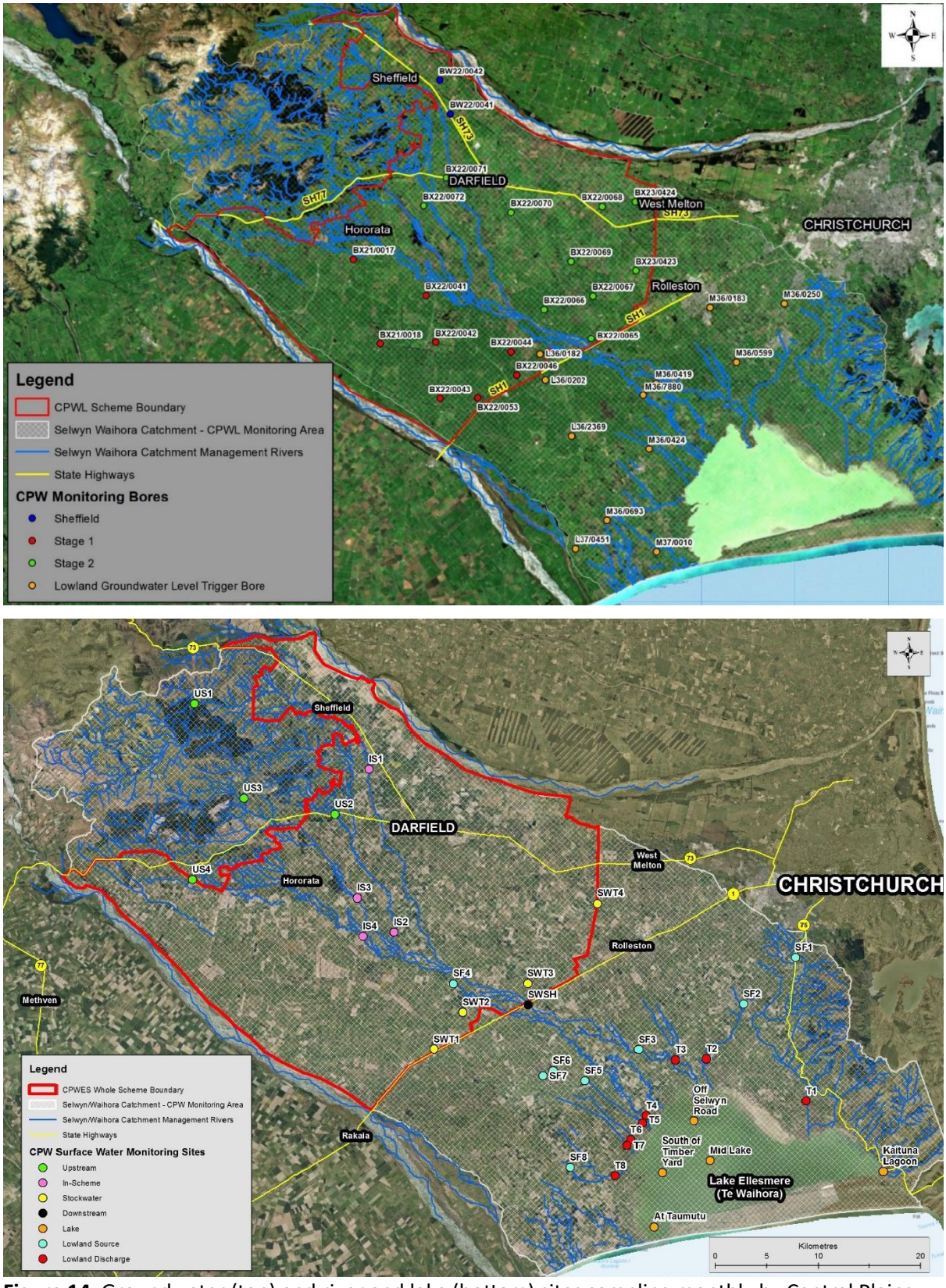


Figure 14. Groundwater (top) and river and lake (bottom) sites sampling monthly by Central Plains Water Limited.

Step 1 (Determine target) and Step 2 (Select farm actions)

Plan Change 1 of the Canterbury Regional Land and Water Plan outline that all farms in Selwyn Waihora with N losses greater than 15 kg N ha^{-1} yr^{-1} will need to reduce their N losses by 30% for dairy, 22% for dairy support, 20% for pigs, 7% for arable, 5% for irrigated dry stock or for fruit, viticulture or vegetables, and 2% for dryland dry stock [203]. To achieve reductions, farmers must have a farm environment plan that assesses and applies a list of suitable GMPs (**Table 12**). Plans are audited for GMP once every three or four years, or more frequently if on-track (grade B) or non-compliant (grades C and D).

Table 12. List of industry-agreed good management practices [131] required to comply with the Canterbury Regional Land and Water Plan [203].

Target	Good management practice
Whole farm	Identify physical and biophysical characteristics of the farm and to water quality and manage appropriately
	Maintain accurate and auditable records of farm inputs, outputs and management practices
Land	Manage farming operations to minimise direct and indirect losses of sediment and nutrients to water, and maintain or enhance soil structure, where agronomically appropriate
	Manage periods of exposed soil between crops/pasture to reduce risk of erosion, surface runoff and leaching
	Retire all Land Use Capability Class 8 and either retire, or actively manage, all Class 7e to ensure intensive soil conservation measures and practices are in place
	Identify risk of surface runoff of sediment and faecal bacteria on the property and implement measures to minimise transport of these to waterbodies
	Locate and manage farm tracks, gateways, water troughs, self-feeding areas, stock camps, wallows and other sources of runoff to minimise risks to water quality
	To the extent that is compatible with landform, stock class and intensity, exclude stock from waterways
	Monitor soil phosphorus levels and maintain them at or below the agronomic optimum for the farm system
Plants	Manage the amount and timing of fertiliser inputs, taking account of all sources of nutrients, to match plant requirements and minimise risk of losses
	Store and load fertiliser to minimise risk of spillage, leaching and loss into waterbodies
	Ensure equipment for spreading fertilisers is well maintained and calibrated
	Manage the amount and timing of irrigation inputs to meet plant demands and minimise risk of leaching and surface runoff
	Design, calibrate and operate irrigation systems to minimise the amount of water needed to meet production objectives
Animals	Store, transport and distribute feed to minimise wastage, leachate and soil damage
	Ensure the effluent system meets industry specific Code of Practice or equivalent standard
	Have sufficient, suitable storage available to enable farm effluent and wastewater to be stored when soil conditions are unsuitable for application
	Ensure equipment for spreading effluent and other organic manures is well maintained and calibrated
	Apply effluent to pasture and crops at depths, rates and times to match plant requirements and minimise risk to waterbodies
	Select appropriate paddocks for intensive grazing, recognising and mitigating possible nutrient and sediment loss from critical source areas
	Manage grazing to minimise losses from critical source areas

Step 3 (Plan and implement actions)

For the CPW scheme, in-house farm environment plans were first audited (over a six-month period) by CWPL in 2016/2017 when 97 shareholders joined the scheme in 2015. On average 98 audits have been done each year up to 2023/2024 with a maximum of 185 in 2019/2020 following on from an additional 164 shareholders joining the scheme in 2017 and 2019, and a minimum of 36 in 2022/2023 coinciding with a non-audit period for most shareholders. Data were combined into two-year periods to reflect audit and non-audit years. A small proportion of audits were conducted by Synlait as part of compliance to their Lead with Pride programme [204]. These audits were considered equivalent to the audits conducted by CPWL (**Table 13**).

Table 13. Count and the proportion (in parentheses) of audits of farm environment plans conducted by CWPL, aligned audits by Synlait under their Lead with Pride programme [204], or not audited for two-year periods beginning in July of the first year and ending in June of the final year. Note that by 2015, 2017 and 2019, 97, 131 and 262 properties were in the scheme, respectively.

Period	Audits completed	Aligned audits	Not audited
2016-2018	132 (68)	22 (11)	40 (21)
2018-2020	311 (66)	31 (7)	129 (27)
2020-2022	119 (44)	42 (15)	110 (41)
2022-2024	152 (48)	43 (14)	121 (38)

The proportion of shareholders achieving an A grade was initially 40% and below that expected over the full period of assessment (*chi*-squared test for data from 2016-2024 **Figure 15**). However, the proportion of shareholders receiving an A grade had reached 84% by the 2022-2024 period, with 15% reaching a B grade and 1% receiving a D grade.

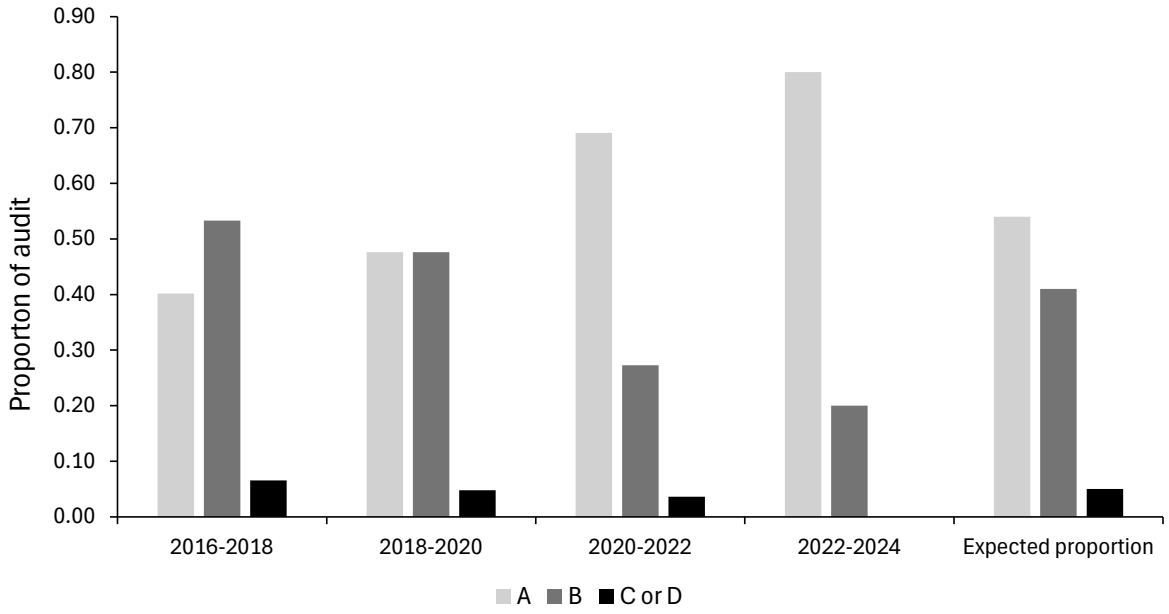


Figure 15. The proportion of grades for farm environment plans audited each two-year period by Central Plains Water Limited compared to the expected proportion for each grade (χ^2 test, P<0.001).

Commensurate with farm environment plans, farm data were collected and a nutrient budget completed using the latest version of Overseer. The N losses estimated in each budget was assessed against the reduction required to meet the scheme's N discharge allowance (as per 2022) and calculate the reduction achieved and remaining. The N discharge allowance in 2022 was 2,676 tonnes while the estimated N losses in 2024 was estimated to be 1,425 tonnes; a 47% reduction. However, as noted in section 3.3 and by other researchers [205], the implementation of actions is variable. Although audited, we were only confident that full implementation of actions occurred in those achieving an A grade. We therefore discounted by half any reduction to a property audited as achieving an average grade B, C or D since the joining the scheme. This decreased the mean reduction at a property scale from 42 to 32%.

Step 4 (Maximise the likelihood of detecting reductions) and Step 5 (Attribution of improvements to actions)

Property scale reductions in the mass of N loss (by leaching) were summed and compared to the likelihood of detecting the reduction in nearby groundwater with and without regard for a lag time. To make this comparison we:

1. Isolated a zone to capture the leaching of nitrate-N from topsoil.
2. Augmented existing data on mean residence time (MRT) in ECan-monitored wells with new data for the CPWL-monitored wells to inform modelling of lag times.
3. Modelled the future and past concentration of nitrate-N from each capture zone and receiving well using a new Bayesian Approach to Source Estimation (termed 'BASE').
4. Used the new BASE model to estimate the standard deviation of the model fit to inform a power analysis of the likelihood of detecting a percentage reduction in nitrate-N leaching (in step 3 above) from the capture zone over different time periods and frequency of sampling.
5. Make recommendations on the ability of farm environment plans to reduce and reinforce the reduction of nitrate-N concentrations in groundwater (step 5).

