Kōmanawa Solutions Ltd

Our Land and Water National Science Challenge:

Monitoring Freshwater Improvements

Report No: Z22014_01

Water quality monitoring for management of diffuse nitrate pollution

Kōmanawa:

1. (verb) spring, well up (of water)
2. (verb) to spring, well up (of thoughts, ideas)

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Authors

Zeb Etheridge

Director & Principal Water Resource Scientist, Kōmanawa Solutions Ltd

Matt Hanson Dumont

Computational Hydrogeologist, Kōmanawa Solutions Ltd

Evelyn Charlesworth

Water Resource Scientist, Kōmanawa Solutions Ltd

Contributors

Olivier Ausseil (Traverse Environmental Ltd), Richard Mc Dowell (OLW /AgResearch), Alasdair Noble (AgResearch).

The following individuals and organisations have also generously contributed their time and expertise to this project: Shaun Thomsen (Environment Canterbury), Maree Patterson (Horizons), Rebeccah Morris (Greater Wellington), Darren May (Environment Southland).

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1 Executive Summary

Aotearoa New Zealand is making significant investment in national policy and standards, regional plan rules and on-farm actions to reduce the impact of intensive land use on groundwater quality and stream health. Existing freshwater monitoring networks in New Zealand and overseas have predominantly been developed to yield information on the state and trend of freshwater but are not designed, and have often proven ill-suited, to robustly establish cause-effect relationships between improvement actions and water quality outcomes.

Although scientists and statisticians have long recommended conducting a power analysis to inform robust study design, and determination of trends is usually listed as a key monitoring network design objective, implementation of this objective in groundwater quality network design appears to be rare. Water quality monitoring network design and review processes generally focus on the spatial representativeness of the monitoring network in terms of geographic spread and hydrogeological and/or water quality typologies with little consideration for temporal representativeness. The outcome of the current approach is network optimisation bias towards characterisation of water quality state at a broad scale, with little or no consideration of whether the network will yield robust change detection information within the timeframes required by landowners, custodians, communities, and regulators.

Our current approach to monitoring means that we often have limited evidence on whether specific policies or land management actions are improving the health of our waterbodies, and if so, to what extent. This presents a potential obstacle for the NPS-FM aim of achieving water quality improvements within five years and puts significant constraints on our ability to manage water quality effectively. Poor knowledge of the statistical power of our water quality monitoring networks also constrains our ability to identify broader cause and effect relationships between land use and environmental impacts. Unless we change how we monitor freshwater, we are unlikely be able to determine the effectiveness of the significant investment being made to improve freshwater quality or to confidently identify where land use is causing water quality degradation.

This document provides background information on approaches to water quality monitoring network design (with a focus on groundwater), statistical power analysis, and the requirements for effective change-detection monitoring network design. A water quality change detection monitoring design framework is provided in conjunction with a Mitigation Effectiveness Monitoring Design tool suite comprising two main components:

1. A national scale interactive web application which estimates the probability of correctly detecting a reduction in groundwater and/or surface water nitrate concentrations in response to land management actions. The underlying analysis assumes no lags between nitrate loss reductions and associated nitrate concentration reductions at the monitoring site. Note that this assumption can yield significant overestimates of detection power; therefore the web application is best used as a screening tool to preclude spending resource on sites where the outcome is unlikely to be detected.
2. A set of Detection Power python tools hosted in a GitHub Repository. The repository holds associated methodology details and user instructions for site-by-site analysis of statistical power, accounting for lag times and age dispersion.

Three change detection case studies are provided to demonstrate application of the monitoring design framework and tools. The case studies include an analysis of the capability of the existing national SOE groundwater quality network to detect a reduction from current measured nitrate concentrations to a hypothetical 2.4 mg/L NO_3-N target implemented over a 30-year period, based on 30 years of quarterly sampling. The analysis suggests that the improvement would only be detected with a suitable degree of confidence in 40% of wells after 30 years. This means that the effectiveness of policies intended to reduce nitrate concentrations are very unlikely to be identifiable in the groundwater monitoring network within the timeframes required for effective natural resource management. This study concludes that we need to rethink our approach to change detection water quality monitoring. Integrated design of surface water and groundwater quality change detection networks to select optimal change detection monitoring sites across the interconnected hydrological system will be a key part of this.

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2 Introduction

2.1 Background

Aotearoa New Zealand is making significant investment in reducing nitrate discharges to water via national policy and standards, regional plan rules and on-farm actions to reduce the impact of intensive land use on groundwater quality and stream health. The Essential Freshwater package, implemented in September 2020 to protect and restore New Zealand's freshwater includes the new National Policy Statement on Freshwater Management (NPS-FM), National Environmental Standards for Freshwater (NES-F), stock exclusion regulations, and updates to the regulations which cover the measurement and reporting of water takes (MFE, 2020). The NPS-FM aims to "start making immediate improvements so water quality improves within five years" (MFE, 2020).

Existing freshwater monitoring networks in New Zealand and overseas have been developed to yield information on the state and trend of freshwater but are not designed, and have often proven ill-suited, to robustly establish cause-effect relationships between improvement actions (e.g., riparian buffers, wetland restoration, stock exclusion) and water quality outcomes (e.g., reduced contaminant load, improved ecological health). Time lags, attenuation, spatial and temporal variability in contaminant discharge and transport, and other complexities make the measurement of freshwater improvement challenging.

As a result, we often have limited evidence on whether specific policies or land management actions are improving the overall health of our waterbodies, or to what extent. This presents a potential obstacle for the NPS-FM aim of achieving water quality improvements within five years. Unless we change how we monitor freshwater we are unlikely be able to determine the effectiveness of the significant investment being made to improve our freshwater resource.

2.2 Monitoring Freshwater Improvements programme

The aim of the Monitoring Freshwater Improvements programme is to develop a toolkit that aids potential end-users (i.e., all groups involved in freshwater improvement actions, including councils, iwi, co-governance entities and catchment groups) to robustly monitor the improvement of freshwater by land mitigation, land management and in-system (e.g., in-lake, in-stream) actions. The toolkit provides guidance on how to robustly monitor specific attributes in rivers, lakes and groundwater; where, when (including how often and how long), with what technology and the associated cost. The toolkit enables end-users to address the following:

1. Existing network performance: Test whether the existing monitoring network is suitable to detect the potential freshwater changes (i.e., improvements) brought by a given mitigation plan.
2. Developing optimised networks: Provide guidance on monitoring network design and monitoring technologies that optimise the ability to detect the effects of the mitigation plan.
3. Preventing wasted resource: demonstrate where monitoring is unlikely to be effective or is being undertaken at a higher frequency than is needed; allowing resources to be re-distributed to other sites.

2.3 Report overview

This document focuses on groundwater monitoring network design for management of diffuse nitrate contamination with a focus on change detection and includes the following components:

- A review of current approaches to groundwater quality monitoring nationally and internationally;
- An evaluation of the requirements for change detection monitoring;
- A recommended approach for change detection monitoring for diffuse nitrate contamination;
- A summary of work undertaken to assess the change detection capability of New Zealand's national groundwater monitoring programme using the recommended approach; and
- Example applications of the recommended approach for two case study catchments

3 Current approaches to water quality monitoring

3.1 Summary

Consideration of statistical error in the analysis of water quality monitoring data is a key part of robust data interpretation. Quantifying the likelihood of a Type I error (a false positive, e.g. the analysis of monitoring results incorrectly concluding that an ineffective management action is improving water quality) and Type II error (a false negative, e.g. analysis incorrectly concluding that an effective action is not improving water quality) is key when making land management decisions from monitoring results.

Although scientists and statisticians have long recommended conducting a power analysis to inform robust study design (Weiser et al., 2021) and determination of trends is often listed as a key monitoring network design objective (e.g., Moreau-Fournier & Daughney, 2012), effective implementation of this objective in network design appears to be rare.

Power analysis is straightforward for low complexity study systems and questions, with lookup tables available for some simple tests (e.g., t-tests). For more complex situations, such as a study that repeatedly monitors multiple sampling units ("sites") [as per a groundwater quality monitoring network], a simulation-based approach to power analysis is more flexible and accurate. Performing computer simulations requires technical expertise and computing resources, however, and designers of monitoring programmes might not have the necessary resources to conduct a simulation-based power analysis. As a result, power analyses are often neglected for monitoring programmes and other scientific studies (Weiser et al., 2021).

We reviewed groundwater quality monitoring network design and review processes in New Zealand, Europe and Scotland to understand current approaches and found that these tend to focus on the spatial representativeness of the monitoring network in terms of geographic spread and hydrogeological and/or water quality typologies. Integrated design of groundwater and surface water quality monitoring networks for optimal change detection appears to be rare. The outcome of the current approach is network optimisation bias towards characterisation of water quality state at a broad scale, with little or no consideration of whether the network will yield robust decision-support information within the timeframes required by land managers, custodians, and regulators.

3.2 Groundwater quality monitoring in New Zealand

Groundwater quality monitoring programmes in New Zealand fall within four main categories:

- The National Groundwater Monitoring Programme
- State of the Environment monitoring
- Targeted water quality investigations
- Compliance monitoring for consented activities

The National Groundwater Monitoring Programme (NGMP) is a long-term research and monitoring programme operated by GNS Science in collaboration with 15 Regional Councils and includes 110 monitoring sites across the country, monitored quarterly. The stated aims of the NGMP are to 1) provide a national overview of groundwater quality in New Zealand, including determination of the natural "baseline", 2) identify spatial and temporal patterns in groundwater quality and associate them with certain causes, such as human influence, at a broad scale; and 3) develop and convey best-practice methods for groundwater sampling, chemical analysis and interpretation (Moreau-Fournier & Daughney, 2012).

State of the Environment (SOE) monitoring in New Zealand is undertaken by 15 regional councils and unitary authorities and aims to: 1) characterise groundwater quality in terms of current state and trends; 2) associate observed state and trends in groundwater quality with specific causes such as land use, pollution or natural processes; and 3) provide data to assess the effectiveness of groundwater management policies (Moreau-Fournier & Daughney, 2012). These combined monitoring networks currently include 763 sites with regular monitoring of nitrate of sufficient duration for the purposes of our study. The SOE networks have grown organically over time, with periodic network reviews undertaken by some regional councils. Reviews typically focus on the spatial and hydrogeological typology coverage of the network.

Targeted water quality investigations are undertaken by some regional councils to provide more detailed local information. This may be required where specific water quality issues have been identified (e.g. Pearson et al., 2022) or when increased spatial resolution is required to understand the current state of water quality to support development of water quality management rules in regional planning processes (e.g. Scott et al., 2016) A national survey of pesticides in groundwater has been undertaken since 1990 covering approximately 165 wells which are tested for > 80 pesticides.

Daughney et al., (2012) assess the representativeness of the NGMP network through comparison to the regional SOE networks via hierarchical cluster analysis. The authors consider that sites comprising any baseline monitoring network must be selected to provide a representative perspective of groundwater quality across the aquifer(s) of interest. The national and regional networks were compared in terms of the number of water quality categories identified in each network, the hydrochemistry at the centroids of these water quality categories, the proportions of monitoring sites assigned to each water quality category, and the range of concentrations for each analyte within each water-quality category. The study concluded that the National Groundwater Monitoring Programme (117 sites) provides highly representative perspective of groundwater quality across New Zealand relative to the amalgamated regional (SOE) monitoring networks (~700 sites).

3.3 Groundwater quality monitoring under the WFD

3.3.1 Monitoring requirements

The European Union Water Framework Directive (WFD) requires member states to establish programmes for the monitoring of groundwater to determine groundwater quality status, long-term trends in natural conditions and trends in groundwater bodies resulting from human activity (European Commission, 2007). The WFD sets out the requirements for the different groundwater monitoring programmes which must include:

- A quantitative monitoring network to facilitate status assessment (e.g., determination of whether the chemical status of a groundwater body should be classified as Good/Poor)
- A surveillance monitoring network to: (a) supplement and validate the water quality status assessment and assess the risks of failing to achieve good groundwater chemical status; (b) provide information for assessment of long term naturally driven and anthropogenically driven trends and (c) to establish, in conjunction with the risk assessment, the need for operational monitoring.
- An operational monitoring network to: (a) establish the status of all groundwater bodies, or groups of bodies, determined as being 'at risk', and (b) establish the presence of significant and sustained upward trends in the concentration of pollutants.

The requirements for monitoring results evaluation include the following:

- establish the chemical status of groundwater;
- validate water quality status risk assessments;
- evaluate the effectiveness of programmes of measures to protect/restore water bodies to reach good status;
- characterise the natural quality of groundwater including natural trends (baseline); and
- identify anthropogenically induced trends in pollutant concentrations and their reversal (European Commission, 2007).

Surveillance monitoring must be undertaken during each six-year planning cycle. The two primary goals of this monitoring (to support assessment of water quality status and to detect long-term trends in status) require different monitoring frequencies: status assessment may only require monitoring in one or two years of the six-year management cycle while high frequency monitoring over many years may be needed to provide the power needed to detect trends (Carvalho et al., 2019).

The WFD monitoring programme is intended to focus on phenomena affecting the overall state of the groundwater body. Local scale pollution processes which do not affect the overall state of the groundwater body should be the target of different monitoring activities run by the appropriate competent authorities (European Commission, 2007).

3.3.2 Monitoring design guidance

The Common Implementation Strategy for the Water Framework Directive (WFD) Guidance on Groundwater Monitoring (European Commission, 2007) provides guidance on establishing groundwater monitoring programmes to meet the requirements of the WFD and the Groundwater Directive. Although much of the guidance is specific to the WFD requirements, several recommendations are relevant for all diffuse groundwater pollution monitoring. The guidance advocates a conceptual hydrogeological model-based approach to network design. This approach emphasises the importance of founding monitoring design on a good understanding of the hydrogeological system and sources of contamination and highlights the following processes and variables:

- Flow through the system in three dimensions, considering vertical variations in aquifer properties and stratification of water quality
- The spatial and temporal variability of recharge sources, pollution sources and flow paths
- Travel times through the groundwater system at different depths and locations
- The effects of groundwater abstraction on flow paths

The guidance advises that monitoring frequency should be determined based on the variability of the system. Shallow groundwater systems are generally more dynamic and hence more frequent sampling may be required; a reduced sampling frequency (e.g. two samples per year) may be sufficient initially for surveillance monitoring in less dynamic systems. If this monitoring shows no significant variation, a further reduction of sampling frequency may be appropriate according to the guidance (European Commission, 2007, p. 17).

The guidance also notes that sampling of surface water may provide a representative groundwater sample where groundwater contributes significantly to (base)flow. A selection of sampling points with relatively young water is recommended.

European Commission Guidance Document No. 7: Monitoring under the water framework directive (European Commission, 2003) includes a discussion of network design for the purpose of trend or difference detection and notes that there are two types of error to consider:

- Type I error - the likelihood of a false positive (i.e. analysis of monitoring results incorrectly concludes that an ineffective mitigation is improving water quality)
- Type II error - the likelihood of a false negative (i.e. analysis of monitoring results incorrectly concludes that an effective mitigation is not improving water quality)

This means that the following should be considered in monitoring network design:

- The parameter to be estimated (e.g. the before-after mean difference, or the slope of a trend line);
- The required confidence (C%) associated with any assertion that a change has been detected (e.g. 90%, 99%). The likelihood of a Type I error is then given by (100 - C) %; and
- The required confidence that a Type II error has been avoided.

3.3.3 Groundwater nitrate monitoring in Scotland

O' Dochartaigh et al. (2007) reviewed the effectiveness of Scotland's groundwater nitrate monitoring network, which comprised 219 monitoring points at the time of the review. Evaluation of the network effectiveness was based on an assessment of the reliability of individual monitoring points (vulnerability to direct local point source contamination, access, and sampling procedures) and a holistic network evaluation. The latter comprised estimation of the zone of influence of each sampling point by scaling a shuttlecock shaped recharge area, such that the area of the shape was equal to annual abstraction from the monitoring well divided by the annual rate of recharge to the land surface. The characteristics of the network were evaluated in terms of land use within the estimated zone of influence, monitoring point type (bore/well/spring and bore depth class), aquifer productivity class and groundwater vulnerability class. Well depth and groundwater vulnerability were used as proxies for groundwater age, with deep wells and locations with deep vadose zones and impermeable cover associated with old groundwater and long lag times. The existing network was compared to a hypothetical idealised network designed to have a wide geographic spread based primarily on nitrate loading. The approach determined that monitoring well density should be proportional to nitrate loading rates, with areas of the country with high loading rates requiring more monitoring points. The representativeness of the network was also assessed against variability in geology, soil types and hydrogeology across the country. The study concluded that the Scottish groundwater nitrate monitoring network generally represents the diversity of land use and groundwater abstraction point types across Scotland and that the current geographic distribution of the network compares well with a hypothetical network based on nitrate loading and land use, with some areas for improvement identified. The study noted that without a representative network of reliable monitoring points, there is a risk that groundwater management policies could be developed and implemented based on poor evidence.

4 Change detection monitoring requirements

4.1 Overview

Early and robust detection of water quality management action effectiveness is vital given the high cost of nitrate loss mitigations, the associated requirement for long term financial planning and the goal of achieving water quality improvements within five years. The key requirements for robust detection are responsiveness, detection power, and representativeness.

Monitoring wells installed in hydrogeological units with low transport velocities between contaminant sources and the monitoring well locations (i.e. long lag times) will not be responsive to nitrate loss reductions within the timeframes required to determine whether mitigations are being implemented and achieving the desired outcome. Conversely, surface water monitoring sites fed by shallow groundwater systems may respond to loss reductions relatively quickly.

Impractically long monitoring periods may be required to confidently evaluate policy and or mitigation effectiveness for surface and groundwater monitoring sites with a high ratio of noise (e.g. background variability due to weather and climate) to signal (nitrate concentration reduction due to mitigation), i.e. a low detection power. Surface water sites with intermittent runoff from low intensity hill catchments coupled with baseflow from groundwater drainage from intensively farmed lowland catchment may be noisier than groundwater monitoring sites.

Because nitrate management actions are neither undertaken on all land uses nor to all land within a farm boundary equally and/or with equal effectiveness, monitoring results from a small number of monitoring wells with unknown groundwater recharge areas or surface water catchments which do not capture flow from areas where mitigations are implemented may not provide a reliable information on the effectiveness of nitrate management actions. As such, the probability of Type I or Type II statistical errors may be too high for robust decision support purposes. Surface water sites integrate water quality from a much wider area than monitoring wells and are generally expected to provide a more spatially representative sample than individual groundwater monitoring sites.

4.2 Water age and monitoring network responsiveness analysis

Determination of the lag time between a change in nitrogen inputs (e.g. implementation of a mitigation programme) and the associated response at a given monitoring location (well, stream, river or lake) is a pre-requisite for determination of the monitoring duration required to detect whether the mitigation has been successful.

Groundwater flow is always accompanied by dispersion or mixing on a range of scales. At the fine scale, dispersion relates to the various paths taken by water through the rock matrix or between grains. At a larger scale, waters following completely different paths through an aquifer, or different aquifers can be drawn into a well screen by pumping. This means that the well water can have a range of ages from young to old, and its age is characterised by a mean age (M. Stewart, 2006). Realistic assessments of groundwater age measurements in heterogeneous systems or where dilution processes take place (e.g. recharge, matrix diffusion) should recognise that water samples contain a distribution of ages. The travel time probability density function characterises the distribution of possible travel times that a water molecule might experience in moving from the recharge zones to the measurement points (Varni & Carrera, 1998).

Water age is often inferred via sampling of radiometric tracers (tritium and carbobn-14) and/or CFC and SF6. The age distribution at a point can vary between the delta distribution, corresponding to piston flow, and the exponential distribution that assumes perfect mixing in the aquifer. The only case in which the radiometric age equals the mean age is when the piston flow model can be applied, i.e. when the sample is not the subject of mixing along the flow path (Varni & Carrera, 1998).

The Exponential Piston-flow Model (EPM) 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 (M. K. Stewart, 2012).

A double EPM (DEPM) model can be used to describe short-residence-time and long-residence-time flow components in groundwater system, for example near-surface and deep flows to a river, or shallow lateral and deep flows to a groundwater well. The DEPM model is formed by adding the two EPM models (M. K. Stewart, 2012):

$$DEPM = b(EPM_1) + (1 - b)(EPM_2)$$

The differences between these models are illustrated in Figure 4-1, which plots groundwater age as a cumulative density function for EPM and DEPM models with an equal (8-year) MRT and 60% mixing. DEPM 1 comprises 50% water with a one-year MRT and 50% water with a 15-year MRT; DEPM2 comprises 30% water with a 1-year MRT and 70% water with an 11-year MRT.

Considering this possible range of age distributions in the context of nitrate mitigation effectiveness monitoring for a leaching rate reduction of 50%, for example, a groundwater sample with an MRT of 8 years could potentially record a ~20% concentration reduction within three years in the case of DEPM 1 but no change in concentration in the case of the EPM. Failing to account for age distribution, alternative age distribution models, and model parameterisations when designing and interpreting groundwater nitrate monitoring results increases the likelihood of statistical error.

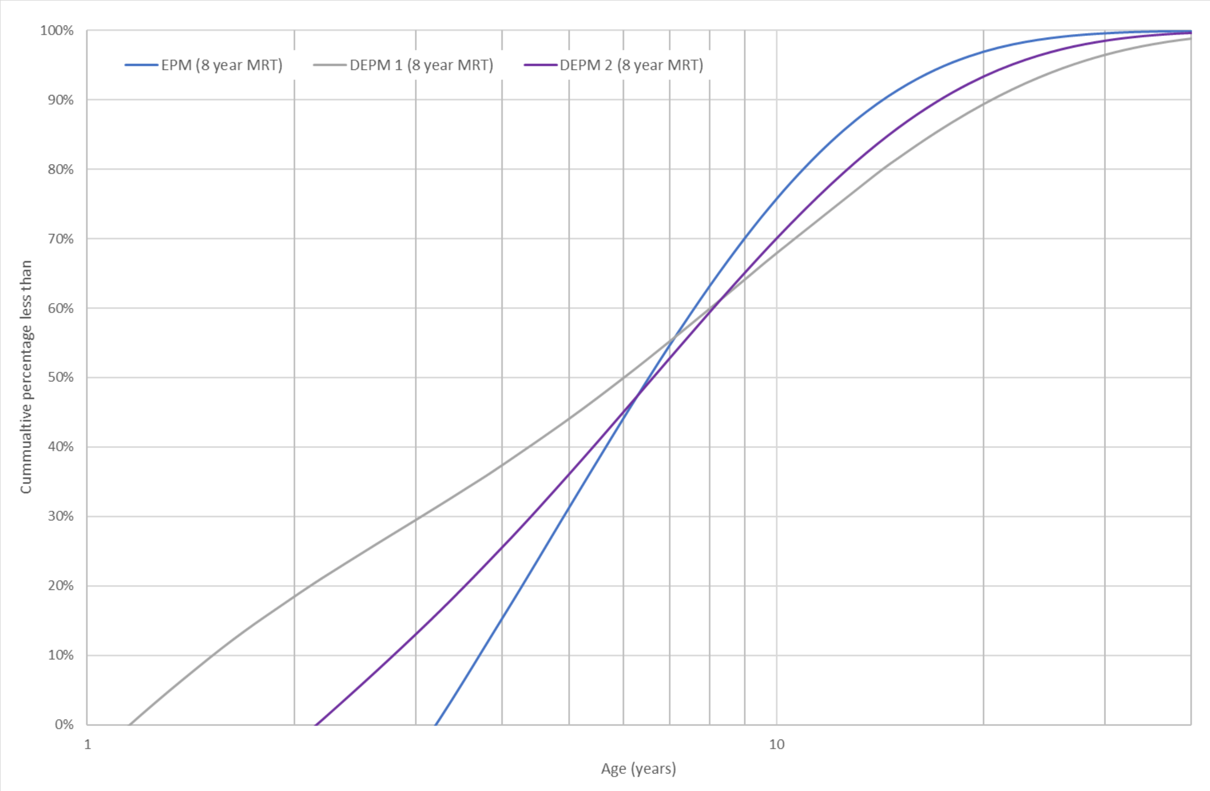


Figure 4-1 Age distribution for 8-year MRT under EPM and BMMs

4.3 Monitoring network detection power

4.3.1 Introduction

The success of any scientific study or monitoring program relies on its ability to accurately measure the desired attribute or relationship. Long-term monitoring programmes are typically designed to track changes over time, but insufficient statistical power can lead to failed monitoring programme objectives and wasted resources. If the monitoring program detects a trend when none is present (Type I error), fails to detect a real trend (Type II error), or estimates a trend that is opposite to the one present (Type III error), any management decisions based on the apparent population trend would be counterproductive to the management objective(s) (Weiser et al., 2021).

Power analysis allows us to determine the sample size required to detect an effect (e.g., a load change) of a given size with a specified degree of confidence. Conversely, it can be used to determine the probability of detecting an effect of a given size with a given level of confidence for a specified sampling frequency and duration (Whitehead, 2021). Power analyses can therefore be used in several ways, e.g.

I. Testing the ability of the existing monitoring network to detect the expected nitrate concentration change for a given policy or mitigation plan. The likelihood of detecting a change within five years via an existing monitoring network at a given sampling frequency could be evaluated, for example.
II. Determine locations where the likelihood of detecting a change within a given period is highest.
III. Test whether increasing sampling frequency, or changing the sampling regime, is likely to increase the change detection power (Ausseil et al., 2021).
IV. Identifying locations where resource is being wasted either because detection is unlikely or because sampling frequency is higher than needed for the given purpose.

Power analysis is straightforward for low complexity study systems and questions, with lookup tables available for some simple tests (e.g., t-tests). For more complex situations, such as a study that repeatedly monitors multiple sampling units ("sites") [as per a groundwater quality monitoring network], a simulation-based approach to power analysis is more flexible and accurate. Performing computer simulations requires technical expertise and computing resources, however, and designers of monitoring programmes might not have the necessary resources to conduct a simulation-based power analysis. As a result, power analyses are often neglected for monitoring programmes and other scientific studies (Weiser et al., 2021).

4.3.2 Power analysis for detecting effect size

Whitehead (2021) explains that the following four power analyses terms are inter-related:

- **Sample size:** the number of samples or sites.
- **Effect size:** difference between group means or the group mean and a reference state or regulatory limit.
- **Significance level:** P (Type I error) or probability of finding an effect that is not there (i.e. the likelihood of a false positive).
- **Power:** 1 - P (Type II error) or probability of finding an effect that is there.

If any three of these terms are known, the fourth can be calculated.

The power to detect change is also related to the variability of the data set, where a higher standard deviation relative to the mean will result in lower power to detect an effect. These effects are illustrated in Figure 4-2. Panel A shows the measured time series of Nitrate-Nitrite Nitrogen (NNN) at each site. Note that the y-axes differ between sites, with considerably lower NNN values measured at the Ashley River site. Panel B shows the power available (colour) to detect a % reduction in NNN (x-axis) for different numbers of monthly samples (y-axis). The dotted black line indicates the contour at which we would be 80% confident of detecting a reduction (power = 0.8). The power to detect a reduction increases with both the number of samples and the magnitude of the reduction at both sites. However, there is less power to detect a reduction at the Ashley River site because the relative magnitude of the reduction is small compared to the variability in the data[1].

[1] Calculated using *power.t.test()* in R

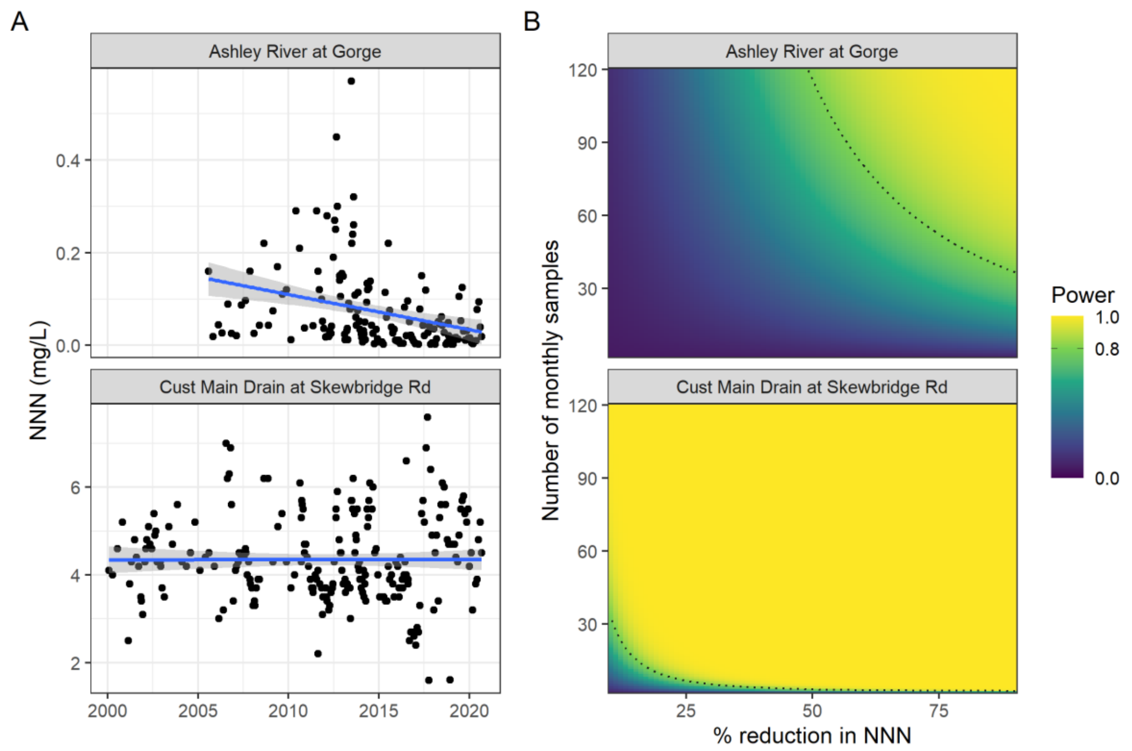


Figure 4-2 Number of monthly samples required to detect a nitrate nitrite nitrogen (NNN) reduction at two river sites in Canterbury (from (Whitehead, 2021)).

Because high-frequency monitoring can generate a significantly larger sample of nitrate time series data than discrete sampling, the probability of detecting statistically significant changes within a given period is higher. This may mean that the effectiveness of nitrate loss management actions can be determined more quickly and/or with more certainty. The magnitude of this potential time saving/certainty improvement is a function of serial correlation, the periodicity of background variability (aka "noise") at the monitoring site and the statistical method used to analyse the data.

Background variability due to weather, climate and anthropogenic influences can produce "noise" at a range of frequencies. Periods of groundwater recharge following rain can cause short term decreases or increases in nitrate concentrations, for example, depending on whether the recharge mainly dilutes nitrate already present in groundwater or mainly transports nitrate stored in the soil and vadose zone into the aquifer. The latter can occur when rainfall follows an extended dry period, during which nitrate has accumulated in soil/vadose zone pore water. In the mid to longer term, nitrate concentrations can vary in response to climate variability at scales ranging from inter-annual to inter-decadal. Snelder et al., (2021) evaluated the correlation between climate and surface water quality observations using the Southern Oscillation Index (SOI) as a climate pattern indicator. Model results indicated that SOI trends are associated with trends in the six water quality variables (including nitrate) at the 10-year timescale and, to a lesser degree, at the 20-year timescale. Land use change signals at the 10-year timescale were generally swamped by the noise of climate variability but were more discernible at the 20-year timescale.