We constrained the source of nitrate-N to a capture zone defined by a simplified bore-gradient approach. We used this simplified approach in lieu of a lengthy model-based approach that would consider the advective, dispersive, and heterogeneous nature of the aquifer, but was beyond the scope of this project. Hence, these source zones should not be used for any purpose beyond high level assessments about detecting change.

The bore-grade approach (**Figure 16**) assumes a grade (Vertical distance / horizontal distance) is used in conjunction with the bore depth and location, and groundwater flow direction to set an 'up-gradient daylight point' as the maximum up gradient extent of the source path. At the bore, the source path is assumed to be 100 m orthogonal to the flow direction. This distance is based on the capture zone radius used in Schedule 1 of the Environment Canterbury Land and Water Regional plan [203]. The width of the source path at the daylight point is defined as 100 times the up-gradient

distance multiplied by 0.1 to incorporate elements of advective flow. Finally, the capture zone is defined as the fractional distance of the path which receives recharge and defined by the age *Fp* parameter (see below). The fraction of exponential flow in the exponential piston flow model is considered the proportion of the distance travelled, which is contributing to recharge relative to the distance travelled to the well. For this work we delineated two different capture zones. The first (default) sets the width at the bore to be 200 m (100 m either side of the bore), has a gradient of 100 m horizontal distance for each metre vertically and a *Fp* set at 0.3 or 0.7 (e.g., 30% or 70% of the path contributes to the source). For the other zone, the width at the bore is set to 1 km, the gradient is 300 m horizontal distance for each metre vertically and a *Fp* set at 1 i.e., all the path contributes to the source.

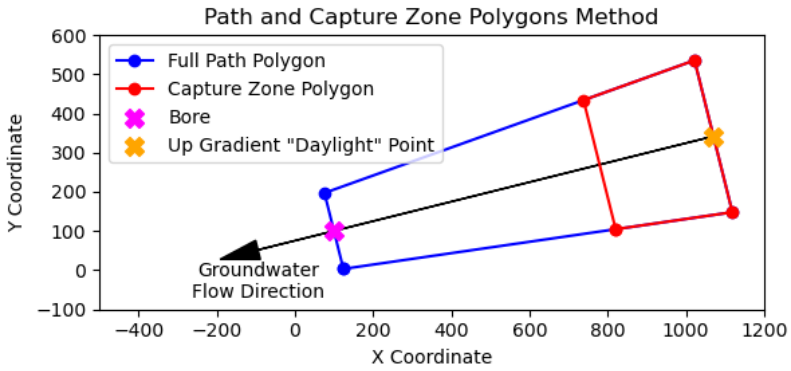


Figure 16. Delineation of the default capture zone used to route nitrate-N losses to a groundwater bore.

Following the isolation of capture zones, we overlayed property specific percentage reductions to obtain an area-weighted percentage reduction for the capture zone (**Figure 17**).

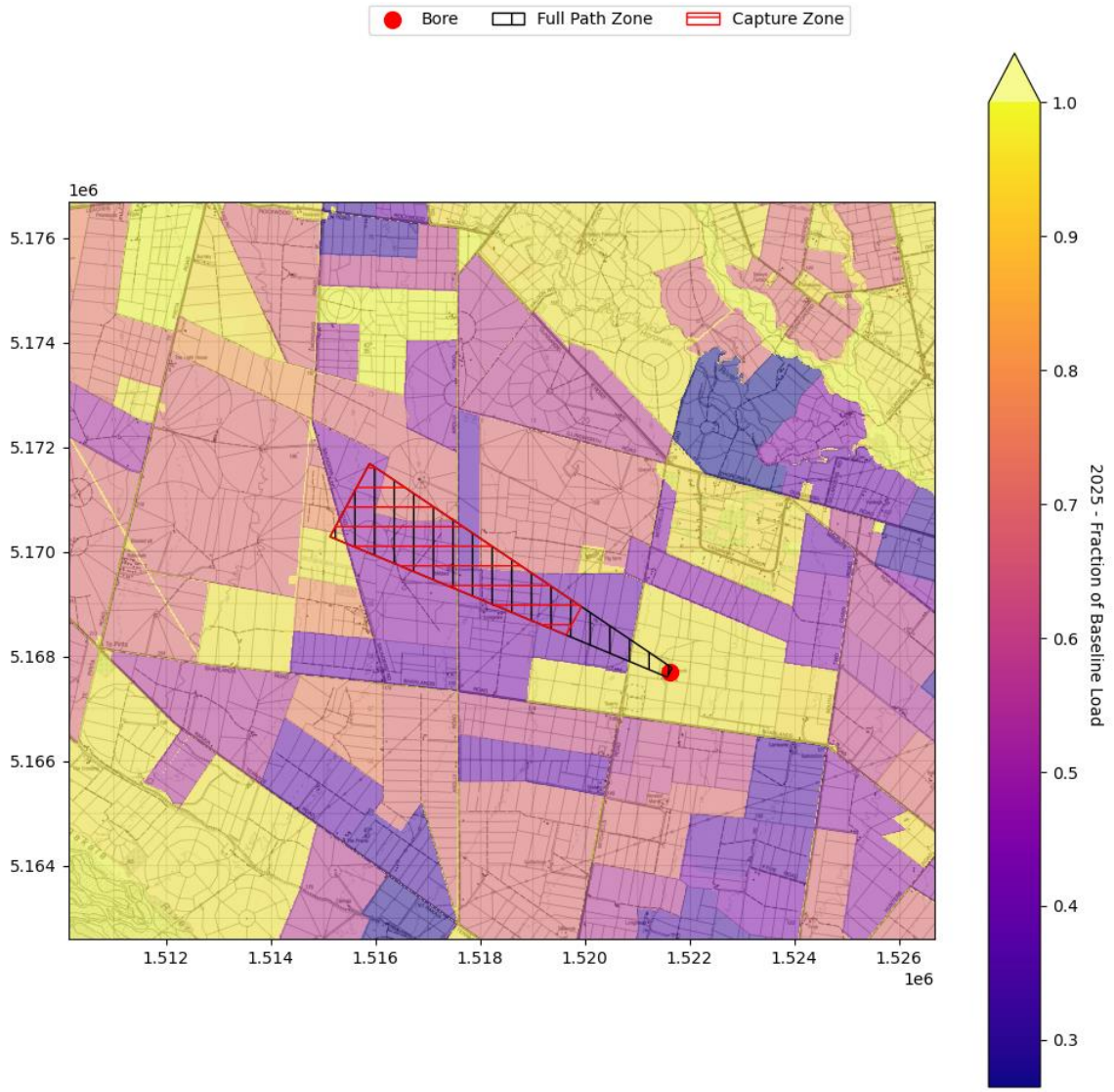


Figure 17. Overlay of the default capture zone with the property specific capture zones.

We augmented those wells without data on tritium concentrations by sampling each well over a six-week period in February 2025.

To predict nitrate-N concentrations from the capture zone and in the well we used a new Bayesian data driven approach. The fundamental inputs to BASE are: 1) the measured nitrate-N concentration history at the well; 2) the estimated groundwater age distribution derived from measured tritium concentrations; and 3) some estimate or constraint on the likely capture zone concentration history. This estimate is known as a prior distribution and can be as simple as a maximum and minimum concentration or more complex incorporating other data sources such as land use history, nitrate-N leaching estimates, etc.

The outputs of BASE include: 1) predictions (a posterior distribution) of the likely capture zone concentration history; 2) past predictions of the nitrate-N concentration history at the well; and 3) the ability to predict future nitrate-N concentrations at the well for a range of future scenarios.

Conceptually the method has four steps. Firstly, we sample a random source concentration history from the prior distribution. Secondly, we use an Exponential Piston Flow Model (EPFM) [206] to predict the nitrate-N concentration history at the well. The EPFM combines a flow path section with exponential transit times followed by a piston flow section, to give a model with parameters of mean residence time (τ_m) and exponential fraction (ƒ). The response function is given by:

$$h(\tau) = 0 for \tau < \tau_m(1-f)$$

$$h(\tau) = (f\tau_m)^{-1} e^{\frac{\tau}{f\tau_m}+\frac{1}{f}-1} for \tau \geq \tau_m(1-f)$$

where τ is the residence time, h(τ) is the flow model or response function of the hydrological system, ƒ is the ratio of the exponential to the total volumes, and τ_m (1 - ƒ) the time required for water to flow through the piston flow section [207]. The exponential fraction parameter can be either estimated independently, via multiple samples at the same bore, or by assuming multiple bores share similar ƒ parameters. The standard approach in New Zealand, and the one used here, is to apply around 70% of the exponential age distribution within a total exponential piston flow volume. We also calculated an additional analysis with an assumed exponential fraction of 30% as a sensitivity analysis (available from the report's authors).

Thirdly, BASE compares the predicted concentration history to the observed concentration history. The model used to transform the capture zone concentration to the well concentration is:

$$C_{rt} = (1-d) \times \sum_{a=0}^{n} p_{t-a} \times C_{s(t-a)}$$

where C_{rt} is the predicted nitrate-N concentration at the well at time *t*, *d* is a time invariant dilution factor, p_{t-a} is the proportion of water at the well at time *t* which is *a* years old, and $C_{s(t-a)}$ is the nitrate-N concentration in the capture zone at time $t - a$. p_{t-a} is calculated from the age distribution of the water at the well, which is usually estimated by fitting a 1-d age model to a sampled tracer (e.g., tritium).

Fourthly, if the predicted concentration history is like the observed concentration history, then the capture zone concentration history is likely to be similar to the random sample and the random sample is retained, otherwise the random sample is discarded.

Steps 1-4 are repeated many times to build up a distribution of likely capture zone concentration histories. The BASE solver uses the MT-DREAM(ZS) algorithm [208] to efficiently constrain the prior distribution into the posterior. This process begins with very poor model results and then through iteration improves the model fit. Eventually the algorithm converges on a posterior distribution that is a good fit to the observed data.

An example BASE fit is given in **Figure 18** showing the predicted concentration in the capture zone (also called the source) and the well (also called the receptor).