Serial correlation (also known as autocorrelation) describes the degree to which the value of a data point in a time series is related to the values of one or more previous time points. It is essentially a measure of how data points in a time series are correlated with each other over time. A high degree of serial correlation could reduce the value of higher frequency data. By way of example, Close (1989) analysed the serial correlation of a groundwater nitrate dataset comprising 87 monthly readings in north Canterbury and found an optimal sampling frequency between 5 and 13 months for detection of step or linear trends in the data. The author noted that erroneous conclusions could be reached if the effects of serial correlation are ignored. Constraining sampling frequency based on autocorrelation analysis results may be counterproductive, however, because autocorrelation information could provide a basis for signal decomposition to remove what might otherwise appear to be random noise. Examples include rainfall event-based nitrate flushes from the soil and vadose zone to the water table, seasonal patterns in nitrate leaching rates from the soil profile (see Trolove et al., 2019) and the effects of climate variability in response to interdecadal cycles on nutrient concentrations in surface waters (Snelder et al., 2021). Characterising these patterns and removing them from the monitoring data would improve the statistical power of the monitoring site.

The following conclusions can be drawn from the information above:

- Increasing monitoring frequency has the potential to reduce the time required to confidently detect change but the periodicity and magnitude of the major components of “noise” must be accounted for.
- Increasing sampling frequency will have diminishing returns at some point; however, this point is likely to be site specific, and difficult to predict. For instance, a shallow well with a short mean residence time will likely benefit more from higher sampling frequency than a deep bore with a large MRT.
- Sampling at higher frequency (e.g., weekly) can also provide information on previously unexplained noise components. If variations in NO_3-N can be attributed to physical processes, they can then be accounted for in data analysis (e.g., signal decomposition) to reduce the observed noise and increase the detection power. The likelihood and extent of these improvements are likely to be site specific and difficult to predict but can be significant.

4.3.3 Power analysis for detecting trends

Power analysis can be used in trend detection to estimate the probability of rejecting the hypothesis that no trend is present when in fact there is a trend (Irvine et al., 2012), i.e. a Type II error. The statistical power of a monitoring network can be increased by increasing the sample size (e.g., the number of years of data collection) and/or reducing the variation (noise) in the data (e.g., by selecting sampling sites with lower variance). Other factors that contribute to statistical power include the specific effect size of interest and the probability of Type 1 error (α), as explained previously. The various combinations of sampling duration, frequency and effect size that equate to a given power for a given parameter at a given location can be evaluated iteratively via power calculations. An example of the use of power analysis for detecting trends using measured nitrate (mg/L) data from a spring fed stream in Canterbury (Kaiapoi River at Island Road) is provided in Figure 4-3. Panel A shows the time series of measured data, while Panel B estimates the power available to detect a trend at alpha = 0.05 as successive samples from the time series are added. This analysis suggests that ~83 samples would be required to be 80% confident of detecting a significant trend, which equates to approximately 12 years based on the frequency of sampling in this time series (Whitehead, 2021), ignoring lag times.

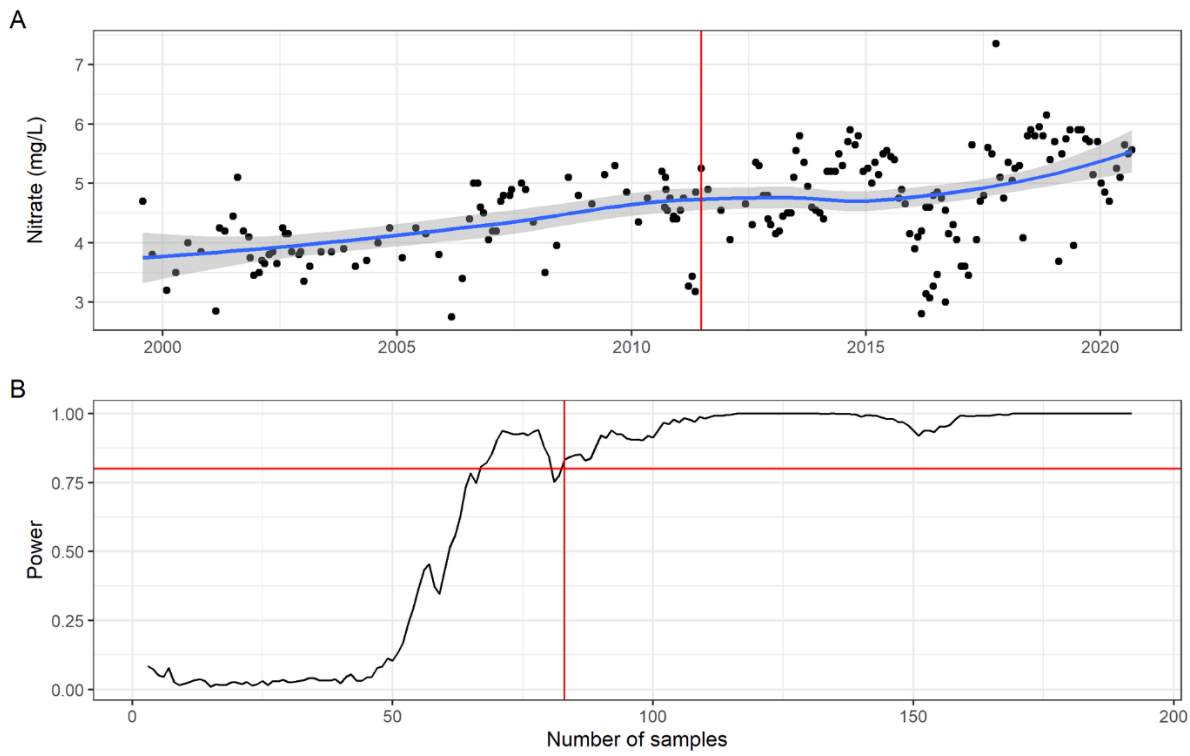


Figure 4-3 Example of power analysis for detecting trends (from Whitehead, 2021)[2]

4.4 Spatially representative sampling

The representativeness of groundwater quality monitoring networks has typically been considered on broad scales in terms of spatial coverage, land use classes and hydrogeological and hydrogeochemical typologies as discussed previously. A different approach is required for nitrate loss mitigation effectiveness monitoring network design and evaluation: representativeness analysis must consider spatial variance in mitigation at the scale of the recharge area of the monitoring site(s).

Lilburne et al. (2012) note that because urine patches are the primary source of nitrate in drainage waters from grazed pastures, any paddock-scale sampling system must adequately sample the urine patches in the correct proportion to non-urine areas. The study evaluated the spatial coverage areas of suction cup and different sized lysimeters and found that an impractical number of samplers are needed to achieve nitrate leaching rate estimates accurate to within ±20% of the true value. The authors concluded that rather than trying to directly measure paddock-scale leaching under grazing, further consideration should be given to wider deployment of controlled nitrogen application onto a few lysimeters followed by extrapolation from the resulting measurements of leached nitrate to the paddock scale and beyond based on urine patch coverage.

Whitehead and Etheridge (2021) applied statistical power analysis to land use, nitrate loss mitigation and numerically modelled nitrate concentration spatial variance to estimate the number of groundwater monitoring sites required for representative sampling of the target population via a case study of a network of monitoring wells in the Waimakariri District, north of Christchurch. Key components of their methodology included:

1. Determination of the expected nitrogen loss reductions for each land use type under nitrate reductions mandated by the regional plan.

[2] Calculated using *power.trend()* in the *emon* package in R

2. Assessment of the spatial variability in both nitrate loss rates and the expected rate reductions within the monitoring area.
3. Evaluation of the spatial standard deviation of groundwater nitrate concentrations from numerical groundwater model simulation results for the water table layer. The numerical MODFLOW-MT3D model incorporated nitrate leaching rates from a biophysical process model (OVERSEER) and land surface recharge from a soil water budget and simulated dispersion and mixing at the water table.
4. A 15% nitrate loss reduction was assumed for dairy farms and a 5% loss reduction for other land uses which require a land use consent under the regional plan for the study area (e.g. sheep, beef, deer, horticulture, pig farming) based on the regional plan rules proposed at the time of the study.
5. In the absence of information on the recharge/capture zone for each monitoring well it was assumed that the modelled nitrate concentrations in each well is representative of the land use and soil type classification polygon within which it was located. The median area of the land use/soil type polygons intersected by monitoring wells was 20 ha. Taking the average modelled land surface recharge rate for the model domain of approximately 0.25 m/year, a 20-ha area would provide enough water to supply a well pumping at ~140 m^3/d on average. 13 of the 20 monitoring wells evaluated comprise domestic supply wells that are likely to be pumped at a rate in the order of 10 m^3/d, and therefore draw water from a much smaller recharge area. Three of the 20 are dedicated monitoring wells which are likely to intercept water from a very small recharge area. The true recharge area of the monitoring wells is therefore likely to fall within a single polygon and hence the recharge area assumption is unlikely to have overestimated spatial variance.

Given that the method used to estimate spatial variability in nitrate loss rates and the expected rate reductions is very likely to under-represent the true variance due to the coarse land use and soil type polygons used to generate these data, this methodology is much more likely to overestimate the detection power of the network than to underestimate it.

Power analysis was undertaken to estimate the ability of the case study monitoring network to detect the expected reduction in nitrate across the region, assuming a Type I error target of 0.05. Results plotted in Figure 4-4 show that the power available to detect a reduction in nitrate increases as the magnitude of the reduction and/or the number of monitoring wells increases. The existing monitoring network of 16 wells is unlikely to detect the median expected change in nitrate (power = 0.264). Based on this analysis, a minimum of 100 sites (assuming that they are randomly distributed with respect to spatial variability in nitrate loss mitigations) would be required to detect a 20% reduction in dairy farm nitrate leaching. A median reduction in nitrate leaching of at least 47% would be required to provide 80% confidence of detecting the reduction via the existing 16 monitoring sites.

Noting: a) that installation and maintenance of 100 monitoring sites is likely to be prohibitively expensive; and b) the under-representation of variance in the method (such that 100 sites may be an underestimate); adoption of targeted mitigation monitoring sites was suggested by the authors as a potentially more practical alternative for groundwater monitoring.

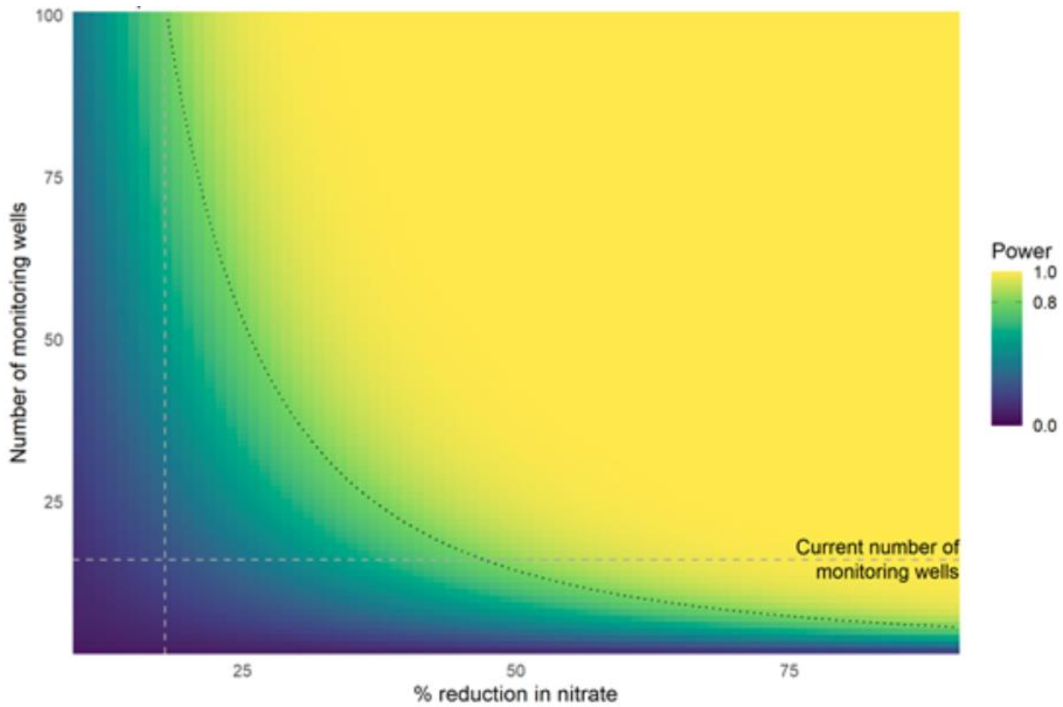


Figure 4-4 **Power analysis results for network spatial representativeness analysis (from Whitehead and Etheridge, 2021)**

Notes: power available (colour) to detect a % reduction in nitrate (x-axis) across the region as the number of monitoring wells increases (y-axis). The dashed grey lines represent the current number of monitoring wells (16) and the expected median reduction under the nitrate loss reduction scenario (regional median = 17.9%). The dotted black line indicates the contour at which we would be 80% confident of detecting a reduction (power = 0.8).

The key finding of our review of Lilburne et al. (2012) and Whitehead and Etheridge (2021) is that, despite significant differences in the scale of monitoring point coverage and spatial variability in nitrogen losses, both studies show that the groundwater monitoring network density required to achieve an acceptable probability of statistical error via randomly distributed monitoring sites is unlikely to be practically achievable. Selection of a limited set of targeted monitoring wells with identifiable catchment areas under an experimental design framework such as the Before-After Control-Impact (BACI) methodology (see Stewart-Oaten & Bence, 2001) may prove to be a cost effective approach for determination of mitigation and/or policy effectiveness and progress rates. Surface water monitoring results may provide insights into the representativeness of a groundwater monitoring network and in some cases will be the optimal approach to change detection monitoring.

5 Nitrate mitigation effectiveness monitoring design framework and toolkit

5.1 Monitoring design framework

The key nitrate mitigation effectiveness water quality monitoring design requirements of responsiveness, detection power and representativeness can be achieved by via the following network review and design process:

1. Define mitigation plans/scenarios and monitoring goals (e.g., determine whether mitigations are reducing nitrate concentrations in an FMU within 10 years with 80% confidence).
2. Develop a conceptual model of the monitoring area to include nitrate load distribution and expected reduction rates, travel paths, attenuation and transit times between sources and receptors. Identify and fill or account for key knowledge gaps.
3. Carry out an integrated analysis of groundwater and surface water detection power for existing sites in the monitoring area, accounting for transit times where appropriate, and identify the highest detection power sites.
4. Evaluate representativeness of priority monitoring sites in relation to the expected spatial distribution and distribution of nitrate loss reductions and the number of sites required to confidently detect change via the methods illustrated in Section 4.4.
5. Identify new monitoring sites if existing network detection power and/or representativeness is inadequate.
6. Undertake a sampling frequency cost-benefit analysis.
7. Finalise network and monitoring design.
8. Review data after 1, 3 and 5 years of sampling to determine whether detection power and timeframe requirements have changed in light of new information.

The purpose of step 4 is to select a spatially representative network of monitoring sites, considering nitrate leaching and mitigation spatial variance and distribution and monitoring site recharge/capture areas in relation to the variance and distribution.

Case study examples provided in Section 6 illustrate the application of this framework, with a summary provided in Figure 6-1.

As per the Section 4.4 discussion, a possible outcome of step 5 is that an impractically large number of monitoring sites may be required to achieve an acceptable degree of confidence in mitigation effectiveness monitoring results. Selection of a limited set of targeted monitoring wells with identifiable catchment areas under an experimental design framework may prove to be a cost-effective alternative. Development of a methodology for this alternative approach is beyond the scope of this document.

5.2 Monitoring design toolkit

5.2.1 Overview

The Groundwater Quality element of the Mitigation Effectiveness Monitoring Design tool suite comprises two main components:

1. A national scale interactive web application which estimates the probability of correctly detecting a reduction in groundwater and/or surface water nitrate concentrations in response to land management actions. The underlying analysis assumes no lags between nitrate loss reductions and associated nitrate concentration reductions at the monitoring site. Note that this assumption can yield significant overestimates of detection power; therefore the web application is best used as a screening tool to identify sites where change detection power is likely to be poor.

2. A set of Detection Power python tools hosted in a GitHub Repository. The repository holds associated methodology details and user instructions for site-by-site analysis of statistical power, accounting for lag times and age dispersion.

Although our work focuses on detection of changes in nitrate concentrations associated with management actions to reduce leaching rates, the Groundwater Detection Power repository and the framework above are applicable to a) change detection in either direction; b) to other contaminants; and c) to both groundwater and surface water sites. This means that the tools and guidance can be used to support design of integrated water quality monitoring networks for a range of contaminants and for situations where concentrations are expected to increase or decrease due to land use change and land management actions.

An underlying assumption of the tools and methods is that the contaminants are conservative, i.e., no process other than dilution and dispersion will affect contaminant concentrations. The use of these tools for non-conservative tracers (e.g., nitrate where denitrification is occurring) may yield incorrect results.

5.2.2 How to use the web application

Full details of the web application are provided in the Mitigation Effectiveness Monitoring Design WebApp User Guide. A summary of the groundwater application and its intended usage is provided below.

The interactive web application for groundwater is populated with information and modelling results from approximately 950 regional council monitoring wells[3] and has the following set of user-selected variables:

(1) Select a Regional Council to access data for your area of interest.
(2) Select an expected or proposed percentage reduction in groundwater nitrate concentrations associated with a set of land use/management actions.
(3) Select an indicator (nitrate is the only option for groundwater).
(4) Select the sampling duration. This is the period over which you will collect samples to determine whether the land management actions have reduced nitrate concentrations; and
(5) Select a sampling frequency (how often you will collect nitrate samples).
(6) Download the results. This step is optional because the results are displayed on the interactive map. The user can interrogate the map by clicking on each coloured dot.

The colour-coded dots on the map symbolise the detection power of each monitoring well based on the selected sampling length and frequency. The example below shows the detection power for each well in the Hawkes Bay region based on five years of sampling at a monthly interval (a) and 20 years of monthly sampling (b). A nitrate concentration reduction of 25% is specified in both instances. Focusing on the two wells to the southeast of Hastings and immediately northeast of Havelock North, the probability of correctly detecting a successful nitrate loss reduction would be in the order of 20-30% for (a) and 70-80% for (b).

Clicking on a well opens an information summary box with the specified nitrate reduction, the well depth and mean residence time information, as shown in Figure 1. The mean residence time summary data are based on a search of all available MRT data provided to us by regional councils either for that well (where available) or for wells with MRT data available (>80% of monitoring wells in New Zealand with no mean residence time data). In the latter case the application searches for the minimum, median, and maximum recorded MRTs within a distance radius and a depth window as follows:

Depth band = well depth ± 10 m

[3] Note that groundwater quality data were not available from Otago Regional Council at the time the analysis was undertaken and hence there is no information for this council.

Distance search = 0.5 km, 1 km, 2.5 km, 5 km, 10 km, 25 km, 50 km, 100 km

For the examples below, there are no MRT data for wells within 50 km of well 10496 within the 0 to 18 m depth band but there is at least one well with an MRT <1 year (shown as 0 years) within 100 km. There are no wells within 50 km of well 22 within the 4-24 m depth band within 50 km, but there are several wells within 100 km distance with MRTs ranging from <1 year (shown as 0 years) to 44 years. Given the significance distance from wells 10496 and 22 to those wells with MRT data, the MRT data may not be representative of groundwater age in this area.

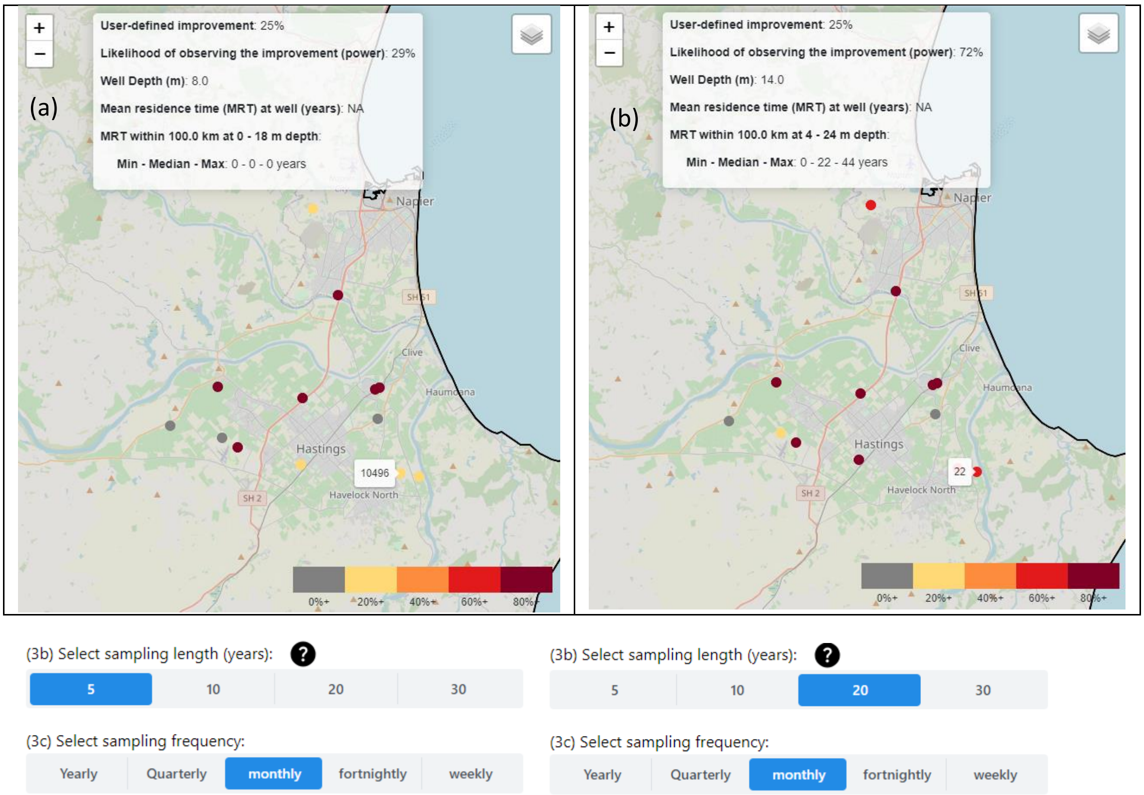


Figure 1: Example of Groundwater Quality interactive map

5.2.3 When to use the web application

The application provides an indication of the following:

1. The maximum probability of correctly detecting a specified concentration reduction at a given sampling frequency and for a given sampling duration/implementation period in existing wells.
2. The sampling length required to successfully detect a concentration reduction at a given frequency in existing wells with a young groundwater age relative to the sampling duration/implementation period (see below).
3. The extent to which the probability of successfully detecting a concentration reduction can be increased by sampling at a higher frequency in wells with a young groundwater age relative to the sampling duration.
4. The relationship between detection power, the rate of nitrate concentration change (i.e., the specified concentration reduction) and sampling duration and the sampling frequency.

Note that the sampling duration and nitrate loss reduction implementation period are assumed to be the same based on the statistical analysis method used. For example, setting a sampling period of 10 years and a nitrate loss reduction rate of 10% means that a reduction rate of 1% per year is assumed. Because the statistical power associated with a 10% reduction over 10 years will be less than that of a 10% reduction over two years, for example, the interactive web application is unlikely to provide a robust power estimate if the expected loss reduction rate is significantly different to the proposed monitoring period. The Groundwater Detection Power repository tools (see below) should be used in these circumstances.

The statistical power for the web application is based on the following methodology:

1. Determine whether the data show an existing trend (e.g., via a Mann-Kendall technique)
2. Estimate the NO_3-N noise for the site.
   - **Where no trend exists:** the NO_3-N noise is assumed to be the standard deviation of the observed data.
   - **Where a trend exists:**
     - Fit a linear regression through all nitrate time series data for each site.
     - Calculate the residuals between the observed data and the linear regression.
     - The NO_3-N noise is assumed to be the standard deviation of these residuals.
3. Define the expected change (e.g., a 20% reduction in concentration) which is then applied linearly to the starting concentration over the observation period (e.g., 10 years)
4. Generate synthetic noise concentration time series data based in the standard deviation derived in step 2 and resample from these data at the specified sampling frequency and duration (e.g., every 3 months for 10 years
5. Add the noise (5.) to the concentration data (4.), conduct a linear regression and evaluate its significance.
6. Repeat steps 5 & 6 many times (10,000) to generate a distribution. The detection power is the percent of the distribution which detects a significant (e.g., $p < 0.05$) linear trend in the right direction (e.g., a reduction).

This method will generate a reasonable estimate of detection power where nitrate concentrations are monotonically increasing, decreasing or broadly stable throughout the monitoring period. For sites with distinct concentration trend epochs (e.g., a 10-year period of stable concentrations followed by a five-year period of increasing concentrations), fitting a linear trend through the whole data series will exaggerate the noise and hence underestimate the statistical power relative to a more nuanced analysis of the data, which might fit the trend line only through the more recent data and hence yield a lower degree of noise. The Groundwater Detection Power repository tools can be used in these circumstances.

The web application cannot be used to estimate the sampling period required to successfully detect a concentration reduction at sites with a high groundwater age relative to the sampling duration. This is because water age (both mean residence time and the age distribution) become an increasingly important determinant of detection power as water age increases relative to sampling duration. By way of example, a sample duration of 10 years is very unlikely to successfully detect a concentration reduction in a well with a mean residence time of 100 years. As a general rule, detection powers derived from this tool where the ratio of MRT:sampling duration is less than:

- 1:10 are **very likely** (> 75%) to overestimate the detection power.
- 1:50 are **likely** (c. 50%) overestimate the detection power.
- 1:100 or more, the tool is **unlikely** (<10%) to overestimate the detection power.

The web application provides an efficient tool for identification of low detection power sites which are unlikely to be useful for mitigation effectiveness monitoring. For example, if a monitoring well is shown to have a 20% chance of detecting a change before lag is accounted for, the well is unlikely to provide useful change detection data unless the linear trend-fitting method described above has significantly overestimated the level of noise in the data. Considering groundwater travel times will only decrease the detection power. The interactive web application is likely to provide useful estimates of detection power (e.g., the detection power is high enough to indicate sampling) only for wells with a low mean residence time (e.g. ≤ 1 year) and a long proposed sampling length (e.g. >20 years) and with monotonically increasing, decreasing or broadly stable nitrate concentrations throughout the monitoring period. Approximately 12% of regional council monitoring wells with age data meet the <1 year criterion.

For monitoring networks where the interactive web application usage limitations above do not meet user needs, the detection power analysis tools provided in the Groundwater Detection Power repository should be used to support monitoring network design/review or optimisation.

5.2.4 How to use the Groundwater Detection Power python tools

The repository provides tools for users who can interact with scripts in the Python programming environment. Usage instructions are provided in the repository as follows:

1. Access and review the historical concentration data and review and potentially remove outliers.
2. If feasible, you may choose to decompose the historical data to remove influences of seasonal/annual/inter-annual cycles, weather events etc. (optional).
3. Ascertain whether or not the historical concentration data has a statistically robust trend (e.g. via a Mann-Kendall test).
4. Estimate the noise in the receptor concentration time series:

a. If the historical concentration data has a statistically robust trend then noise can be estimated as the standard deviation of the residuals from a model (e.g. a linear regression or Sen-slope/ Sen-intercept).

b. If the historical concentration data does not have a statistically robust trend then noise can be estimated as the standard deviation of the receptor concentration time series.

5. Gather data to inform the groundwater age distribution of the site. For instance, a MRT and the parameters for a binary piston flow age distribution model.
6. Estimate the source concentration from the historical trend (if any) and the groundwater age distribution. The following tools are provided to support this process:
   a. the gw_detect_power.truets_from_binary_exp_piston_flow function; or
   b. the gw_age_tools.predict_historical_source_conc function.
7. Define the reduction expected in the source concentration over the implementation period to create a full (past and projected future) concentration time series.
8. Predict the true receptor concentration time series (e.g. the concentration at the receptor if there was no noise) based on the past and future source concentration time series and the groundwater age distribution e.g., using gw_age_tools.predict_future_conc_bepm function.
9. Resample the true receptor concentration time series to the desired sampling frequency and duration. For example, every 3 months for 10 years.
10. Estimate the probability of detecting the change in concentration based on the predicted true receptor concentration time series and the noise in the receptor concentration time series. This can be done using the gw_detect_power.power_calc function and pass your own true receptor time series option.

Importantly, the groundwater detection power tools can assess the detection power of several statistical tests including linear regressions, Mann Kendall tests, and Multipart Mann – Kendall tests. The Multipart Mann-Kendall test is essential for sites with lag and a significant increasing historical trend (Frollini et al., 2021). Briefly, the multipart Mann-Kendall technique identifies all breakpoints in the data where the expected trend (i.e., increasing, then decreasing) is identified by a Mann-Kendall test and is statistically significant ($p<0.05$). This methodology can accurately identify trends where the true receptor concentrations initially increase (as the lagged effects of higher source concentrations continue to move through the groundwater system) before decreasing in response to a NO_3-N loss mitigation. Fitting a simple Mann-Kendall trend in these situations would significantly underestimate the detection power. A two-part Mann Kendall identifies this inflection point without *a priori* information (e.g., the time of the maximum concentration) and is therefore an appropriate analogue to real world change detection. Including multipart Mann-Kendall tests in water quality trend analysis is an essential part of change detection. To that end we have produced a set of python tools to implement this method. Frollini et al. (2021) provide a spreadsheet tool for two-part Mann-Kendall analysis.

5.2.5 Sampling costs

Groundwater sampling costs provided by three regional councils and surface water sampling costs for four regional councils are summarised below. A detailed breakdown of groundwater samplings costs was provided by Environment Southland and Environment Canterbury as per Appendix 1. These costs include vehicle running costs and staff costs but exclude laboratory costs. Analysis costs for nitrate are typically $10-15/sample. A fact sheet on monitoring nitrate in groundwater which includes a discussion of the pros and cons of high frequency sampling is provided in Appendix 2.

Table 5-1 Water sampling costs

Region	Groundwater $/sample	Surface water $/sample
Horizons	250	264
Environment Southland	256	-
Environment Canterbury	130	167
Greater Wellington	-	233
Waikato Regional Council	-	110

6 Case studies

6.1 National SOE network analysis

Dumont et al (2023, in press) assessed the ability of the New Zealand national groundwater monitoring network to detect the nitrate loss mitigations required to meet national policy goals (i.e., the National Policy Statement for Freshwater Management, 2020). The authors found that only 40% of the current network is likely to detect NO_3-N reductions from current concentrations to a 2.4 mg L^{-1} target if reductions are implemented over 30 years and results are assessed after 30 years of quarterly sampling. This timeframe does not meet the requirements of natural resource managers, policy makers and stakeholders, who typically require information on policy and mitigation action effectiveness within 5-10 years. 60% of the network could potentially detect these changes after 30 years with increased monitoring frequency; however, a 100-300% expenditure increase relative to the status quo would be required. Earlier detection (i.e., in 5-10 years) is very unlikely (0-20% of sites), regardless of the sampling frequency, due to lag times. Finally, assessments of less aggressive NO_3-N reductions (1.5% / year) showed even lower detectability. The authors conclude that the current monitoring network is unlikely to be fit for the purpose of detecting NO_3-N reductions within practical timeframes; bespoke change-detection monitoring networks are likely to be required.

6.2 Pokaiwhenua catchment

6.2.1 Summary

Figure 6-1 summarises the change detection monitoring design process and outcomes for the Pokaiwhenua catchment. A detailed illustration of how the design framework can be applied is provided in the subsequent sections.