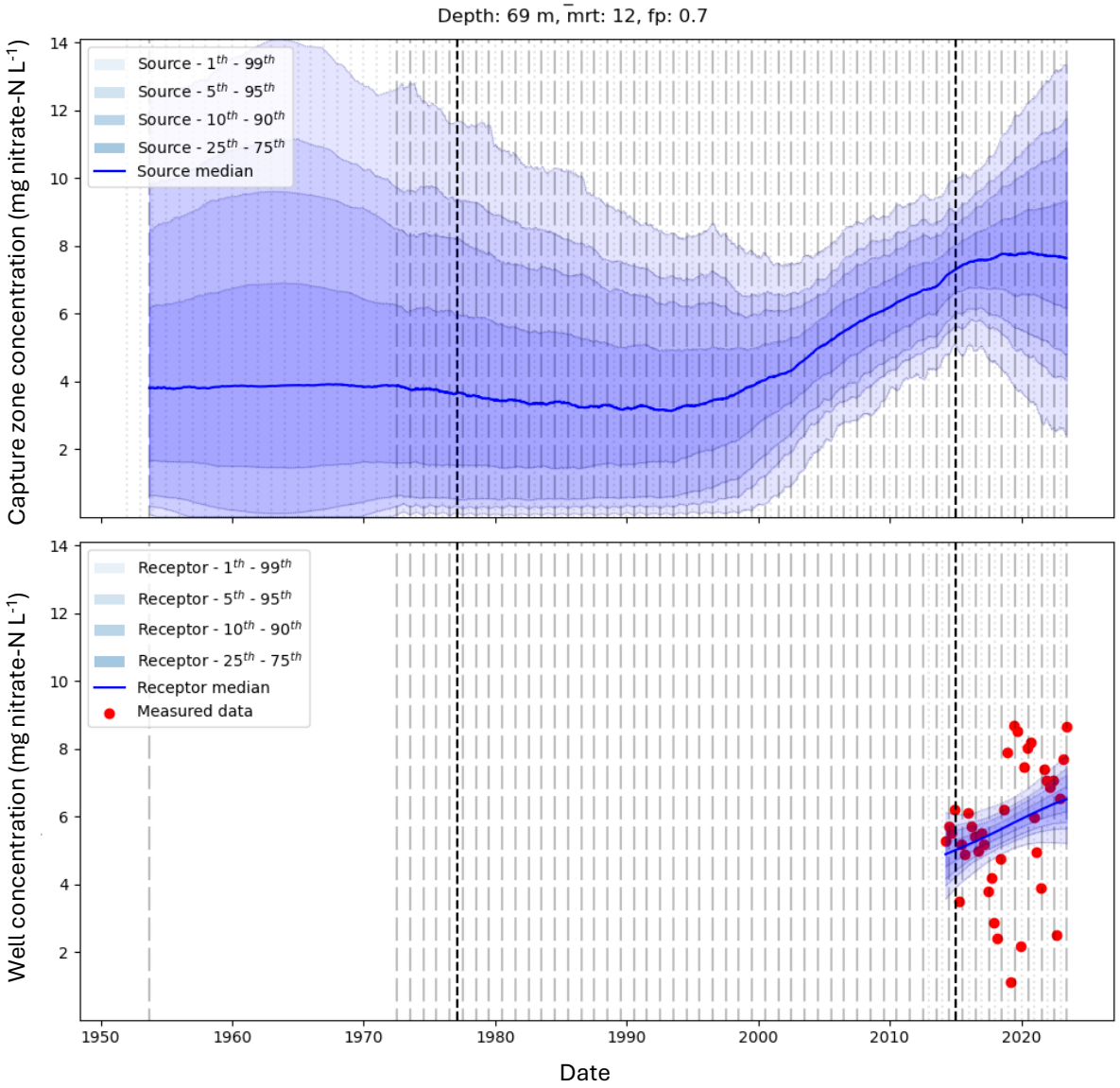


Figure 18. The predicted concentration and confidence intervals in the capture zone (also called the source; top) and the well (also called the receptor; bottom). The two dark vertical dashed lines represent the time over which 90% of historical source concentrations are informed by well concentrations.

This modelling assumes that there was no nitrate removal during the transport path (e.g., all groundwater is oxic and nitrate behaves as a conservative tracer) and that the age distribution of water in the well is static (e.g. does not change) over time. Given the small area of the capture zone and uniformity of aquifer materials in the scheme we are relatively confident of the first assumption. We have no data to prove that the age distribution was static. However, by augmenting existing age data with sampling conducted over a six-week period in February 2025 we obtain a snapshot of ages that are likely to be the product of similar conditions and more likely to have robust age data than earlier estimates owing to the decay of tritium from nuclear testing [209].

Methodology to account for lag times

Including lag times into power analyses to estimate the likelihood of detecting a reduction requires quantification of two temporal components in groundwater flow processes: 1) the delay in water from the capture zone reaching the well (lag); and 2) the mixing of waters of variable age along the flow path (temporal dispersion). To determine if a percentage reduction can be detected we conducted a power analysis which relies on estimating the standard deviation of the residual fits of BASE predictions of the observed well nitrate-N concentrations. Here statistical power is the probability that the percentage reduction is real. It is common to use a binary threshold for statistical power of ≥0.80 as the level at which the likelihood of an effect (one-sided test) is real [149], but this is influenced by the critical level (α), sample size, the desired level of percentage reduction and variability in the data. If the critical level is set at $P<0.05$, we can use the variability (viz. standard deviation) of nitrate-N concentrations to calculate either the percentage reduction detectable with a specified number of samples or the number of samples required to meet a specified percentage reduction. This helps those involved in implementing water quality policy avoid missing the opportunity to detect an effect and coming to the wrong conclusion, but it also helps forecast the time it may take to detect the change and the cost of detection since our main method of increasing power is to increase the number of samples taken [150].

Predictions for the well nitrate-N concentrations were resampled over 5, 20 or 50 years at monthly or quarterly frequencies. A synthetic sub-sample is also generated of a normal distribution of the standard deviation of residual fits and combined with the predictions at the appropriate frequency and duration of sampling. A Mann-Kendall test was then conducted to test for the significance of the slope and a Wilcoxon signed rank test used to determine if the concentration (at the percentage reduction) is less than expected if no action has taken place. This cycle sub-sampling and testing is repeated 1000 times with the probability of detecting the change as the number of times the synthetic concentration time series had a statistically robust trend divided by the number of iterations or prediction.

An example power analysis for a site is given in **Figure 19**. At this site, nitrate-N concentrations exceed 7 mg L^{-1} by 2030 if no action had taken place (bottom plot). However, the implementation of actions leads to concentrations that are lower than their starting point (i.e., decreasing slope) by 2034 with monthly sampling and 2040 with quarterly sampling (top plot). We also know that the difference to taking no action is likely to be real in 2032 with monthly sampling and 2034 with quarterly sampling (middle plot).

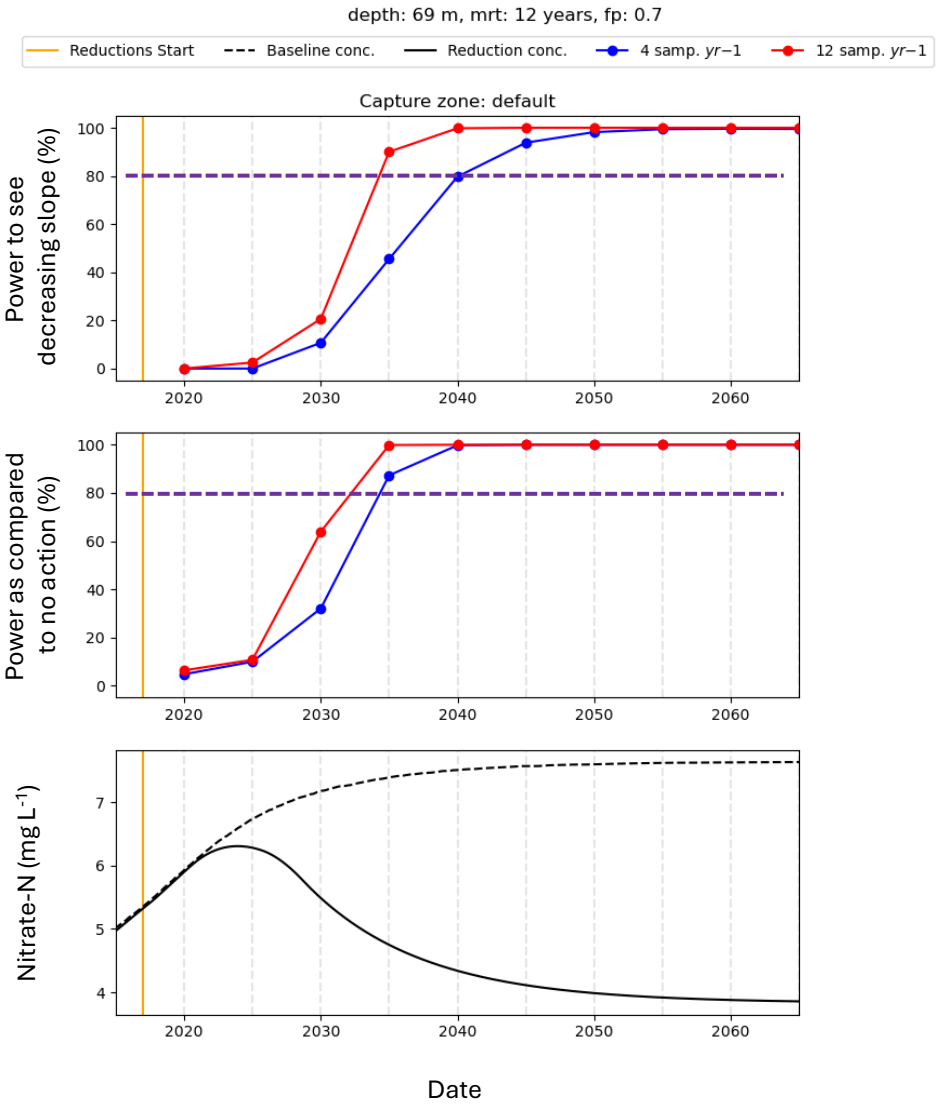


Figure 19. Statistical power with monthly or quarterly sampling to detect a decrease in nitrate-N concentration (top = 200 m initial width) and to concentrations compared to doing no action was taken (middle). The solid and dashed lines in the bottom plot represent nitrate-N concentrations if actions were implemented (solid line) or not implemented (dashed line). The purple dashed lines correspond to the level at which changes can be deemed real.

Another way of visualising the effect of reductions is to examine the predicted nitrate-N concentrations in the capture zone and well (**Figure 20**). Here the effect of a history of nitrate-N enrichment is evident in the increase in concentrations after the implementation of actions (**Figure 20**, top). While nitrate-N leaching from the capture zone is theoretically reduced quickly through the implementation of actions and stable thereafter, the mixing of waters of different age results in a gradual change in well concentrations (**Figure 20**, bottom). It is notable that the lag in the well and the relatively recent increases in leached nitrate-N concentration means that concentrations would increase significantly from the 2017 baseline period without any changes in anthropogenic actions. Therefore, the percentage reductions (e.g., 25-40%) are insufficient to decrease concentrations to 30% below the 2017 baseline year.