Step 1. Define mitigation plans & monitoring goals

Proposed Plan Change 1 to the Waikato Regional Plan seeks to reduce contaminants entering the Waikato and Waipā catchments. It includes long term and short-term nitrate concentration targets for three surface water courses in the Pokaiwhenua catchment:

- The Pokaiwhenua Stream: 1 mg/L long-term; a 60% reduction
- Mangamingi Stream: 2.4 mg/L long-term; 0 25% reduction
- Whakauru Stream: 0.26 mg/L long-term; a 90% reduction
- Short term targets are set at 20% of the long-term reduction targets to be achieved within 10 years of PC1 being operative.

Illustrative monitoring goals based on these targets are:

1. Determine whether short-term targets have been achieved within 10 years with 80% confidence.
2. Determine whether progress towards long-term targets has been achieved after 20 years with 80% confidence.

Step 2. Develop a conceptual model

The conceptual model of the monitoring area includes nitrate load distribution, travel paths, attenuation and transit times between sources and receptors.

The Mitigation Effectiveness Monitoring Design web application provides estimates of the maximum plausible nitrate loss reductions that may be achievable as a spatial dataset. These data show low nitrate loss reduction potentials in the upper (southeastern) Pokaiwhenua catchment, which is dominated by exotic and native forest. The reduction potential is high in the lower Pokaiwhenua catchment and upper Whakauru catchment.

Installation of new monitoring sites in sub-catchments with a high reduction potential will maximise the potential to detect the effectiveness of mitigations actions.

Pokaiwhenua Catchment - Mitigation Monitoring Design Recommendations for Nitrate

- Nitrate mitigation monitoring should focus on the Whakauru catchment.
- Monthly monitoring is recommended.
- Further investigation of the upper Whakauru Stream is recommended with a view to installing a new surface water monitoring site, potentially on Poaka Road.
- Detailed records of actions undertaken to reduce nitrate losses in the catchment should be kept to support future understanding of cause and effect.

The dominance of deep groundwater flow paths and long travel times present challenges for early detection of mitigation effectiveness. Mean Transport Times range from 7 to 20 years for surface water sites and 20 to 140 years for the two SOE groundwater monitoring wells.

The potential for nitrate attenuation in parts of the southern (upper) catchment will introduce additional uncertainty for monitoring sites here.

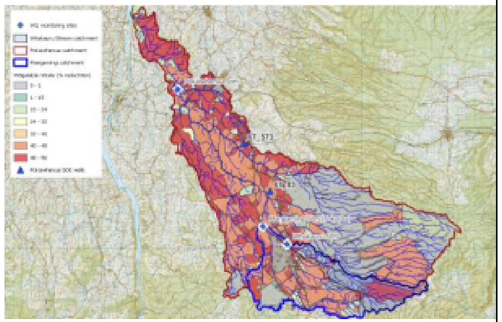
Step 3. Integrated analysis of detection power

High lag times in this catchment and the requirement to explore monitoring durations and nitrate mitigation implementation periods in accordance with the regional plan mean that the interactive web application will not provide accurate estimates of change detection power for this catchment. The Groundwater Detection Power repository tools were used to evaluate the detection power of two SOE groundwater monitoring wells and three surface water monitoring sites.

The detection power analysis results show that monitoring data from Well 67_83 and the Whakauru Stream are likely to provide much earlier indications of mitigation effectiveness than the other monitoring sites.

Step 4. Evaluate representativeness of priority sites

The recharge area for Well 67_83 would need to be determined to assess the extent to which monitoring data are likely to be representative of land management actions to reduce nitrate across the catchment. The Whakauru Stream catchment is more easily defined and covers a much larger area and is therefore likely to provide more representative information on mitigation effectiveness at scale. The Whakauru Stream should therefore be the focus of mitigation effectiveness monitoring and evaluation.



Step 5. Identify new monitoring sites (if required)

Further investigation of the upper Whakauru Stream is recommended with a view to installing a new surface water monitoring site, potentially on Poaka Road, to maximise the likelihood of determining mitigation effectiveness. The investigation could include synoptic nitrate sampling under a range of stream flows and collection of age tracer samples from one or more locations in the upper stream catchment.

Figure 6-1 Monitoring design for the Pokaiwhenua catchment

6.2.2 Introduction

The purpose of this case study is to illustrate the application of steps 1-5 of the monitoring design framework presented in Section 5.1 using the Pokaiwhenua catchment as an example.

The Pokaiwhenua catchment is located in the southern Waikato region to the east of Rotorua and immediately north of Tokoroa as per Figure 6-7. The catchment is drained by a series of northerly and north westerly draining watercourses which discharge to the Pokaiwhenua stream. The Pokaiwhenua stream issues from the hills of the Mamaku Plateau in the southeast of the catchment and discharges to Lake Karapiro at the northern terminus of the catchment. Lake Karapiro is a hydro lake on the Waikato River. Land elevation ranges from 700 m asl on the Mamaku Plateau in the southeast to 100 m asl at the Lake Karapiro discharge in the north.

6.2.3 Step 1: Define mitigation plans/scenarios and monitoring goals

Waikato Regional Council routinely monitors groundwater quality at two SOE sites in the Pokaiwhenua catchment and surface water quality in the Pokaiwhenua Stream at Puketurua, the Whakauru Stream and the Mangamingi Stream as per Figure 6-7 D. Time series monitoring data are plotted in Figure 6-2. Surface water nitrate data show seasonal variability with peak concentrations generally in September/October and seasonal lows typically in April/May. A sharp increase in nitrate concentrations since c. 2008 is evident in the Whakauru Stream.

Proposed Plan Change 1 to the Waikato Regional Plan seeks to reduce contaminants entering the Waikato and Waipā catchments and includes long term (80-year) and short term (10-year) nitrate concentration targets for three surface water courses in the Pokaiwhenua catchment: The Pokaiwhenua Stream (1 mg/L long-term; a 60% reduction from recently measured concentrations), Mangamingi Stream (2.4 mg/L long-term; 0 25% reduction from recently measured concentrations) and Whakauru Stream (0.26 mg/L long-term; a 90% reduction from recently measured concentrations). Short term targets are set at 20% of the long-term reduction targets to be achieved within 10 years of PC1 being operative. Assumed monitoring goals based on these targets are:

- Determine whether short-term targets have been achieved within 10 years with 80% confidence.
- Determine whether progress towards long-term targets has been achieved after 20 years with 80% confidence.

We have assumed that the reductions required to achieve short-term targets will be implemented within two years and that the reductions required for the long-term targets will be implemented incrementally after 10 years.

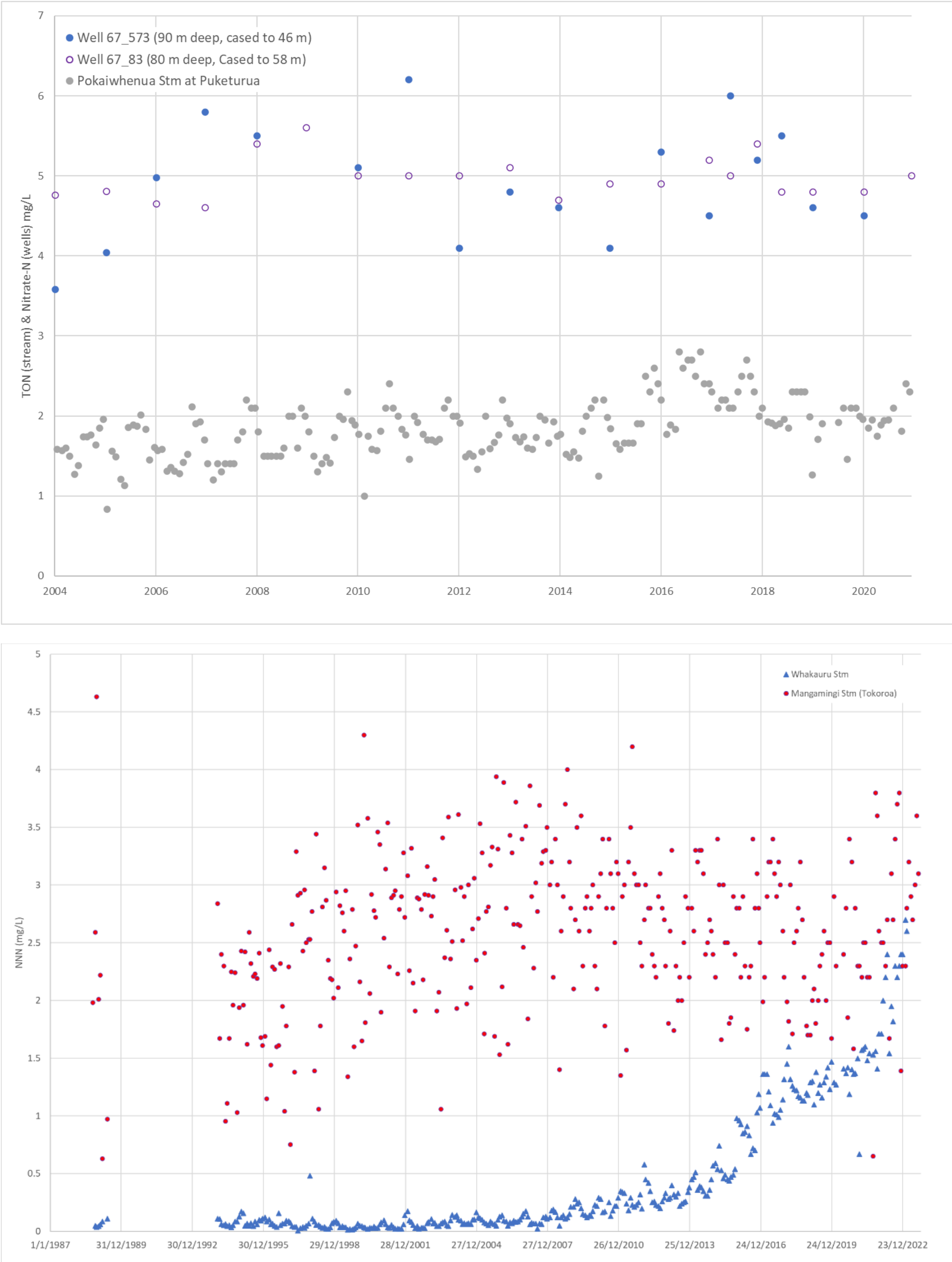


Figure 6-2 Nitrogen concentration time series data

6.2.4 Step 2: Develop a conceptual model of the monitoring area

Nitrate sources

The Mitigation Effectiveness Monitoring Design web application provides estimates of the maximum plausible nitrate loss reductions that may be achievable as a spatial dataset. These data show low nitrate loss reduction potentials in the upper (southeastern) Pokaiwhenua catchment (Figure 6-7 D), which is dominated by exotic and native forest. The reduction potential is high in the lower Pokaiwhenua catchment and upper Whakauru catchment. Installation of new monitoring sites in sub-catchments with a high reduction potential will maximise the potential to detect the effectiveness of mitigations actions.

Hydrology and hydrogeology

The catchment geology is dominated by sandy or gravelly highly porous pumice soils developed on young volcanic deposits (largely from the Taupo eruption 1.8 ka BP). Podzols occur at greater elevation in forested areas characterised by older volcanic deposits and higher rainfall. Mean annual rainfall in the Upper Waikato River catchment is predominantly between 1500 and 1600 mm (Woodward & Stenger, 2018).

Groundwater flow is generally expected to mirror surface topography and hence the main groundwater flow paths in the catchment are likely to be to the west and northwest from the Mamaku Plateau in the southeastern part of the catchment and north eastwards in the northern part of the site.

Lincoln Agritech investigated the Pokaiwhenua catchment as part of the MBIE-funded Transfer Pathways programme (Wilson, 2018), summary findings are as follows:

- The aim of this study was to determine and model the main pathway contributions in the Pokaiwhenua catchment (432 km²) in the Upper Waikato, where substantial pine-to-pasture conversions have recently occurred.
- Principal components analysis (PCA) indicated a predominance of groundwater recharge on the Mamaku Plateau, deep circulation within the Whakamaru ignimbrite, and fault-induced emergence south of Putāruru. Mamaku Plateau recharge accounts for 60% of stream flow at the catchment outlet, whereas 40% is sourced from the more widespread Whakamaru ignimbrite. The proportional contribution changes towards the catchment headwaters, with 90% of the water at the Whakauru monitoring site being sourced from the Mamaku Plateau.
- The study shows that the headwater streams are most vulnerable and quickly respond to local nitrate leaching since they receive less low-nitrate groundwater from the Whakamaru ignimbrite.

Modelled relative contributions to total streamflow and nitrogen loads in the Pokaiwhenua Stream at Puketurua in Figure 6-3 below indicate that ~60% of the flow and ~50% of the current nitrogen load are derived from the deep groundwater flow path. The flow and load contributions from near surface flow paths are small (~15% and <10% respectively).

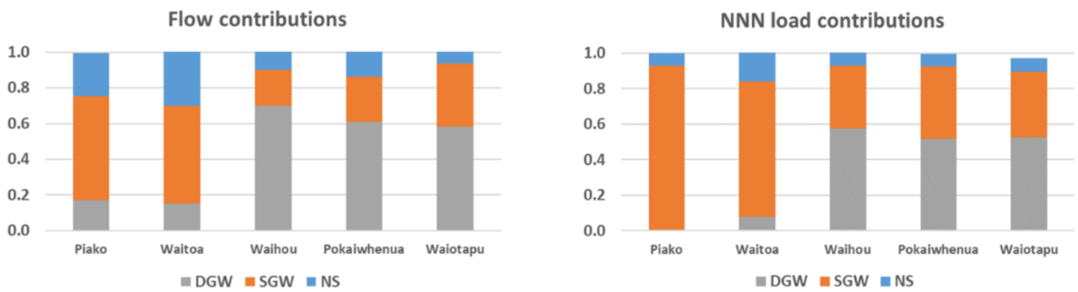


Figure 6-3 15-year averages of the relative contributions of the three pathways to water flow (left) and NNN load (right) (from Stenger et al., 2022)

Note: Near-surface (NS) pathways, Shallow groundwater (SGW), Deep groundwater (DGW)

Transport times and lags

Stenger et al. (2022) collected five tritium samples from the Pokaiwhenua Stream at the Puketurua monitoring site across a range of flows as per Figure 6-4. These data were used in conjunction with the Bayesian chemistry-assisted hydrograph separation method (Woodward & Stenger, 2018) to model the mean transit times for the three flow path end members (deep groundwater, shallow groundwater and near surface). Figure 6-4 shows the distribution of the samples across the flow duration curve for the Pokaiwhenua Stream. Figure 6-5 plots the modelled mean transport time of water in the stream over a one-year period in conjunction with the modelled flow path end member contributions to total stream flow. The results indicate that Mean Transport Times (MTTs) to the stream fall within the 15-35 year range for a significant proportion of the time. MTT data were also obtained by Stenger for the Whakauru and Mangamingi catchments based on three age tritium samples collected under a range of flows. MTTs ranged from 0.3 to 12 years (mean = 8 years) for the Whakauru stream and 0.2 – 10 years (mean = 7 years) for the Mangamingi Stream.