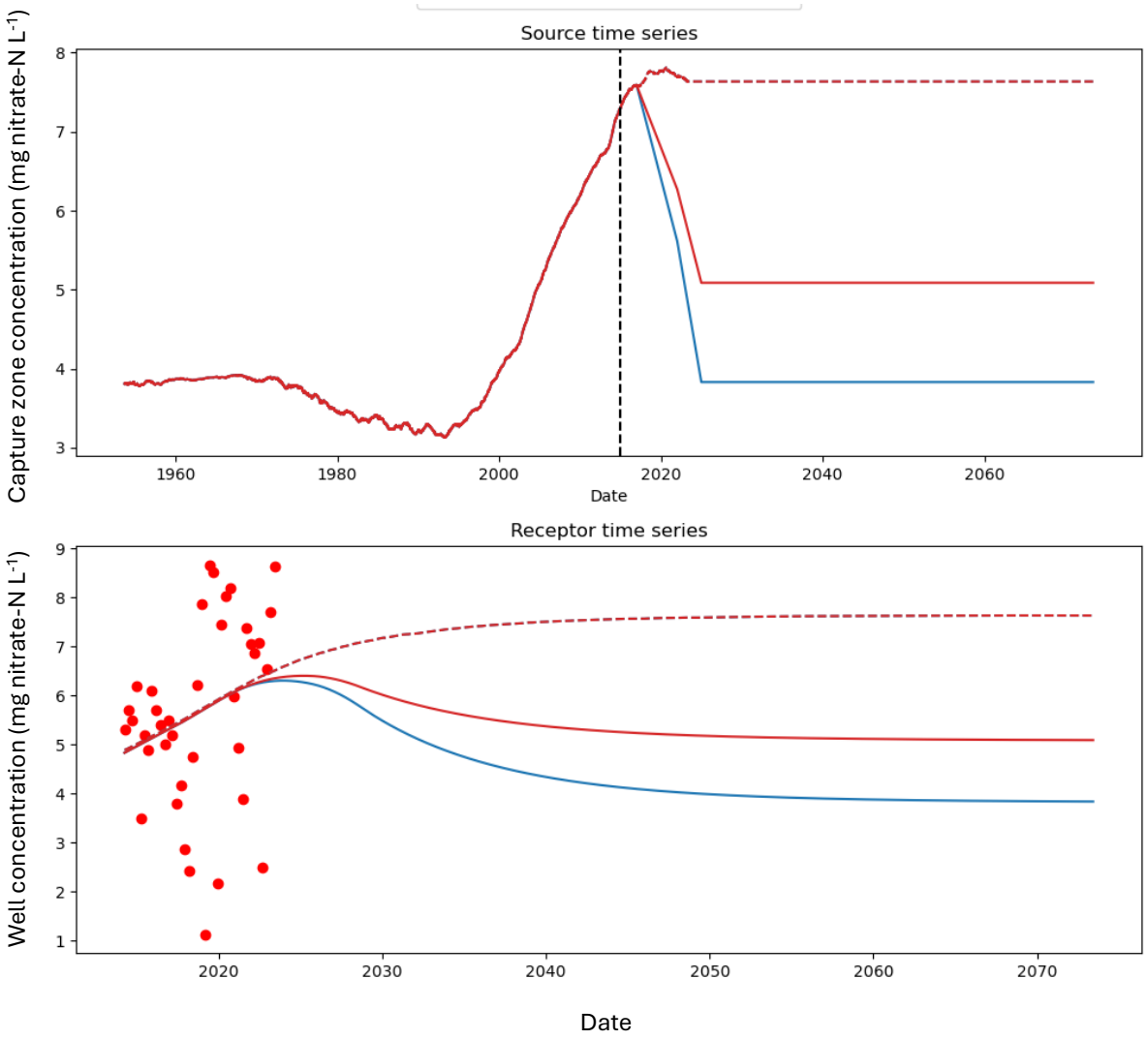


Figure 20. Nitrate-N concentrations in the capture zone (top) and well (bottom) assuming the percentage reductions for properties in the capture zone in 2022 and 2019 relative to the baseline year of 2017 (top) for the default (200 m initial width) and large (1 km initial width) capture zones. The black line indicates the limit for when past information can contribute to predictions because the lag time in delivery means the leachate from the capture zone has not yet reached the well. The dashed line represents no change in nitrate leaching while the solid red and blue lines represent two reduction scenarios, with actions implemented over a two year period.

Groundwater nitrate-N concentrations

Median groundwater nitrate-N concentrations varied from 3.2 in BX21/0018 to 15.7 mg L^{-1} in BX22/0067 (**Figure 21**). Previous analysis of the concentrations over time [210] separated sampling since 2014 into two groups 2014/15-2018/19 and 2019/20-2023/24. This separation was based on a visual inspection of the data and suggested that there was a break point in concentrations in some wells around 2019. A Mann Kendal test of the two groups showed a larger number of wells were increasing rather than decreasing or showing no trend in the first period. However, there were a larger number of wells showing decreasing trends than increasing or no trend in the second period (**Table 14**). On its own this could be interpreted as evidence for the effect of actions reducing N losses. However, variation could be caused by variation in the implementation of actions, flow paths

that don't connect the capture zone to the well, climatic variability, or a lag time bringing water (and nitrate-N) that is from land use prior to the start of the scheme. Indeed, increasing the length of record and rerunning the Mann Kendal test showed some wells (e.g., BX21/0017) still exhibit significant increasing trends while others do not (e.g., BX21/0018) (**Figure 23**). The BASE output for these wells is presented in **Figure 24** showing how variability in the data influenced the fit of the model and therefore the likelihood of detecting change. Furthermore, it is worth highlighting that there was very little land surface recharge (and thus minimal N-losses) into groundwaters in 2014-2016 period (**Figure 22**). Recharge increased again in 2017/2018. This change plus a 1-year lag time means that break points around 2019 are most likely a factor of climate in the Selwyn District. However, some wells do exhibit a continued decrease in nitrate-N concentrations that are out of sync with climatic change, inferring that decreases were caused by the implementation of actions.

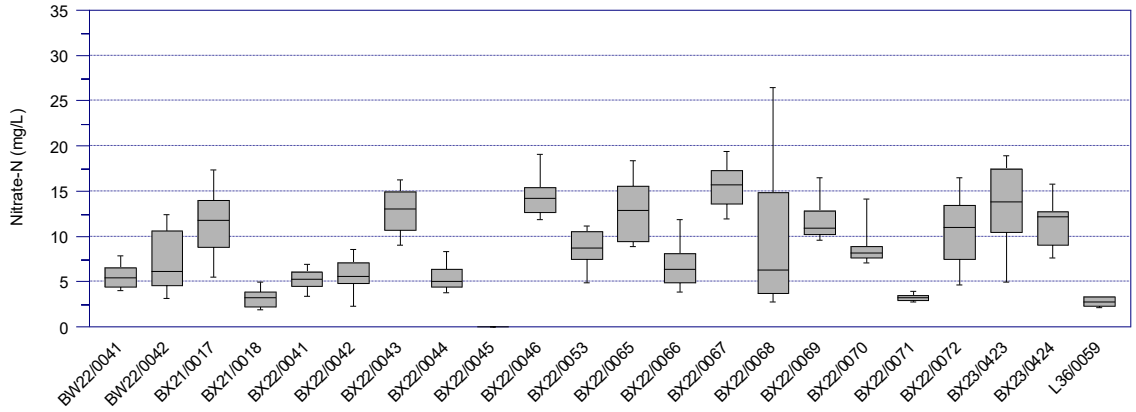


Figure 21. Box plots showing the 5th and 95th percentiles as whiskers either side of a box indicating the 25th, 50th and 75th percentile concentrations of nitrate-N in each groundwater well.

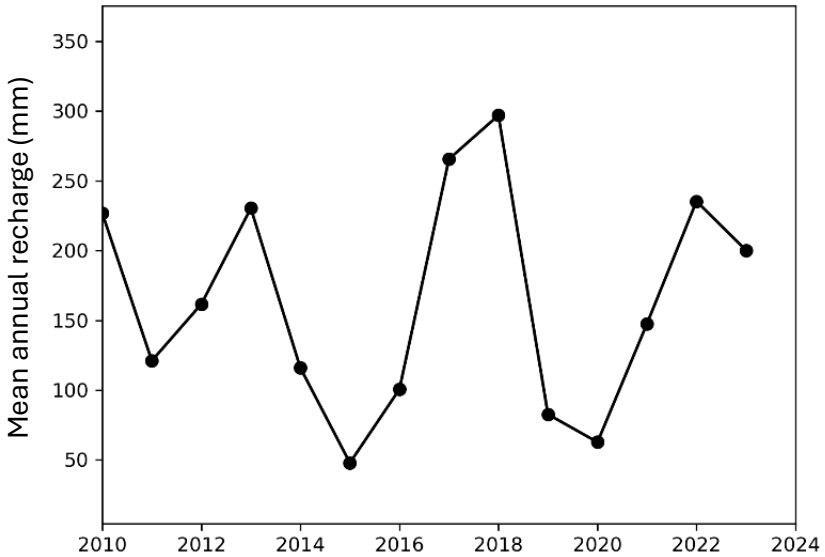


Figure 22. Mean annual recharge across the Selwyn District.

Table 14. Count of bores monitored by CPWL with no trend or significant (P<0.01; likelihood ≥ 0.835) increasing or decreasing trends in groundwater nitrate-N concentrations. Values in parentheses are the mean of the median concentrations (mg L^{-1}) in bores for each trend direction and period.

Trend direction	2014/15 - 2023/24	2014/15 - 2018/19	2019/20 - 2023/24
Increasing	11 (10.8)	9 (7.5)	2 (14.5)
Decreasing	4 (5.6)	4 (7.2)	11 (8.1)
No trend	5 (8.0)	7 (9.3)	7 (9.0)

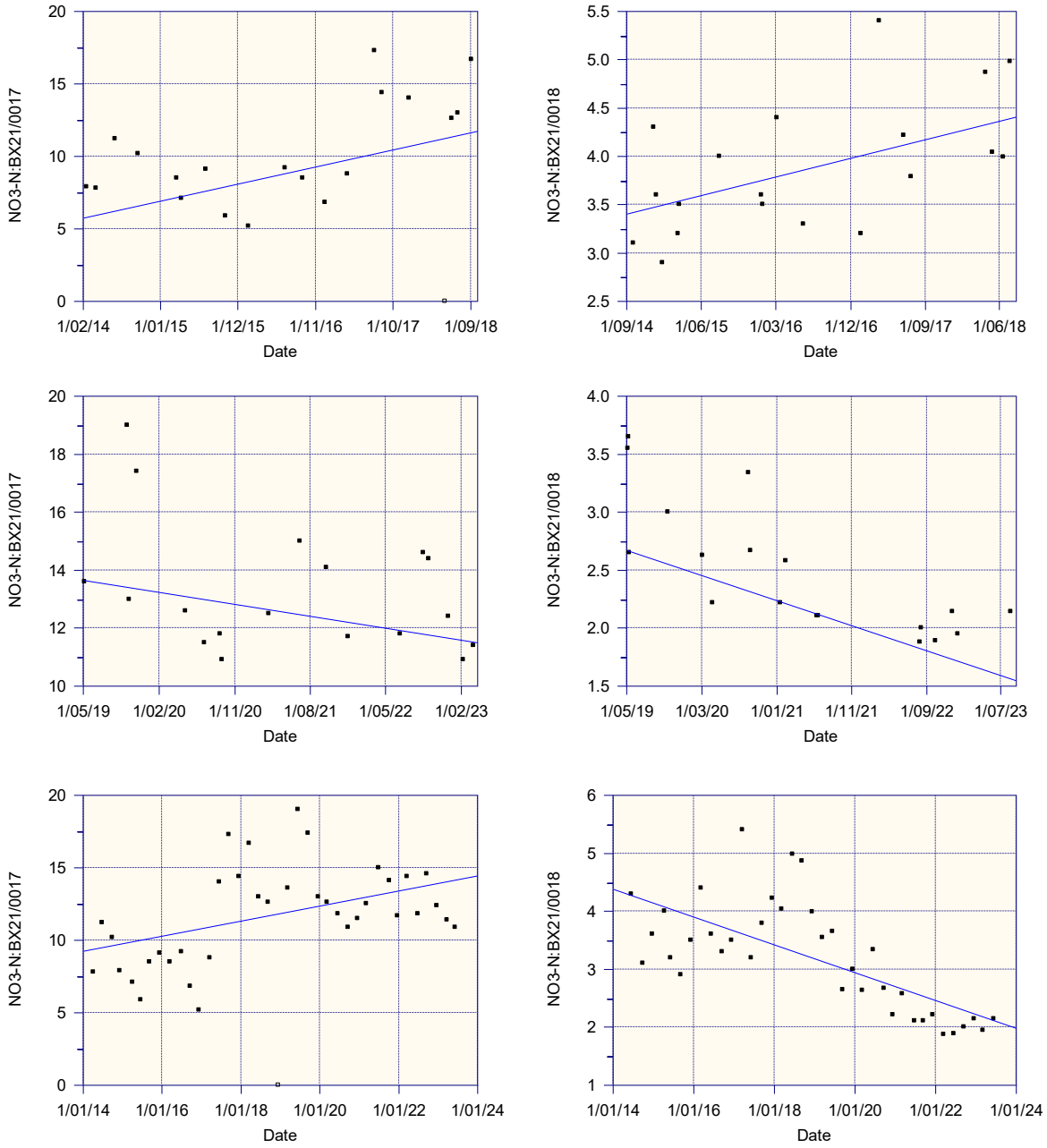


Figure 23. Plots show the effect of selecting different period for trend analysis for nitrate-N concentrations (mg L^{-1}) in two groundwater bores showing an increasing trend from 2014/15 to 2018/19 (top) and a decreasing trend from 2018/19 to 2023-24 (middle), but both increasing and decreasing trends when all data are used. All trends were significant (P<0.01; likelihood ≥ 0.835)

Differences are likely caused by a combination of lag times between nitrate-N being lost from topsoil and being detected in the bore and climate.

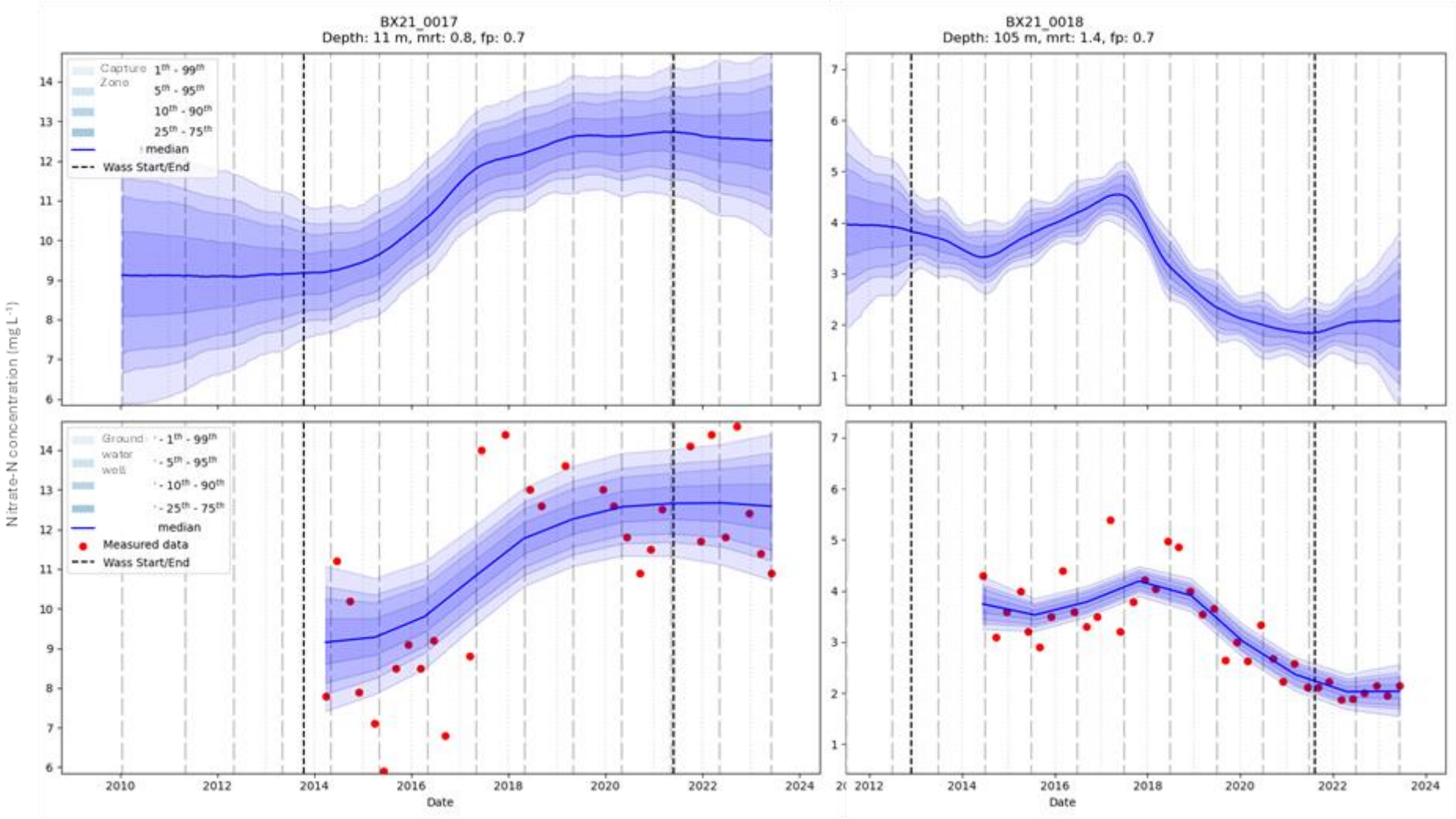


Figure 24. Nitrate-N concentrations (mg L^{-1}) in the capture zone and groundwater of wells showing a high (left) a low (right) magnitude of variation and short residence time leading to a long (never) and short (2025) detection time, respectively.

Accounting for lag times when determining the time to detect change

As mentioned above we assume that implementation has occurred (via audits), and that flow paths connect the capture zone to the well. The influence of increased production on contaminant losses is accounted for by regular updating of farm scale losses. This leaves climate and lag times as dominant factors influencing changing nitrate concentrations. We cannot remove the influence of climate on trends for all sites as too few data are available to represent seasonal impacts (and too little time has yet passed). However, we can remove the impact of lag times on trends which leaves only climate and the effect of actions as the cause for changes in concentrations. Over time, and especially where lag times are short (and change is occurring rapidly), sufficient data may be available to then remove the effect of climate, leaving actions as the sole cause for change. In the meantime, the best way to attribute change is to provide guidance around when changes would be likely detected. Because our estimates contain all variation (middle plot **Figure 13**) they are conservative i.e., likely to indicate more samples (and time to detect change) than if climate had been removed. However, if change has not occurred by our conservative estimates, it is likely that one of the other components of variation is the cause. In other words, if no change is detected but large reductions in nitrate-N losses were predicted it is possible that actions were not implemented effectively, conversely if change happens well before thought detectable, this is evidence that actions have been effective.

Mean residence times in the wells varied from less than a year to 80 years with a median age of 5.8 years (**Table 15**). There were several wells that sampled moderately deep groundwater that was young. These are factors likely to hasten the likelihood of detecting changes in nitrate-N concentrations. Indeed, the mean time to detect change (as a decreasing slope) from the percentage reductions varied from 2034 under quarterly sampling to 2030-2032 (depending on the recharge rate) with monthly sampling. This suggests change would be evident within the current consent period granted by the Canterbury Regional Council (to 2047). However, the years when nitrate-N concentrations from actions would be less than if no action has occurred would be detected on average in 2041-2045 (depending on recharge rate) with quarterly sampling, improving to 2035-2040 with monthly sampling. Spatially, the likelihood of detecting change increased towards the coast and away from the Waimakariri River or Christchurch (**Figure 25**).

There were five instances where variation indicated that improvements would never be detected with monthly sampling which increased to 11 instances under quarterly sampling. This suggests that an increase in sampling frequency improves the likelihood of detecting improvements. In the five cases where improvements could not be detected, two were >11.3 mg nitrate-N L^{-1} a threshold used for drinking water. Additional actions may need to be implemented in these capture zones to reduce concentrations, although additional work should be done to confirm that capture zones and wells are linked especially for shallow wells with short mean residence time. There were four wells where changes should be evident from 2025-2030. In each well, nitrate concentrations have reduced. This is good evidence of change and would be the first wells to further examine and remove any climate effects from to attribute to the implementation of actions.

Table 15. Depth (m) and mean residence time (years) of groundwater in wells sampled by Central Plains Water limited, along with the median nitrate-N concentrations for 2014-now and from 2022-now periods, and the likely dates for quarterly or monthly sampling to detect the percentage reductions in nitrate-N losses from farm environment plans in receiving wells. The two 'year' values refer to solutions using 30 and 70% of recharge from the capture zone. Wells in bold show good evidence of change detection (decreasing concentration).

Well	Depth (m)	Mean reduction in capture zone (%)	Median nitrate-N (mg L^{-1})	Median nitrate-N >2022 (mg L^{-1})	Mean residence time (years)	Year when decreasing slope from actions detected (quarterly sampling)	Year when decreasing slope from actions detected (monthly sampling)	Year when action would be less than no action (quarterly sampling)	Year when action would be less than no action (monthly sampling)
bw22_0041	10	3.1	5.4	5.5	0.2	Never - Never	Never - Never	2020 - 2025	2020 - 2020
bw22_0042	25	16.3	6.1	6.3	0.3	Never - Never	Never - Never	Never - Never	Never - 2075
bx21_0017	11	23.7	11.8	12.1	1.8	Never - Never	2030 - Never	2030 - 2025	2025 - 2025
bx21_0018	105	25.3	3.2	2.0	1.5	2025 - 2025	2025 - 2025	Never - Never	Never - Never
bx22_0041	40	33.4	5.2	5.8	3.8	2030 - 2030	2025 - 2025	2030 - 2030	2025 - 2025
bx22_0042	69	27.4	5.6	7.0	9.3	Never - Never	2035 - Never	2040 - 2040	2035 - 2035
bx22_0043	70	32.4	13.1	15.4	18.7	2040 - 2055	2040 - 2045	2045 - 2040	2040 - 2040
bx22_0044	9	35.9	5.1	4.1	1.8	2025 - 2025	2025 - 2025	2035 - 2040	2030 - 2030
bx22_0046	30	28.3	14.2	14.7	3.2	2025 - 2030	2025 - 2025	2030 - 2040	2030 - 2030
bx22_0053	50	45.6	8.7	5.8	10.5	2030 - 2035	2030 - 2030	2040 - 2040	2035 - 2035
bx22_0065	40	15.8	12.9	11.2	5.8	Never - Never	2030 - 2035	2060 - 2035	2045 - 2030
bx22_0066	45	18.7	6.4	6.5	5.5	Never - Never	2035 - 2035	2045 - 2055	2035 - 2035
bx22_0067	45	26.5	15.7	15.8	9.1	2030 - 2040	2030 - 2035	2035 - 2035	2035 - 2030
bx22_0068	85	10.5	6.3	5.3	79.9	Never - Never	Never - Never	Never - Never	Never - Never
bx22_0069	118	14.9	10.9	11.3	21.4	2055 - Never	2045 - Never	2050 - 2055	2045 - 2050
bx22_0070	101	10.0	8.2	8.2	12.0	2045 - 2040	2040 - 2035	2055 - 2040	2055 - 2035
bx22_0071	79	11.4	3.2	2.9	0.4	Never - Never	Never - 2025	Never - Never	2025 - 2025
bx22_0072	35	11.4	11.0	11.3	1.5	Never - Never	Never - Never	Never - 2040	2050 - 2030
bx23_0423	47	6.1	13.9	15.3	7.3	Never - Never	Never - Never	2045 - 2075	2055 - 2050
bx23_0424	60	4.0	12.2	12.1	17.2	Never - Never	2045 - Never	2055 - 2050	2045 - 2065
l36_0059	47	18.5	2.8	-[a]	41.4	Never - 2025	Never - 2025	2075 - 2025	2070 - 2025

[a] No longer sampled.