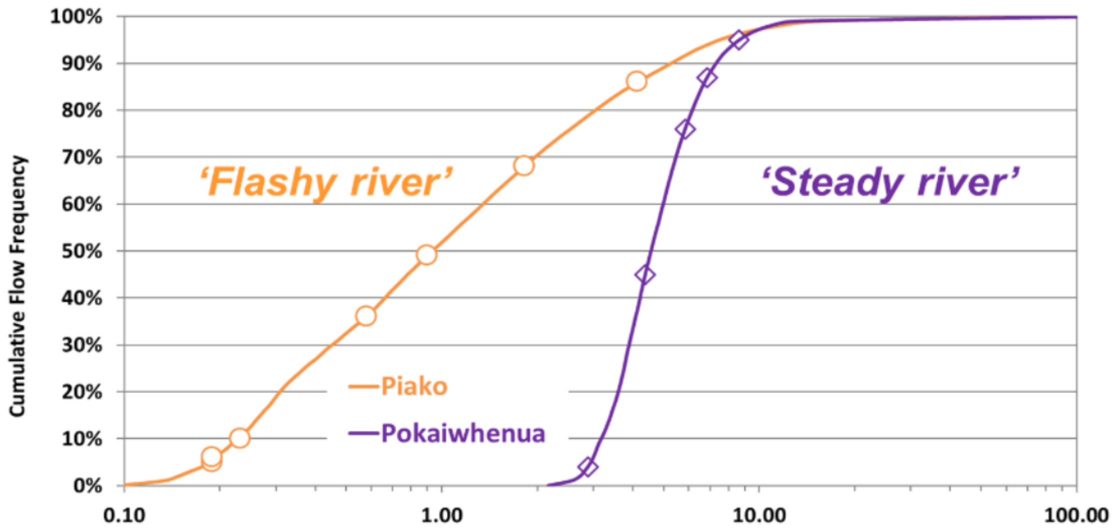


Figure 6-4 **Flow duration curves for the Piako River and Pokaiwhenua River. Tritium-samplings indicated by symbols on graphs (from Stenger et al., 2022)**

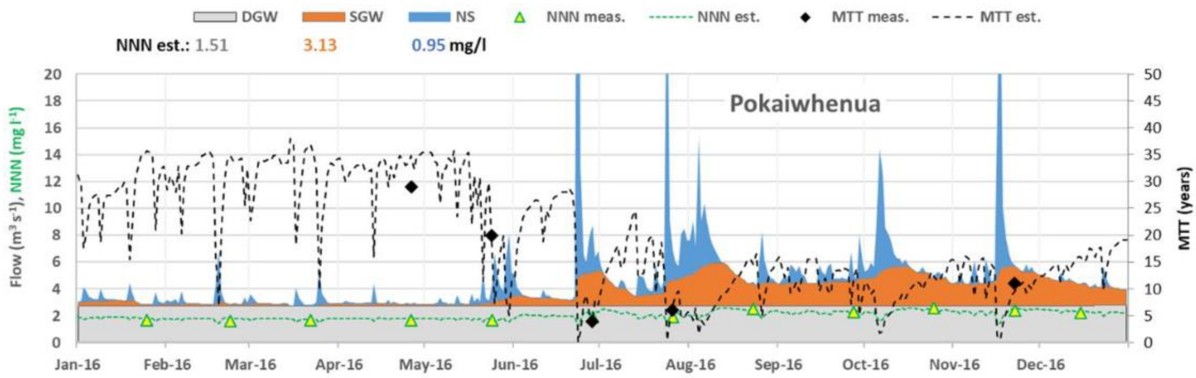


Figure 6-5 **Estimated pathway contributions, MTTs, and NNN concentrations in Pokaiwhenua River during 2016 (from Stenger et al., 2022)**

Note: Black diamond symbols represent MTTs calculated from tritium samples, yellow triangles measured NNN concentrations.

Groundwater age tracer interpretation results provided by Waikato Regional Council for the two SOE groundwater monitoring wells summarised in Table 6-1 give mean transport (or mean residence) times of 142 years and 22 years for tritium. The exponential piston flow model with 90% exponential mixing was used for the age interpretation. Figure 6-6 plots modelled groundwater age as a cumulative density function for the tritium age for both wells using both an Exponential Piston Model (EPM) with 90% exponential flow and a Dual Exponential Piston Model (DEPM) with a 50% young fraction (MTT = 30 years and 2 years) and 50% old fraction (MTT = 254 years and 30 years) for well 37_573 and 67_83 respectively. These ages and fractions give 142 year and 22 year aggregated MTTs to match with the tritium age.

Table 6-1 **Age tracer mean transport times for SOE wells**

Bore Id	Bore depth (m)	Base of casing (m)	CFC-11 age (years)	CFC-12 age (years)	SF6 age (years)	Tritium age (years)
67_573	90	46	99	78	15	142
67_83	80	58			7	22

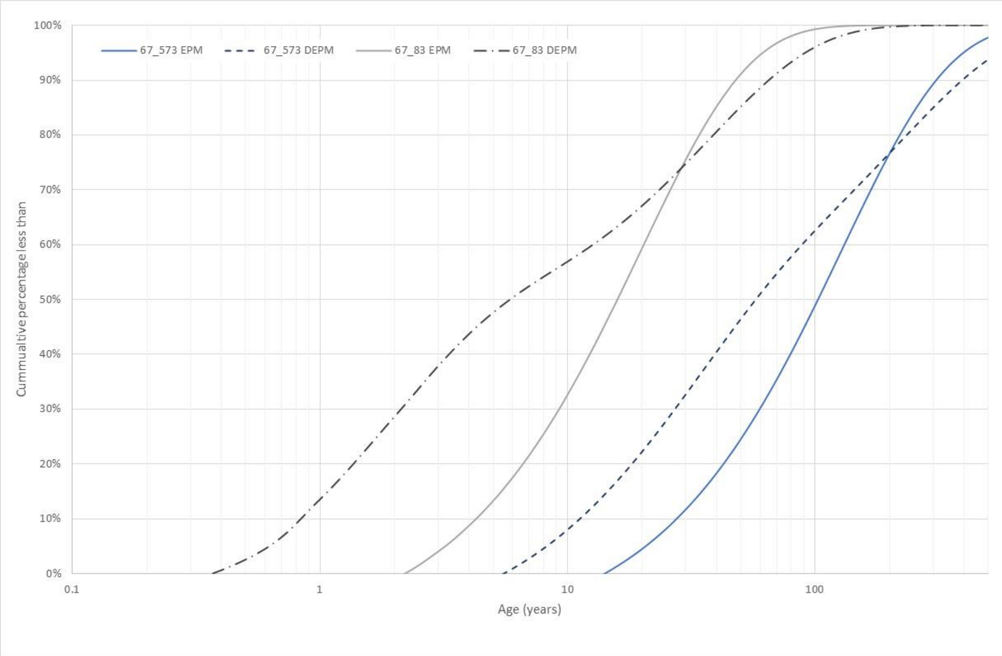


Figure 6-6 Model age distributions for EPM and DEPM for Pokaiwhenua SOE wells

Figure 6-7 C plots depth to groundwater data from the Waikato Regional Council database together with modelled vadose zone transport times (Wilson & Shokri, 2015). The depth to groundwater is significant for much of the catchment with an average depth of 40 m and a 10th – 90th percentile range of 8 – 74 m. The data generally show deep groundwater and long vadose zone transport times (36 - >100 years) for the Mamaku Plateau. Shallower groundwater and shorter modelled vadose zone transport times occur in the vicinity of parts of the stream network in the mid and lower catchment.

Attenuation processes

Understanding the spatial distribution of nitrate attenuation in the groundwater system, for which redox condition is the key indicator (Wilson et al., 2020), is a key factor in mitigation monitoring design. National-scale redox modelling results for the Pokaiwhenua catchment (Figure 6-7 C) show anoxic and mixed redox groundwater in parts of the southern catchment. Spot readings of groundwater nitrate are very low in this area, which may signify nitrate attenuation in this area of dairy land use. Change detection monitoring in areas with variable redox conditions is more likely to yield ambiguous results or a higher probability of statistical error. Monitoring in areas of reduced groundwater could also lead to interpretation challenges unless the rate of nitrate attenuation is known to be constant, regardless of groundwater nitrate concentrations.

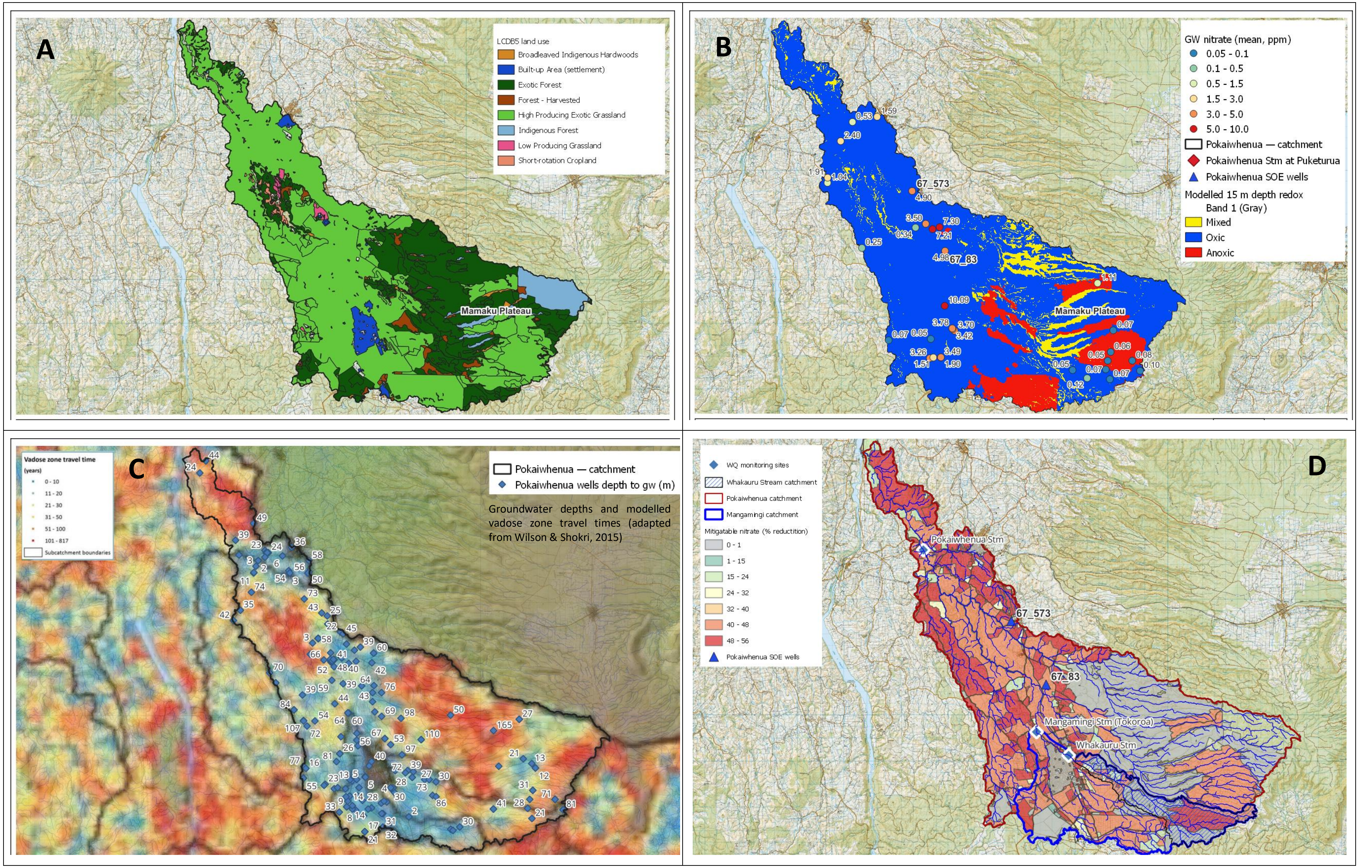


Figure 6-7 Pokaiwhenua catchment environmental data (A = Land use, B = Redox, C = Vadose lag & groundwater depth, D = nitrate monitoring sites & mitigatable N load)

6.2.5 Step 3: Integrated groundwater and surface water detection power analysis

The high lag times in this catchment and the requirement to explore monitoring durations and nitrate mitigation implementation periods in accordance with the regional plan mean that the interactive web application will not provide viable estimates of change detection power for nitrate mitigation for this catchment. The Groundwater Detection Power repository tools were therefore used to evaluate the detection power of the two SOE groundwater monitoring wells and three surface water monitoring sites using the groundwater age data summarised above. The results summarised in Figure 6-8 below show that expected effects of the nitrate loss reductions required under the regional plan should be detected with 80% certainty:

- By 2040 with ≥ biannual sampling in well 67_83
- By 2075 with ≥ monthly sampling in well 67_573
- Not before 2100 with monthly sampling in the Pokaiwhenua Stream
- By 2070 with ≥ monthly sampling in the Mangamingi Stream
- By 2040 with ≥ quarterly sampling in the Whakauru Stream

Detection power does not increase significantly in the Whakauru Stream for sampling frequencies greater than quarterly. Monthly sampling is recommended, however, to provide additional information on short- and medium-term variability because this will support a better understanding of the hydrological system and change processes.

6.2.6 Step 4: Evaluate representativeness of priority monitoring sites

The detection power analysis results above show that monitoring data from well 67_83 and the Whakauru Stream are likely to provide much earlier indications of mitigation effectiveness than the other sites. Determination of the recharge area of well 67_83 would be required to support interpretation of the monitoring results from this site. The catchment area for the Whakauru Stream is both more easily defined and encapsulates a much larger area than the monitoring well recharge zone. This larger sampling area equates to a lower probability of statistical error in mitigation effectiveness determination from the monitoring results. We therefore recommend that mitigation effectiveness monitoring and evaluation should focus on the Whakauru Stream.

6.2.7 Step 5: Identify new monitoring sites as required

The conceptual model indicates that the upper Whakauru catchment is characterised by a high mitigatable nitrate load (~ 50%) and oxic groundwater, both of which are favourable for detection of mitigation effectiveness monitoring. It is possible that a significant proportion of the nitrate loss reductions required to meet the regional plan targets will need to occur in the upper catchment.

Groundwater level data for the upper catchment indicate either that the water table is very deep (20 – 40 m) or that a steep downwards groundwater gradient is present here. The latter appears to be more likely given that the Whakauru is groundwater-fed and the local wells with groundwater level observations are cased to > 60-100 m depth. Further investigation of the upper Whakauru Stream is recommended with a view to installing a new surface water monitoring site, potentially on Poaka Road, to maximise the likelihood of determining mitigation effectiveness. The investigation could include synoptic nitrate sampling under a range of stream flows and collection of age tracer samples from one or more locations in the upper stream catchment.