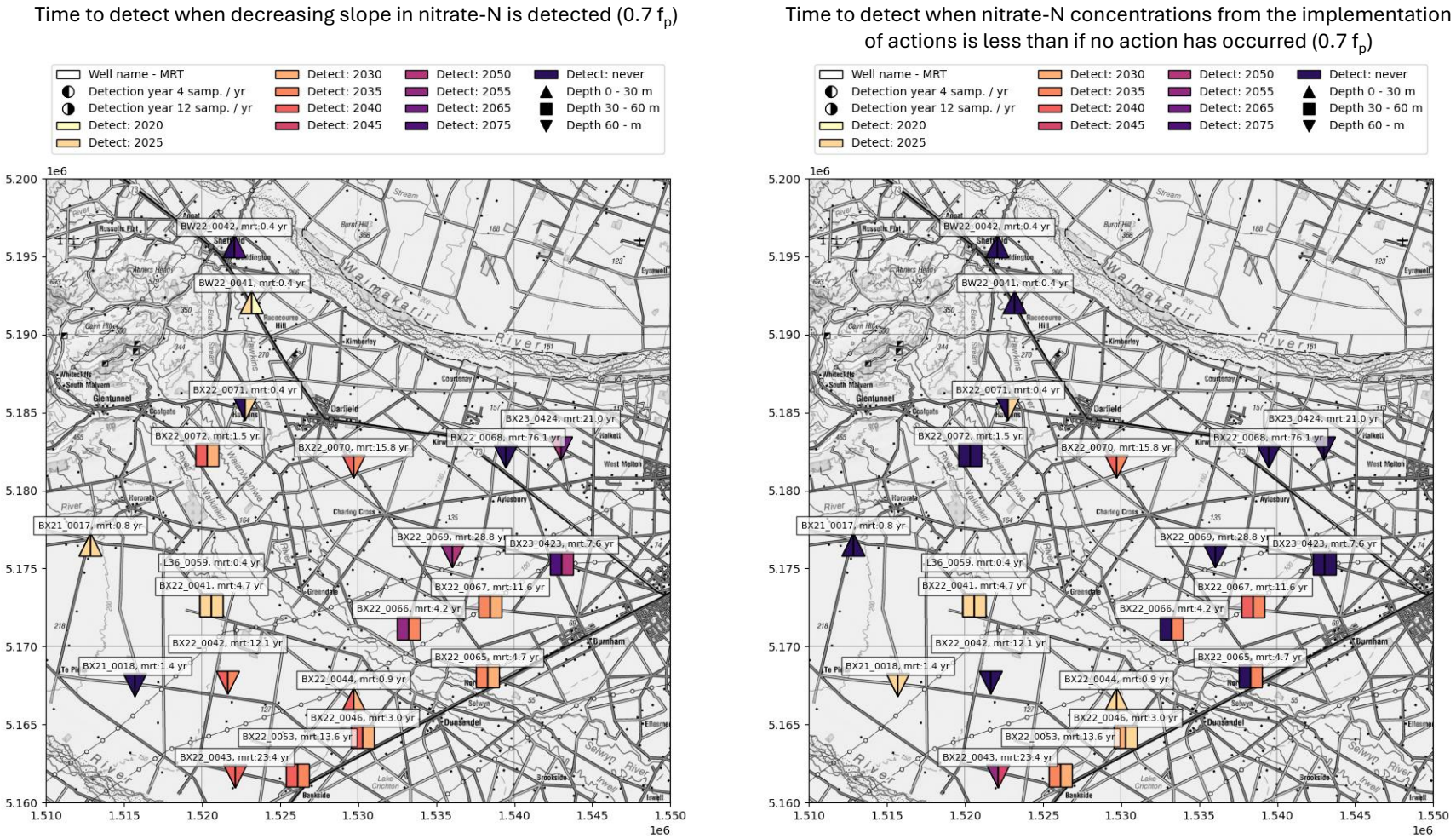


Figure 25. Maps showing the time to detect when a decreasing slope nitrate-N concentrations (from action; left) would be detected or when nitrate-N concentrations after the implementation of actions to reduce nitrate-N concentrations would be less than if no action had occurred (right).

Case Study (surface water): Determining the likelihood of on-farm actions as part of good farming practice in the dairy and sheep and beef sectors to reduce nitrogen and phosphorus concentrations in freshwater management units

Introduction

This case study assumes that **Step 1** (Determine the target) sees the dairy and sheep and beef sectors aim for percentage reductions possible with the implementation of good farming practices and **Step 2** (Select farm actions) and **Step 3** (Plan and implement actions) have been done through farm environment plans endorsed by each industry sector. However, to make this scenario more realistic, we assess the rate of implementation with new industry data on the real uptake of actions. The result is an array of dairy and sheep and beef farm typologies, representative of large areas, with actions implemented and the percentage reduction in N and P recorded compared to a baseline (with a minimum level of action). Each farm type, being representative of a combination of slope, drainage, and management classes, is distributed across freshwater management units (FMUs; delineated as of March 2025 - **Figure 26**). In this case study we calculate the likelihood of detecting the percentage reduction in the area-weighted proportion of land in dairy and sheep and beef farm typologies within a freshwater management unit. We assume the following:

1. Implementation of actions occurs within 1-2 years, resulting in an abrupt change in N and P losses from farms.
2. We do not account for riverine (e.g., attenuation or release from bed sediments) or groundwater processing.
3. All surface water discharged from farms makes it to the outlet of the freshwater management unit and is not lost to groundwater.
4. Lag times between being lost from the farm and detected at the catchment outlet are < 5 years [163] and hence discounted from our assessment.

This cases study focuses on **Step 4** (Maximising the likelihood of detecting change). It does not assess **Step 5** and attribute changes to the implementation of actions, because insufficient data are available to robustly remove the effects of climate or changing production in many freshwater management units. Since it can take many years (even decades) to collect sufficient data to maximise the likelihood of detecting change [155], the time spent collecting data may coincide with having sufficient data to also remove the influence of climatic factors. Having long-term surface water quality sites and information on land use (and production) is therefore crucial in attributing the implementation of actions as the most likely case of water quality improvement.



Figure 26. Map of freshwater management units (red) within each Regional Council (black) assessed.

Step 3 (Plan and implement actions): Updating dairy farm losses with realistic implementation rates for actions

As outlined in Macintosh et al. [211] the original dairy farm typologies (**Figure 27**) were updated using the 2022-2023 data held in DairyBase [212]. Briefly, the farm typologies represent classes of dairy land biophysical characteristics based on climate, slope, drainage, and wetness [123]. Each typology has an estimate of N and P loss calculated using the latest version of the Overseer® Nutrient Budgeting software [213]. Nitrogen losses (kg N ha^{-1} yr^{-1}) modelled by Overseer account for N run-off and leaching below the root zone from both livestock urine patches and from other non-urine sources such as fertiliser, or soil mineralisation. Phosphorus losses (kg P ha^{-1} yr^{-1}) modelled by Overseer relate to both surface runoff (main loss process) and leaching below the root zone. Nutrient losses estimated by Overseer for farm typologies was seen as an appropriate use of the model for policy purposes [144].

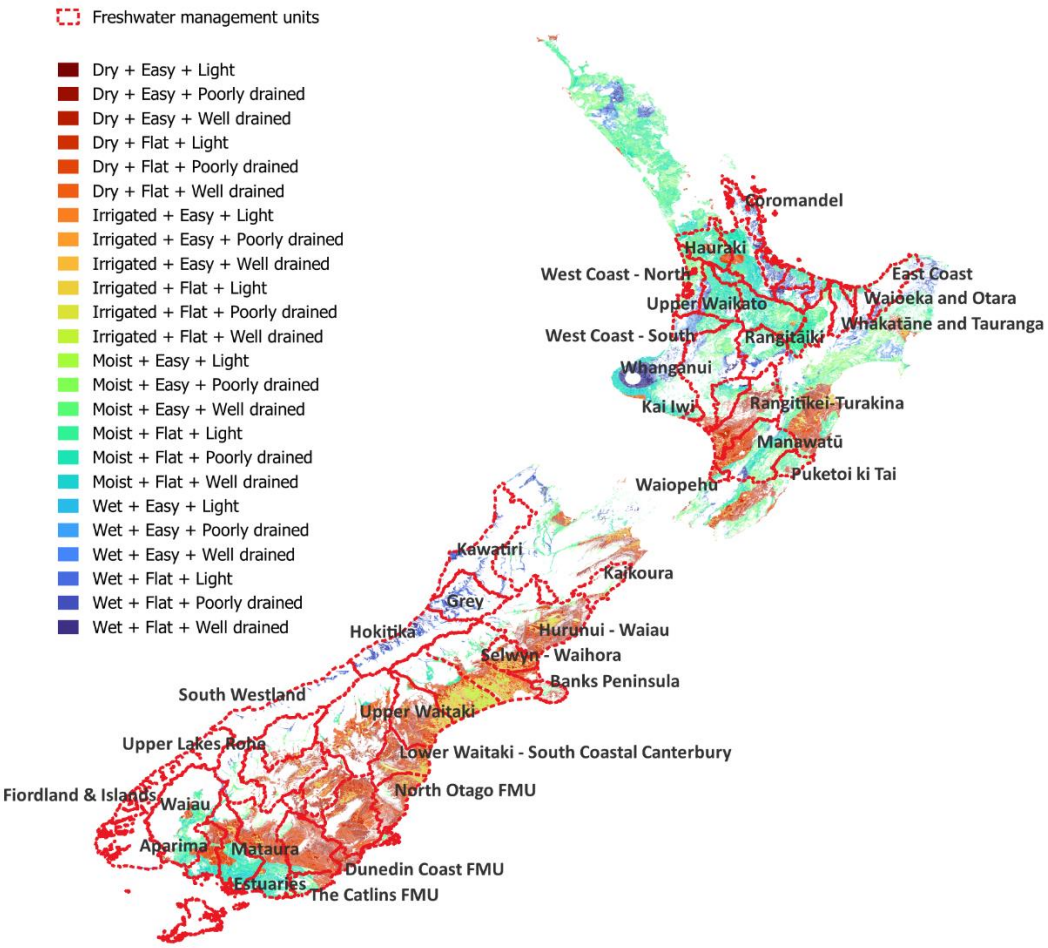


Figure 27. Map showing the Sheep and Beef farm types and freshwater management units.

We assumed that mitigation actions were applied to each farm type following the advice in Step 2. The full effects of mitigation actions on N and P losses were estimated via Overseer assuming 100% implementation of actions and termed full effect. Full implementation was then adjusted by using data for the real proportion of farms implementing actions in 2023. To do this Macintosh et al. [211] extracted data supplied by Fonterra Ltd for the implementation of Good Farming Practices (i.e., 'actions') as part of their Tiaki farm environment plans as of 2023. To anonymise data but keep it spatially representative of an area, a total of 14,214 polygons were generated each representing a unique combination of region, district, catchment (aligned to FMUs), and dairy type. For each polygon, data on the implementation status of actions were aligned (**Table 16**). Note that these actions are a subset of the complete list of good farming practices advocated by DairyNZ [126]. The actions were categorised into those that were:

1. Complete (have been implemented).
2. Planned and in progress.
3. Not applicable to the farm (e.g. improved irrigation practice are not applicable to a non-irrigated farm or stream fencing is not applicable to a farm with no waterways).

The calculated spatial rates of action implementation (as of 2023) were calculated using the ratio of land area with actions complete, then adjusted by the percentage of actions already implemented (assumed to be 20% [202]) in FMUs.

This analysis indicated a good level of national implementation of actions (64%; **Table 16**; **Table 17**). If actions were fully implemented N and P losses to water in FMUs would reduce by 35 and 24%, respectively. However, after accounting for the real rate of implementation, reduction decreased to 22% and 15%, respectively. These data represent our best estimate at a national FMU scale of the magnitude of decreases possible. In Step 4 we also show the likelihood of detecting these reductions. Note that better data are available at catchment studies in some regions.

Table 16: List of actions (good farming practices; https://www.dairynz.co.nz/resources?spotlight=7246) aligned to dairy farm typologies and Master **Table 2** to improve water quality [126] and the national spatial implementation status 'as of 2023'.

Good farming practice area	Number in Master Table 2	Short description	National area implemented as of 2023 (%)
Optimum soil test P	24	Soil phosphorus levels are monitored and maintained below or within the target ranges for the soil-type and crop.	78
Nutrients	22	The amount and timing of fertiliser inputs, takes account of all sources of nitrogen and phosphorus, matches plant requirements, and minimises losses to waterways and groundwater.	78
Nutrients	21	Fertiliser is stored and loaded to minimise the risk of spillage and losses to waterways and groundwater.	66
Nutrients	21	Fertiliser spreading equipment is maintained and calibrated.	83
Nutrients	-	Feed is stored, transported and fed to minimise wastage, leachate and soil damage.	77
Critical Source Areas	34	Critical Source Areas and farm Hot Spots are identified and managed to minimise contaminant losses to waterways.	69
Waterways	13	Stock are excluded from lakes and waterways.	63
Land & Soil	43	Cultivation is managed to reduce the risk of sediment loss and maintain soil structure.	70
Land & Soil	46	Erosion-prone land is managed or retired to minimise soil losses.	38
Land & Soil	-	Grazing of pastures and crops is managed to minimise sediment and contaminant loss.	78
Land & Soil	34	Paddocks selected for Intensive Winter Grazing (including intensive baleage wintering) are low risk and managed to minimise the risk of erosion, run-off to waterways and leaching to groundwater.	83
Effluent	18	Effluent and manure are applied at depths, rates and amounts that match plant requirements and minimise the loss of nutrients or microbial pathogens to waterways and groundwater.	67
Effluent	-	The effluent system is designed, operated and regularly checked to minimise the risk of nutrient and microbial pathogen loss to waterways and groundwater and to prevent microbial contamination.	47
Water Use & Irrigation	26	The depth, rate and timing of irrigation is managed to meet plant demand and minimise the risk of leaching and run-off.	35

Table 17: National and regional N and P loss percentage reductions associated with the implementation of actions in **Table 16** for dairy farms supplying Fonterra (assumed to be representative of all dairy farms) if fully implemented and 'as of 2023' (second value) or if adjusted for real rates of implementation (first value).

Region	Freshwater Management Unit (FMU)	FMU (ha)	Dairy area (ha)	N loss reduction (%)[3]	P loss reduction (%)[3]
Bay of Plenty	Kaituna, Maketu and Pongakawa	107357	21989	18, 30	19, 31
Bay of Plenty	Rotorua Lakes	109990	7098	16, 25	12, 20
Bay of Plenty	Tarawera	79606	9826	21, 40	17, 33
Bay of Plenty	Tauranga Harbour	130583	8728	17, 27	15, 23
Bay of Plenty	Waioeka and Otara	123112	4868	19, 35	17, 32
Bay of Plenty	Whakatane and Tauranga	179383	11265	19, 35	17, 34
Canterbury	Ashburton	618816	98469	29, 52	13, 24
Canterbury	Banks Peninsula	91177	448	8, 13	0, 0
Canterbury	Christchurch - West Melton	48302	88	0, 68	0, 40
Canterbury	Hurunui - Waiau	809495	47637	34, 50	18, 25
Canterbury	Kaikoura	260634	5254	25, 39	14, 22
Canterbury	Orari-Temuka-Opihi-Pareora	504630	42121	28, 49	13, 25
Canterbury	Upper Waitaki	965576	16848	9, 33	4, 13
Manawatu-Whanganui	Kai Iwi	33952	1807	20, 30	18, 28
Manawatu-Whanganui	Manawatu	591251	87461	18, 30	11, 18
Manawatu-Whanganui	Puketoi ki Tai	122152	0	NA, NA	NA, NA
Manawatu-Whanganui	Rangitaikei-Turakina	521844	21390	27, 38	9, 12
Manawatu-Whanganui	Waiopehu	39518	4803	24, 44	9, 16
Manawatu-Whanganui	Whangaehu	199459	5409	26, 36	10, 15
Manawatu-Whanganui	Whanganui	727070	7273	15, 29	9, 20
Otago	Dunedin Coast	111408	6610	6, 14	7, 15
Otago	Manuherikia Rohe	303295	5398	12, 39	6, 16
Otago	North Otago	298312	34237	15, 54	9, 30
Otago	Taieri	570754	14226	11, 39	6, 18
Southland	Aparima	209131	44596	10, 19	7, 14
Southland	Mataura	648230	69495	13, 27	7, 14
Southland	Oreti	408802	67865	12, 25	6, 14
Southland	Waiau	858261	10006	7, 18	5, 12
Waikato	Coromandel	195723	6171	18, 30	18, 28
Waikato	Hauraki	394510	131454	18, 32	19, 32
Waikato	Lake Taupo	349596	2908	10, 15	9, 14
Waikato	Lower Waikato	291172	62426	18, 34	18, 33
Waikato	Middle Waikato	63625	15624	18, 34	18, 34
Waikato	Upper Waikato	432778	125943	17, 27	17, 27
Waikato	West Coast - North	223883	4898	20, 30	21, 31
Waikato	West Coast - South	201048	10313	8, 28	10, 28
West Coast	Hokitika	553774	0	NA, NA	NA, NA
Mean – all FMUs				15, 24	22, 35

Step 3 (Plan and implement actions): Updating sheep and beef losses with realistic implementation rates for actions

We updated (**Figure 28**) a baseline set of sheep and beef farm typologies (i.e. without the effect of mitigation actions) conducted in 2024 (using data from 2020) [125]. The 2024 typologies were themselves an update of the original typologies created in 2020 [123]. The 2024 set of typologies split N and P losses from 251 Overseer outputs into farm types classified by rainfall (dry, moist, wet, and very wet, corresponding to <700, 700-1200, 1201-1700, and >1700 mm annual rainfall, respectively), topography (low, medium, and high, corresponding to <7, 7-14, and >14^{o} slope, respectively), and soil drainage (light, poorly-drained and well-drained, corresponding to plant available water holding capacity to 60 cm soil depth of 85 mm, imperfect or, poor or very poor soil drainage classes [214], and soil classed as well or moderately well drained [214], respectively). Our update used Overseer estimates of N and P loss from Beef and Lamb New Zealand for 553 farms collected between 2020 and 2022.

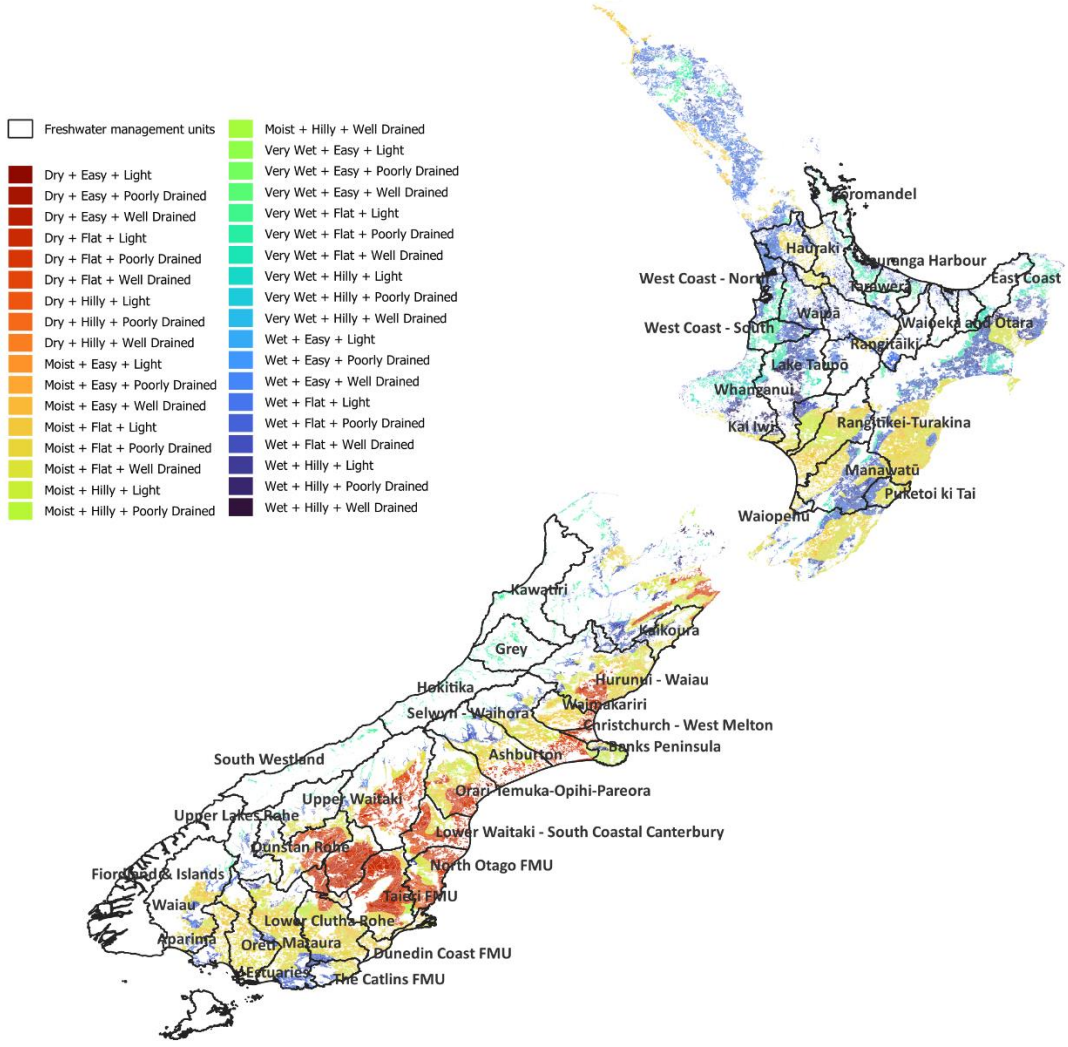


Figure 28. Map showing the Sheep and Beef farm types and freshwater management units.

In addition to Overseer data, the same farms contained data on the implementation of actions in farm environment plans. These data were collected for each farm during interviews of farmers by rural professionals. The interviews assessed the suitability and implantation of actions contained in the Beef and Lamb Farm Plan Environment Module [127] to reduce N and P losses. From the full range of actions within the Environment Module (n = 41) we modelled the effect of actions on the updated baseline types from those actions proven to improve freshwater quality and present in the original analysis of the effect of actions on N and P losses in 2020 [124]. We implemented all actions simultaneously. However, because actions were grouped into well-defined areas: winter forage crops (catch crops, strategic grazing towards the toe of the slope, and on-off grazing), in paddock (plantain, planting to prevent erosion), or edge of field (wetlands, sediment traps, stock exclusion and riparian buffers) [215], we chose to apply the effect of actions using the average of actions in each of these areas. This avoided the effect of actions being applied multiple times.

The interviews asked farmers for their state of readiness for the implementation of 41 actions contained in the Beef and Lamb Farm Plan Environment Module [127]. Readiness was classified as:

- Unsuitable for their farm
- Planned but not started
- Planned and being implemented
- Completed

We used data for planned and being implemented and completed as evidence that the actions are likely to complete by 2030 – the same deadline as used in the original analysis of the effect of actions on N and P losses by farm typologies [124].

Farm typologies were joined with the mean level of implementation (**Table 18**) after being adjusted for existing implementation (again assumed to be 20% [202]) to derive real reductions of N and P losses by region (**Table 19**). This analysis indicated a good level of the national implementation of actions (64%; **Table 18**; **Table 19**). If actions were fully implemented, N and P losses to water in FMUs would reduce by 23 and 54%, respectively. However, after accounting for the real rate of implementation, reduction decreased to 16% and 44%, respectively. These data represent our best estimate at a national FMU scale of the magnitude of decreases possible and realistic. In Step 4 we also show the likelihood of detecting these reductions. Note that better data are available at catchment studies in some regions.

Table 18. List or actions aligned to sheep and beef farm typologies and Master **Table 2** to improve water quality [126] and the national spatial implementation status as of 2020-2022.

Action	Number in Master Table 2	National level of implementation as of 2023 (%)
Winter forage crop management to prevent soil damage and contaminant loss	34, 45	87
Wetlands	6, 7	10*
Sediment traps	8	18

Low water-soluble P fertiliser	23	21
Fencing and riparian planting of altered channels and margins	11, 13	65
Planting to stop erosion, and fencing off gullies	13, 46	35

* This does not account for farms where wetlands were determined to be unsuitable (no data).

Table 19: National and regional N and P loss percentage reductions associated with the implementation of actions in **Table 18** for sheep and beef farms if fully implemented 'as of 2023' (second value) or if adjusted for real rates of implementation (first value).