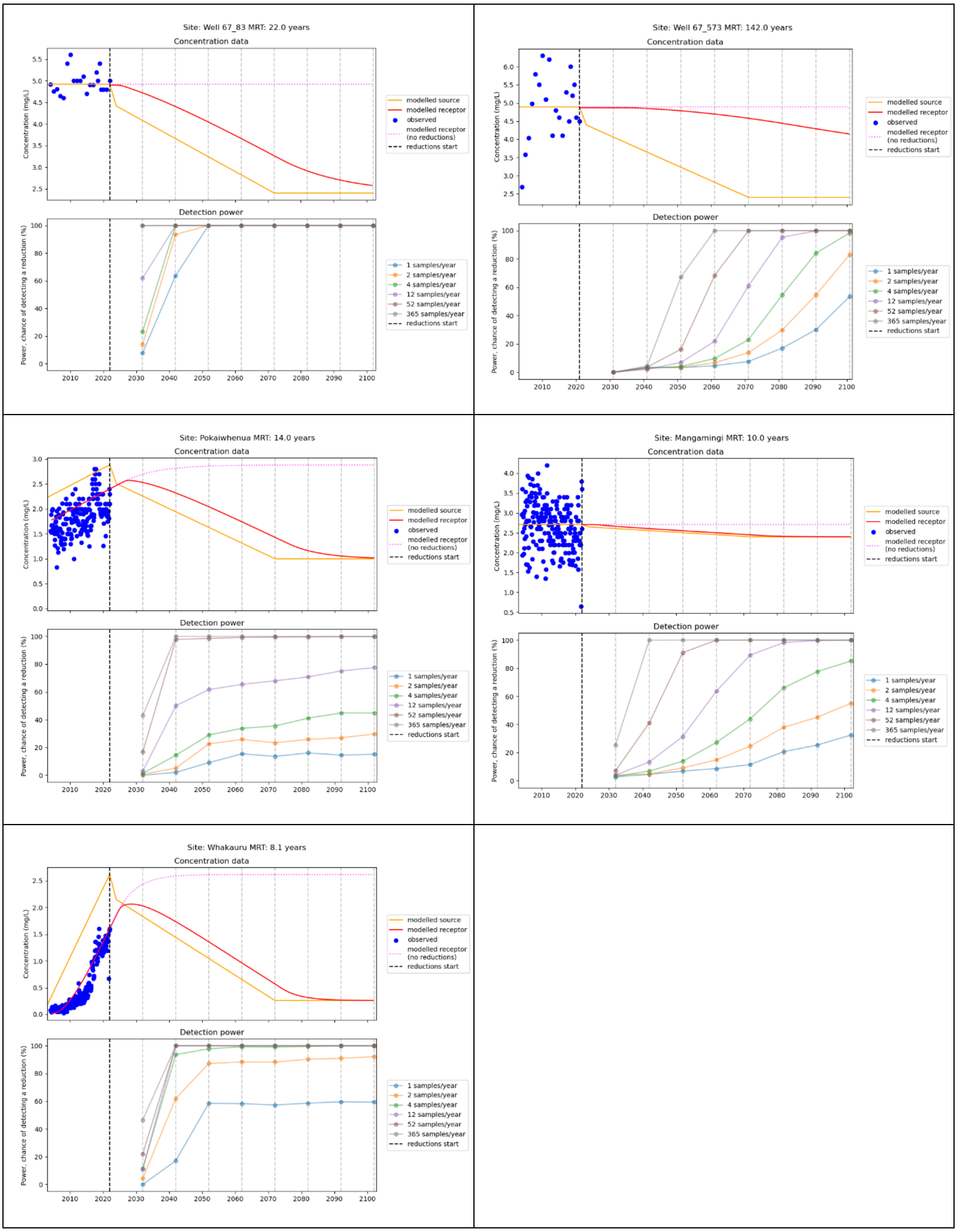


Figure 6-8 Pokaiwhenua catchment monitoring site detection power

6.3 Te Hoiere

6.3.1 Introduction

The purpose of this case study is to illustrate the application of steps 1-5 of the Monitoring design framework presented in Section 5.1 in the Te Hoiere catchment.

The Te Hoiere/Pelorus river is the largest catchment draining into the Marlborough sounds and covers 107,403 km². The Te Hoiere/Pelorus catchment is composed of nine sub-catchments with many rivers and streams that flow into the Motuweka/Havelock and Māhakipaoa estuaries, and ultimately into the Te Hoiere/Pelorus sound. Most of the land cover in the catchment consists of indigenous forest, production forestry and pasture. The soils in Te Hoiere consist largely of clay (up to 60 %) and are considered highly erodible. The geology of the valleys consists of alluvial sediments and greywacke rock.

Currently, Marlborough District Council, Ngāti Kuia, the Department of Conservation and the wider community are in the process of designing a catchment restoration programme in the Te Hoiere/Pelorus Catchment, with the aim to holistically manage the entire catchment from the mountains to the sea (ki uta ki tai) (Morgenstern & Davidson, 2022). Our case study focuses on the Rai River sub-catchment as shown in Figure 6-10.

6.3.2 Step 1: Define mitigation plans/scenarios and monitoring goals

Although the current water quality of Te Hoiere Catchment is relatively good, deterioration is evident in the form of increasing contaminant levels in some of the sub-catchments. Nitrate concentrations in Rai River at the Rai Falls monitoring site have increased by about 40% over the last 10 years of monitoring (Figure 6-9) due to land- use intensification and nitrate concentrations in monitoring Well 10323 near Rai Valley township doubled in 2015 to ~70% of the maximum acceptable value of 11.3 mg/L nitrate-nitrogen (NO_3-N) of the New Zealand drinking water standard (Morgenstern & Davidson, 2022).

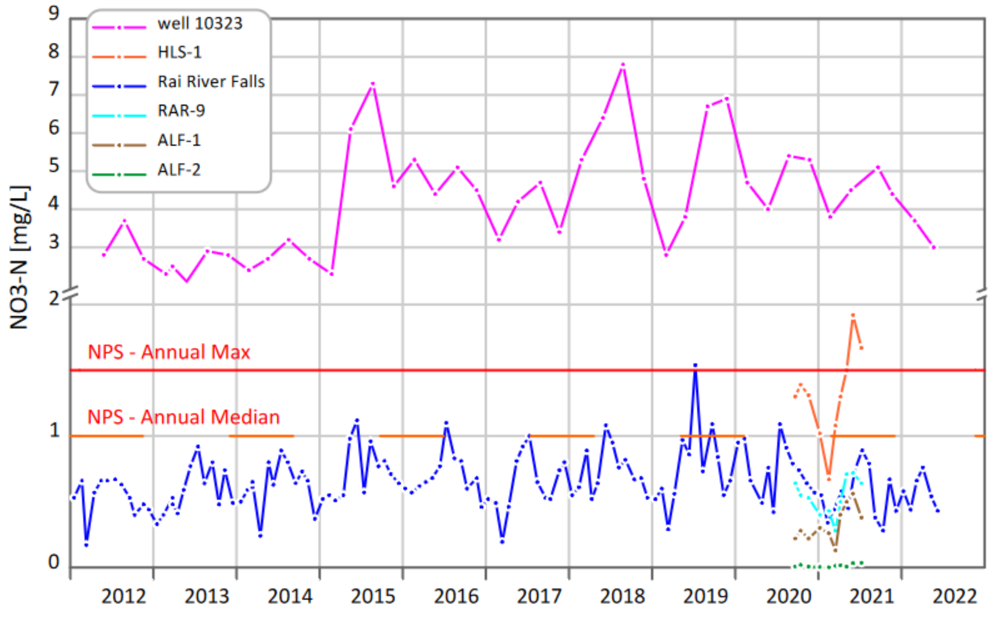


Figure 6-9 Time series of nitrate-nitrogen in groundwater and surface waters near Rai Valley township (from Morgenstern & Davidson, 2022).

Because nitrogen loss reduction targets have not been defined for the catchment, we have assumed for demonstration purposes that a 100% reduction in the maximum mitigatable nitrogen load will be implemented over a 10-year period. This equates to an 18% reduction in the Rai River at Rai Falls, 14% in the Ronga River upstream of the Rai River confluence and 16% in the upper Opouri River at Tunakino Valley Rd. A 20% nitrate concentration reduction was assumed in the Well 10323 recharge zone.

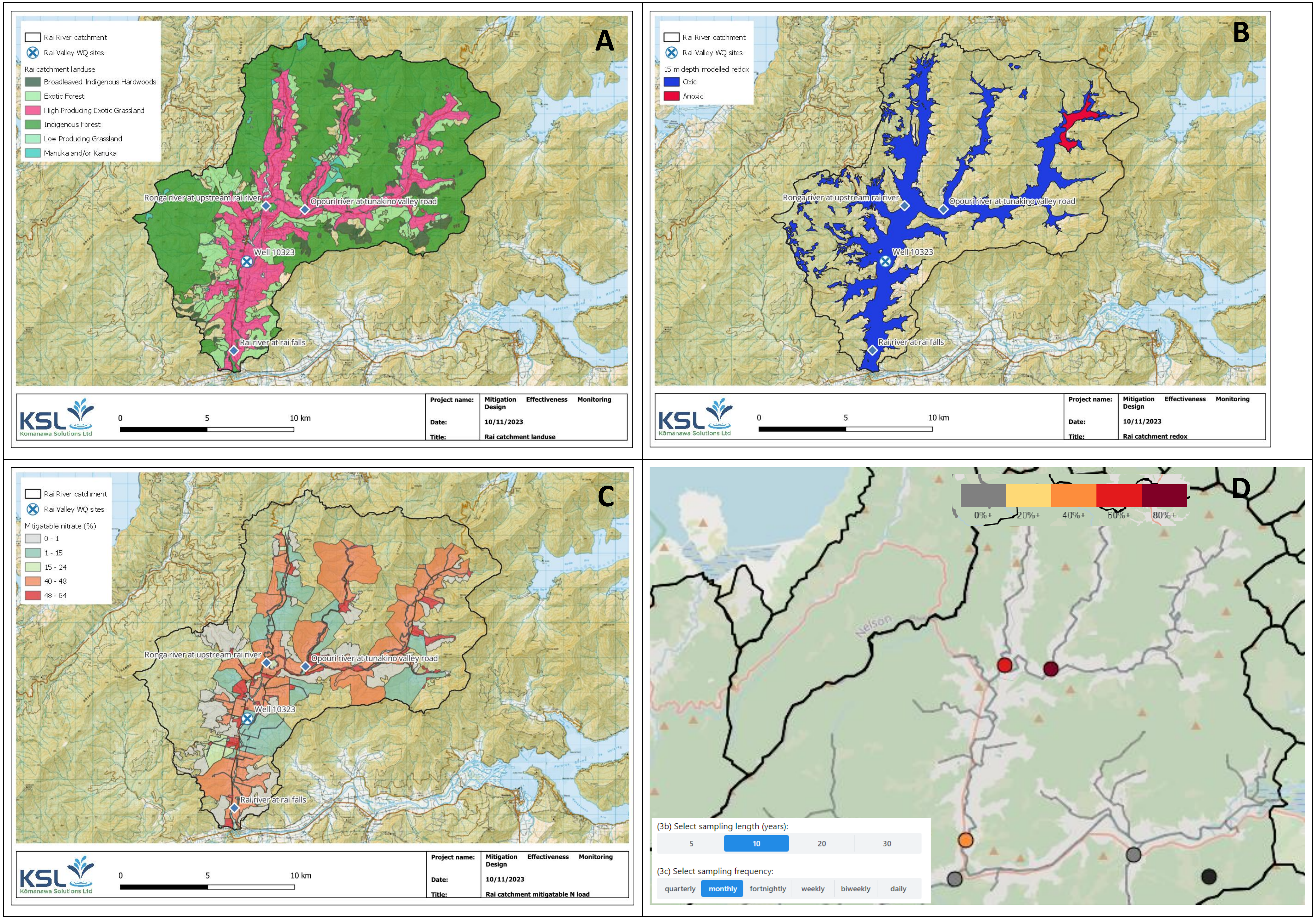


Figure 6-10 Rai catchment environmental data (A = Land use, B = Redox, C = Mitigatable total N load, D = web application detection power results)

6.3.3 Step 2: Develop a conceptual model of the monitoring area

Nitrate sources

Land use in the Rai River catchment predominantly comprises exotic pasture, indigenous forest and exotic (pine) forest as per Figure 6-10 A. Mitigatable nitrate losses are high (~45%) for the exotic pasture in the valley floors and low elsewhere. Note that the orange shading (40-48% mitigatable nitrogen) shown on some valley sides where land use is mapped as indigenous forest are likely to relate to errors due to mapping/modelling resolution.

Hydrology and hydrogeology

Information on the hydrology and hydrogeology of the Rai River catchment is provided in Morgenstern & Davidson (2022) as follows:

- Quaternary alluvium, consisting of poorly consolidated gravel to silt, has been deposited adjacent to all major rivers and streams in the Te Hoiere catchment. Fan deposits are common below steep hillslopes, and these merge with the valley floors.
- Shallow monitoring wells are likely to capture groundwater from seasonally changing sources, for example, very young water from surrounding areas in winter versus older groundwater from the deeper groundwater flow from further up in the catchment in late summer.
- High surface-water radon concentrations were observed in the mid reaches of the Rai River and Ronga River at the Rai River confluence, where extensive side valleys join the Rai River valley via Holocene gravel deposits. This indicates large groundwater discharges into the rivers at these sites, including return flows of stream water lost to the Holocene gravel upstream.

Transport times and lags

Tritium based mean transit times of the water overall are less than four years in the Te Hoiere / Pelorus Catchment, and the majority of nitrate is flushed into the river very quickly via the winter rain pulses (Morgenstern & Davidson, 2022). Mean Residence Times of 2.5 and 3.0 years are reported by the authors for Well 10323 and the Rai River at falls monitoring site respectively.

Attenuation processes

Active groundwater flow was observed in the Rai Valley through anoxic zones, with potential to remove significant fractions of nitrate through denitrification before the groundwater discharges into the river (Morgenstern & Davidson, 2022). Nationally modelled redox data for the Rai Valley (Figure 6-9 C) indicates that anoxic conditions are restricted to the upper Opouri River, with the remainder of the Rai catchment groundwater being oxic. Nitrate attenuation is therefore unlikely to play a significant role in the catchment overall, but any proposed monitoring in the upper Opouri River catchment should account for potential attenuation.

6.3.4 Step 3: Integrated of groundwater and surface water detection power analysis

Detection power analysis was undertaken with both the Mitigation Effectiveness Monitoring Design web application (see Figure 6-10 D) and the Groundwater Detection Power repository (Figure 6-11); results are summarised in Table 6-2.

Table 6-2 Detection power (%) after 10 years of monthly monitoring

Site	Assumed N loss reduction %	Web app	Detection power repository - no lag	Detection power repository - with lag
Rai River	18	79	40	22
Ronga River	14	45	35	22
Opouri River	16	90	50	10
Well 10323	20	7	65	15

The results show the following:

- Comparing the Groundwater Detection Power Repository - *no lag* and - *with lag* results shows that, despite the relatively small Mean Residence Times at these sites, accounting for transit times and age dispersion is important for detection power analysis with a 10-year monitoring duration. The differences become insignificant after 20 years of monitoring[4].
- The web application shows significantly higher detection powers than the Groundwater Detection Power Repository tool results. This relates to the underlying methodology: the web application results are based on analysis of a national dataset which necessarily uses the same approach for each site. The Groundwater Detection Power Repository results are more tailored to the individual sites and therefore provide a more reliable indication of detection power.
- The Groundwater Detection Power Repository - *with lag* results show that none of the sites are likely to reliably detect nitrate loss mitigation effectiveness within 10 years. Detection with 80% confidence would be expected after 20 years of monthly sampling at all sites.
- The analysis highlights the importance of accounting for lags in detection power analysis for both surface and groundwater sites where the monitoring duration/change detection period of interest is relatively short (e.g. 10 years), even where MRTs are small.
- The Rai River and Ronga River provide the highest detection power of the four monitoring sites we evaluated. Increasing the Rai River sampling frequency from monthly to weekly would increase the detection power after 10 years of sampling from 20% to 70%, suggesting that higher sampling frequency would be valuable if nitrate loss mitigations are proposed in these catchments. Monthly sampling would provide an ~80% detection power after 20 years of sampling and hence the benefits of higher frequency sampling would reduce over time following implementation of nitrate loss mitigations.

6.3.5 Step 4: Evaluate representativeness of priority monitoring sites

Although the recharge zone for Well 10323 is unknown, the spatial extent of land influencing nitrate concentrations at this site will be very small relative to the surface water monitoring sites. The detection power of the well is similar to the surface water sites and hence the surface water sites should be the target of any increase in monitoring frequency.

6.3.6 Step 5: Identify new monitoring sites as required

The information reviewed here does not suggest that additional monitoring sites are required. Adding more groundwater monitoring sites would be unlikely to improve the change detection power already available via the surface water monitoring network. Consideration could be given to the installation of monitoring wells with identifiable catchment areas under an experimental design framework (targeting areas where loss mitigations are planned), with the goal of evaluating mitigation effectiveness more quickly. However, although the costs and benefits of this would need to be considered in conjunction with the alternative option of higher frequency surface water sampling, we anticipate that the latter would likely emerge as the optimal approach.

[4] Some results, e.g. the Ronga River show reducing detection power with increasing monitoring duration in the no lag results because the datapoints affected by the load change become a smaller component of the larger dataset generated over this period.