Region	Freshwater Management Unit (FMU)	FMU (ha)	Sheep and beef area (ha)	N loss reduction (%)[3]	P loss reduction (%)[3]
Bay of Plenty	Kaituna, Maketu and Pongakawa	107357	26423	6, 19	23, 29
Bay of Plenty	Rotorua Lakes	109990	24465	9, 21	25, 34
Bay of Plenty	Tarawera	79606	15279	11, 10	18, 27
Bay of Plenty	Tauranga Harbour	130583	33850	7, 23	31, 42
Bay of Plenty	Waioeka and Otara	123112	13036	0, 20	28, 38
Bay of Plenty	Whakatane and Tauranga	179383	9640	4, 18	24, 33
Canterbury	Ashburton	618816	165820	28, 18	42, 52
Canterbury	Banks Peninsula	91177	56169	9, 26	42, 55
Canterbury	Christchurch - West Melton	48302	17497	20, 30	47, 67
Canterbury	Hurunui - Waiau	809495	329861	17, 12	34, 45
Canterbury	Kaikoura	260634	69480	16, 7	24, 37
Canterbury	Orari-Temuka-Opihi-Pareora	504630	237213	11, 20	43, 55
Canterbury	Upper Waitaki	965576	227926	17, 23	30, 39
Manawatu-Whanganui	Kai Iwi	33952	16845	8, 22	36, 47
Manawatu-Whanganui	Manawatu	591251	329051	10, 19	32, 42
Manawatu-Whanganui	Puketoi ki Tai	122152	87480	6, 21	33, 43
Manawatu-Whanganui	Rangitaikei-Turakina	521844	281540	14, 13	38, 46
Manawatu-Whanganui	Waiopehu	39518	10434	22, 15	32, 40
Manawatu-Whanganui	Whangaehu	199459	113670	9, 19	31, 41
Manawatu-Whanganui	Whanganui	727070	243245	6, 18	24, 37
Otago	Dunedin Coast	111408	42818	18, 14	46, 52
Otago	Manuherikia Rohe	303295	181161	19, 24	46, 60
Otago	North Otago	298312	179409	9, 23	42, 56
Otago	Taieri	570754	336690	16, 26	45, 59
Southland	Aparima	209131	75656	14, 20	49, 59
Southland	Mataura	648230	329606	8, 22	42, 52
Southland	Oreti	408802	162812	7, 24	50, 62
Southland	Waiau	858261	157611	20, 17	39, 49
Waikato	Coromandel	195723	29935	5, 12	17, 24
Waikato	Hauraki	394510	78908	4, 27	39, 51
Waikato	Lake Taupo	349596	44779	18, 14	28, 39
Waikato	Lower Waikato	291172	120833	9, 22	41, 54
Waikato	Middle Waikato	63625	25706	22, 28	57, 65
Waikato	Upper Waikato	432778	84346	14, 18	30, 43

Waikato	West Coast - North	223883	119024	7, 13	17, 25
Waikato	West Coast - South	201048	104791	4, 17	16, 25
West Coast	Hokitika	553774	18448	5, 6	23, 30
Mean – all FMUs				16, 23	44, 54

Step 4 (Maximising the likelihood of detecting change)

We calculated the area-proportion reductions in N and P losses by combining reductions from both dairy and sheep and beef typologies and after adjusting them by the area of land in forestry, native bush and arable or horticultural land within each FMU (**Table 20**). The adjustment always resulted in a decrease in reductions owing to the limited range of actions available to, and low magnitude of losses from, forestry and native bush [216] and the small land area occupied by arable and horticultural land [217]. Knowing the percentage reductions for fully implemented actions and the real rate of implementation we used the method of McDowell et al. [155] to calculate the likelihood of detecting each magnitude percentage change in FMUs. This calculation assumed that water quality sampling remained monthly and output the time (in years) it would take for the number of samples to reach an 80% likelihood (viz. power) of detecting the percentage N and P loss reductions. Data were only available for 37 sites at the bottom of 54 FMUs. The remaining FMUs had sites farther up the catchment that may not have captured discharge from a representative mix of grazed (dairy and sheep and beef) and ungrazed land and hence were excluded from further analysis.

Based on the present period of record (usually 15-20 years), we estimated timeframes to detect N loss reductions 'as of 2023' (and adjusted for existing and the real rate of implementation) and for full implementation of actions in grazed land at a mean of 14 and 17 years, respectively for those FMUs where reductions were detectable at the outlet. The commensurate times for P were lower owing to larger percentage reductions. However, in 20 of those FMUs, reductions 'as of 2023' and reductions in 7 FMUs for full implementation could not be detected in 50 years. For P, reductions could not be detected in 4 FMUs as of 2023 but could be detected in all FMUs if full implementation of actions occurred.

Assuming ungrazed (not dairy or sheep and beef) land did not contribute to reductions, monthly sampling to detect the reductions as of 2023 and for full implementation of actions for N was 6 and 12%, respectively, and for P was 9 and 14 %, respectively (**Table 20**). These lower reductions were less detectable resulting in reductions not being detectable in all but 17 and 5 FMUs for N as of 2023 and for full implementation of actions, respectively. Reductions for P were more common, but still not detectable in 14 and 13 FMUs as of 2023 and under full implementation, respectively (**Table 20**).

Short of increasing the magnitude of reductions, to increase the likelihood of detecting change from actions, the frequency of monitoring could be increased at some sites. This would reduce variation, and increase detectability, but in many sites would not as the ability to detect change is more influenced by lag times and/or climatic factors. Here, detecting change is a matter of collecting enough data over time, including data on changing losses from farm production, to remove the confounding effect of climate and lag times on water quality responses. Of course, these FMU-scale results are for farm typologies that may or may not be representative of production and losses across catchments. Hence, more spatially resolved farm-scale data should also be collected on a regular

(e.g., 3 yearly) basis to detect changes in contaminant losses or the effect of actions. Alternatively, detections sites could be moved closer to land with actions implemented to detect change. Owing to the cost of establishing water quality monitoring sites, we temper this recommendation to include a mix of existing long-term sites and new sites located to specifically detect change and give farmers the confidence that the implementation of actions alongside efficiency gains.

Table 20: Nitrogen (N) and Phosphorus (P) loss percentage (%) reductions and timeframes (years) to detect the reductions by freshwater management unit as of 2023 (after adjustment for implementation; first value) and for the full (second values) implementation of actions on combined dairy and sheep and beef land and adjusted for non-grazed land (i.e., discounted by the area in forestry, native bush, arable and horticultural land).

Region	Freshwater Management Unit (FMU)	Combined N reduction (%)	Combined P reduction (%)	Year to detect combined and N loss	Years to detect combined P loss	Combined N loss adjusted for ungrazed land (%)	Combined P loss adjusted for ungrazed land (%)	Year to detect combined and N loss adjusted for ungrazed land	Years to detect combined P loss adjusted for ungrazed land
Bay of Plenty	Kaituna, Maketu and Pongakawa	14, 26	21, 30	15, 5	5, 5	6, 12	9, 14	>50, 20	25, 15
Bay of Plenty	Rotorua Lakes	12, 23	23, 32	15, 5	5, 5	3, 7	7, 9	>50, 40	40, 20
Bay of Plenty	Tarawera	17, 28	18, 29	5, 5	5, 5	5, 9	6, 9	15, 5	15, 5
Bay of Plenty	Tauranga Harbour	11, 25	28, 39	35, 10	10, 5	4, 8	9, 13	>50, >50	50, 30
Bay of Plenty	Waioeka and Otara	7, 28	25, 36	>50, 45	>50, 30	1, 4	4, 5	>50, >50	>50, >50
Bay of Plenty	Whakatane and Tauranga	15, 30	20, 34	>50, 20	45, 20	2, 4	2, 4	>50, >50	>50, >50
Canterbury	Ashburton	29, 45	30, 40	>50, 30	>50, 35	12, 19	13, 17	>50, >50	>50, >50
Canterbury	Banks Peninsula	9, 26	41, 55	>50, >50	20, 15	5, 16	26, 34	>50, >50	>50, 35
Canterbury	Christchurch - West Melton	19, 31	47, 67	20, 10	5, 5	7, 11	17, 24	>50, >50	25, 15
Canterbury	Hurunui - Waiau	25, 30	32, 42	40, 30	25, 15	12, 14	15, 20	>50, >50	>50, >50
Canterbury	Kaikoura	18, 14	24, 36	>50, >50	>50, 25	5, 4	7, 10	>50, >50	>50, >50
Canterbury	Orari-Temuka-Opihi-Pareora	20, 35	38, 49	45, 15	15, 10	11, 20	21, 27	>50, 50	45, 25
Canterbury	Upper Waitaki	16, 25	28, 37	40, 20	15, 10	4, 6	7, 9	>50, >50	>50, >50
Manawatu-Whanganui	Kai Iwi	10, 24	33, 44	>50, 15	10, 5	5, 13	18, 24	>50, 45	25, 15
Manawatu-Whanganui	Manawatu	13, 23	27, 36	>50, 20	15, 10	9, 16	19, 26	>50, 40	30, 20
Manawatu-Whanganui	Puketoi ki Tai	6, 21	33, 43	>50, 25	10, 10	N/A[1]	N/A	5, 5	5, 5
Manawatu-Whanganui	Rangitaikei-Turakina	16, 18	35, 43	>50, >50	30, 25	9, 10	20, 25	>50, >50	>50, >50
Manawatu-Whanganui	Waiopehu	23, 28	22, 29	15, 15	20, 10	9, 11	8, 11	>50, >50	>50, >50
Manawatu-Whanganui	Whangaehu	11, 22	30, 40	>50, 35	20, 10	7, 13	18, 24	>50, >50	50, 30
Manawatu-Whanganui	Whanganui	6, 19	23, 36	>50, >50	40, 20	2, 6	8, 12	>50, >50	>50, >50
Otago	Dunedin Coast	15, 14	42, 48	>50, >50	15, 10	7, 6	19, 21	>50, >50	>50, 50

Otago	Manuherikia Rohe	18, 27	44, 59	50, 25	10, 5	11, 16	27, 36	>50, >50	25, 15
Otago	North Otago	13, 41	34, 50	>50, 10	15, 10	9, 30	24, 35	>50, 20	30, 15
Otago	Taieri	15, 28	44, 57	>50, 20	10, 5	9, 17	27, 35	>50, 50	20, 15
Southland	Aparima	12, 20	33, 42	>50, 30	15, 10	7, 11	19, 24	>50, >50	35, 20
Southland	Mataura	10, 24	36, 46	45, 10	5, 5	6, 15	22, 28	>50, 25	10, 10
Southland	Oreti	10, 24	37, 48	40, 10	5, 5	5, 14	21, 27	>50, 20	10, 5
Southland	Waiau	18, 17	37, 46	45, 50	10, 10	3, 3	7, 9	>50, >50	>50, >50
Waikato	Coromandel	10, 18	17, 25	>50, 50	>50, 25	2, 3	3, 5	>50, >50	>50, >50
Waikato	Hauraki	15, 31	26, 38	35, 10	15, 5	8, 17	14, 20	>50, 30	40, 20
Waikato	Lake Taupo	17, 14	27, 38	35, 50	15, 10	N/A	N/A	5, 5	5, 5
Waikato	Lower Waikato	14, 29	32, 45	>50, 20	15, 10	9, 18	20, 29	>50, 45	35, 20
Waikato	Middle Waikato	20, 32	37, 49	15, 10	5, 5	13, 21	24, 32	40, 15	15, 10
Waikato	Upper Waikato	16, 25	22, 33	15, 10	10, 5	8, 12	11, 16	>50, 25	35, 15
Waikato	West Coast - North	8, 15	17, 25	>50, >50	40, 20	5, 8	9, 14	>50, >50	>50, >50
Waikato	West Coast - South	5, 19	16, 25	>50, 35	50, 25	3, 11	9, 14	>50, >50	>50, >50
West Coast	Hokitika	5, 6	23, 30	>50, >50	50, 30	N/A	N/A	5, 5	5, 5
FMU mean for sites detectable within 50 years		14, 26	21, 30	13.8, 17.4	17.5, 12.7	6, 12	9, 14	11.7, 24.7	26.1, 17.5
FMU assuming sites detectable in 55 years				42.0, 27.8	21.6, 12.7			48.0, 40.3	37.8, 30.7

[1] Power could not be determined.

Conclusions

This report presents evidence that on-farm mitigation actions can lead to measurable improvements in catchment water quality. This evidence is structured around a five-step process designed to maximise confidence in attributing water quality improvements to specific farm-level actions. We highlight several examples**,** both in New Zealand and internationally**,** where applying most or all of this process has provided a strong basis for attribution. However, we also identify shortcomings in many otherwise well-intentioned efforts. These often stem from limitations in the quality and availability of farm-scale data needed to determine where, when, and how effectively actions have been implemented, as well as from insufficient monitoring of water quality outcomes. To address these gaps, we offer 16 recommendations and outline critical data required to support full implementation of the five-step process. We argue that embedding this process within policy—particularly in the context of Freshwater Farm Plans—will empower farmers, industry, communities, and government to take collective ownership of outcomes and establish a credible, evidence-based path toward improved catchment water quality in New Zealand.

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Data availability statement

The data for farm typologies used in the national case study are lodged in Figshare at: https://doi.org/10.57935/AGR.29362100.v1

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