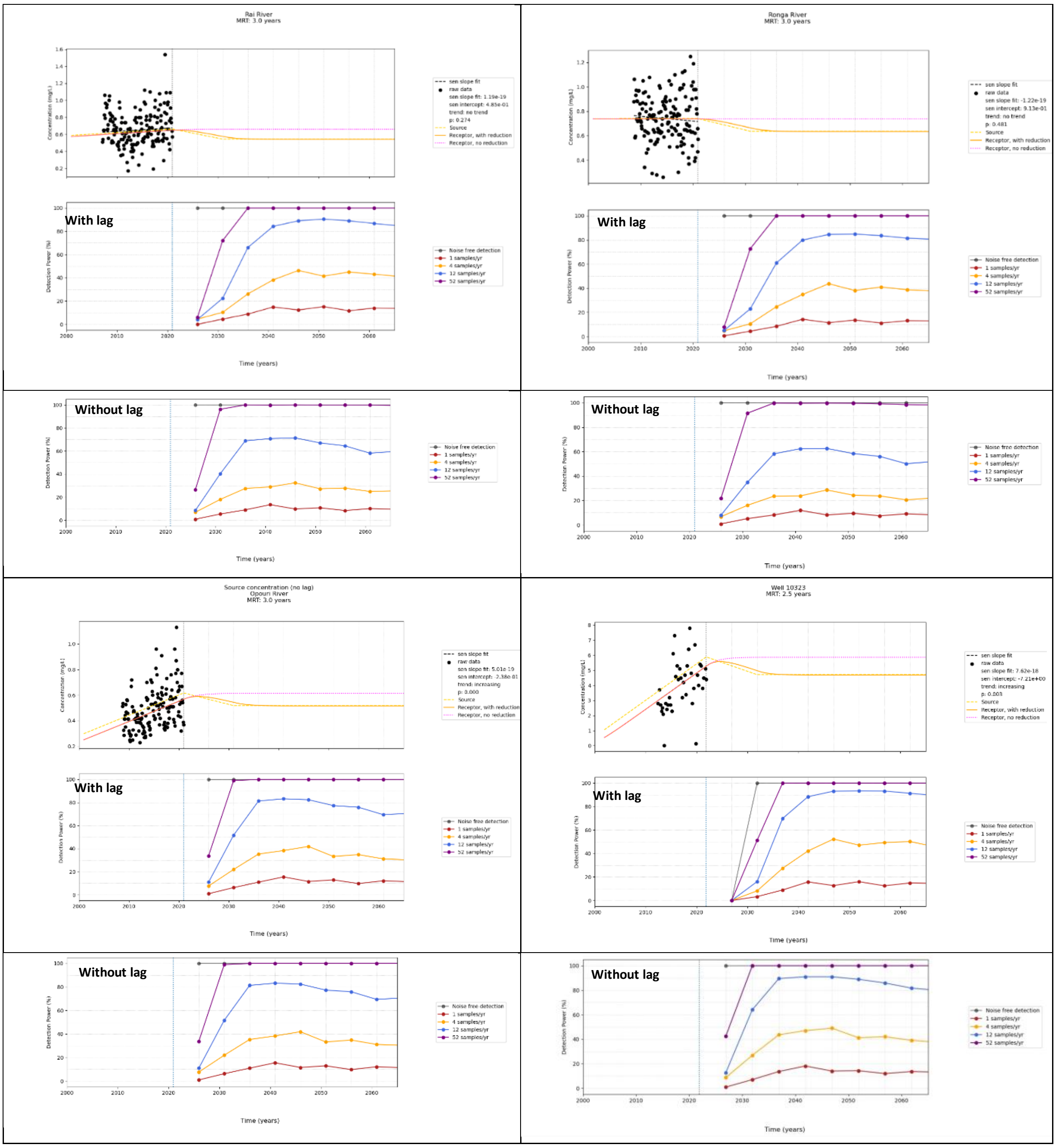


Figure 6-11 Nitrate change detection power analysis for Te Hoiere monitoring sites

7 Conclusions

Early and robust detection of nitrate management policy and mitigation action effectiveness is vital given the high cost of mitigations, the associated requirement for long term financial planning and the goal of achieving water quality improvements within five years. The key requirements for change detection are statistical power, representativeness, and responsiveness.

The success of any scientific study or monitoring program relies on its ability to accurately measure the desired attribute or relationship. Long-term monitoring programmes are typically designed to track changes over time, but insufficient statistical power can lead to failed monitoring programme objectives and wasted resources.

If the monitoring program detects a trend when none is present (Type I error), fails to detect a real trend (Type II error), or estimates a trend that is opposite to the one present (Type III error), any management decisions based on the monitoring results could undermine rather than support the management objectives.

Statistical power analysis allows us to determine the sample size required to detect an effect of a given size (e.g., a load change) with a specified degree of confidence. Conversely, it can be used to determine the probability of detecting an effect of a given size with a given level of confidence for a specified sampling frequency and duration. Although scientists and statisticians have long recommended conducting a power analysis to inform robust study design, and determination of trends is often listed as a key monitoring network design objective, effective implementation of this objective in network design appears to be rare.

Groundwater quality monitoring network design and review processes generally focus on the representativeness of the monitoring network in terms of geographic spread and hydrogeological and/or water quality typologies. Integrated analysis of surface and groundwater monitoring networks to optimise change detection monitoring appears to be rare. The outcome of the current approach is network optimisation bias towards broad scale water quality characterisation, with little or no consideration of whether the network will yield robust change detection information within the timeframes required by land managers, custodians and regulators.

High spatial variance in nitrate leachate concentrations and loss reductions coupled with the relatively small capture zones of typical monitoring wells means that an impractically large number of wells will often be required to obtain a representative network. Drawing conclusions from a monitoring network without understanding its representativeness could result in statistical error and poor management decisions.

Determination of the lag time between a nitrogen input change (e.g. implementation of a mitigation programme) and the associated response at a monitoring site is a pre-requisite for determination of monitoring duration. Our analysis shows that inclusion of lag and age dispersion in statistical power analysis is essential for most groundwater sites and possibly for many surface water sites, especially where the change detection monitoring period is relatively short, e.g. 10 years or less. Mean Residence Time data are currently only available for approximately 20% of the groundwater nitrate monitoring sites we analysed nationally, and even fewer surface water sites.

A monitoring design framework and accompanying toolkit are presented in this report. The framework provides a step-by-step guide for review and optimisation of the change detection power of existing monitoring networks and is demonstrated via two case studies. The toolkit includes a national interactive web application which provides an initial screening of the change detection power for existing monitoring sites. A Groundwater Detection Power Repository (which can also be used for surface water sites) has been developed to support site-by-site change detection power analysis using more nuanced analysis methods, accounting for lag and age dispersion. Sampling cost information is also provided.

Applying the Groundwater Detection Power Repository tools to all monitoring wells in New Zealand's SOE groundwater network wells with current nitrate-N concentrations >2.4 mg/L shows that early detection of hypothetical nitrate loss reductions to achieve 2.4 mg/L within 30 years would be very unlikely at most sites. The analysis concludes that the current monitoring network is unlikely to be fit for the purpose of detecting NO_3-N reductions of this magnitude within practical timeframes.

Identification of a targeted set of monitoring sites with identifiable catchment areas, short lag times and high detection power under an experimental design framework may be required to determine policy and/or mitigation action effectiveness in many instances. Integrated design of surface water and groundwater quality networks is also required to optimise monitoring efficiency and change detection power.

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Appendices

Appendix 1. Sampling cost breakdown

Table 8-1 Environment Southland sampling cost breakdown

Activity	Cost per site per year	Cost per site per visit	Assumptions
Vehicle	-	-	Vehicles used across all monitoring projects
Field meter	$67.71	$16.93	Hard to define as the field meters are used across multiple sampling projects. Assume one field meter for this project out of the 7 that we run.
Mileage	$150.28	$37.57	Based on $0.687 per km
Consumables (meter)	$56.25	$14.06	Hard to define as the consumables are used across multiple sampling projects. Assume 2L of pH 4,7,10 and 1L 1413, 148 condy buffer per quarterly sampling
PPE	-	-	PPE used across all projects
Staff time (Sampling + Prep)	$623	$155.86	Average charge out rate $95hr. Assuming one person per sampling run. In reality there is often more than one for training purposes.
Staff Time (Data)	$47.50	$11.88	Average charge out rate $95hr. 4hrs/32 = 0.125hrs per site
Upkeep of field meters etc.	$78.13	$19.53	Hard to define as the field meters are used across multiple sampling projects. Assume one field meter for this project out of the 7 that we run.
Total	**$1,023.30**	**$255.83**	**32 sites sampled quarterly on 7 sampling runs - 7.5hrs per run**

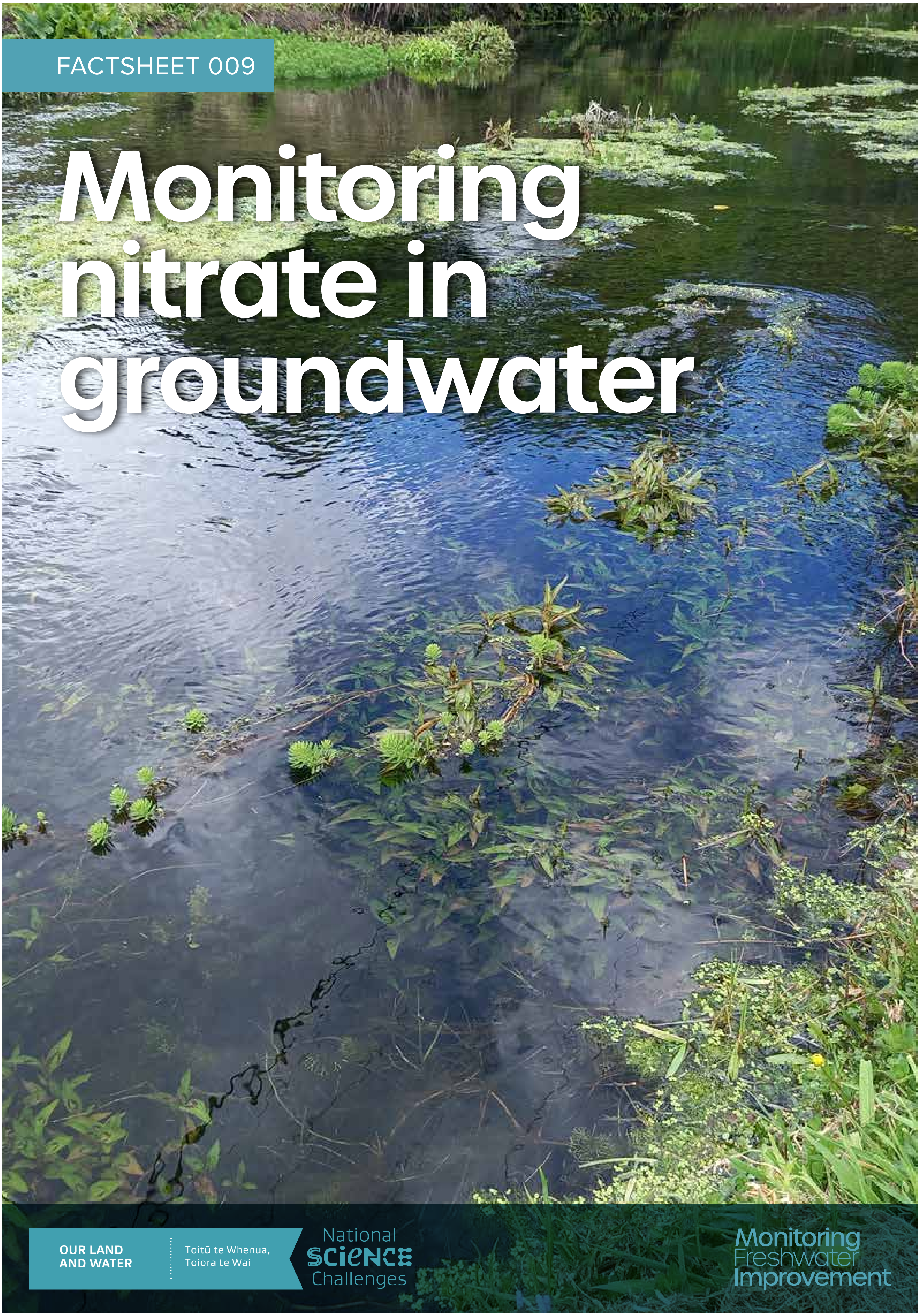
Table 8-2 **Environment Canterbury sampling cost breakdown**

Labour	**Cost**	**Notes**
Average hourly rate inclusive of overheads:	$70.00	Excludes cost for IT infrastructure, maintaining systems
Daily rate based on 8 hours inclusive of overheads	$560.00	Exclude field meter cost - extra cost for more samples is marginal for regional councils. Field meter cost may be prohibitive for catchment group
Average number of samples collected per day	6	OK for regional councils. Would get more/day if local catchment group
Labour cost per sample	**$95.00**	
Vehicle Running		
Average distance travelled per day	200km	Based on average miles over a few weeks for a few weeks with local and distant sites
Vehicle charge out rate per km	$0.75	
Vehicle running cost per day based on a 200km day	$150.00	
Average number of samples collected per day	6	
Vehicle running cost per sample	**$25.00**	
Diesel		
Average distance travelled per day	200 km	
Diesel tank size	65L	
Cost of diesel per litre	$2.80	
Cost to fill tank with Diesel	$180.00	
Distance travelled on tank of diesel	600km	
Number of average days per tank of diesel	3	
Cost of diesel per average day	$60.00	
Average number of samples collected per day	6	
Diesel cost per sample	**$10.00**	
Total cost per sample	**$130.00**	

Appendix 2. Nitrate monitoring fact sheet

FACTSHEET 009

Monitoring nitrate in groundwater



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Nitrate-nitrogen is a stable form of nitrogen found in freshwater ecosystems; it is highly soluble and can be readily used by vascular plants and algae for growth. Nitrate occurs naturally in New Zealand groundwaters but generally at very low concentrations. However, nitrate can leach through the soil and enter groundwater systems in high concentrations in areas of intensive agriculture and horticulture.

Elevated nitrate concentrations in groundwater poses a health risk to people if the groundwater is used for drinking water. It is also an issue where groundwater feeds into rivers and streams (for example, springs and rivers that are recharged by aquifers). High nitrate in surface waters can result in eutrophication (excessive nutrients), which can change the growth and types of aquatic plants and animals and lead to algal blooms. At high concentrations, nitrate can also be toxic to aquatic life.

How do we monitor nitrate in groundwater?

To date, most nitrate monitoring in groundwater in New Zealand has involved taking manual water samples that are sent for testing in a laboratory. For information on how to collect, store and transport groundwater samples for nitrate analysis in a laboratory, see the New Zealand National Environmental Monitoring Standard (NEMS): *Water Quality, Part 1 of 4: Sampling, Measuring, Processing and Archiving of Discrete Groundwater Quality Data.*

In New Zealand, laboratory results and guidelines relative to nitrate are generally expressed as nitrate-nitrogen (NO_3–N) concentrations[1].

Because 'discrete' nitrate sampling doesn't tell us what is happening between sampling events, high-frequency nitrate monitoring via sensors in monitoring wells is currently being trialled by several regional councils. High-frequency sampling can give us additional insights into peak concentrations and hydrological processes such as vadose zone storage and release to the water table. Sampling at a high frequency will also increase the statistical power for change detection. We discuss the pros and cons of discrete and high-frequency sampling more below.

High-frequency nitrate monitoring: how it works

There are two types of high-frequency sensors for nitrate monitoring: ultra-violet (UV) spectral sensors and ion selective electrode (ISE) sensors.

UV spectral sensors operate on the principle that nitrate absorbs radiation in the UV region of the electromagnetic spectrum at a characteristic wavelength of about 220 nm. Pulses of UV light are transmitted through a sample path of water (typically 20 mm), received by the sensor and spectrally analysed. The amount of UV absorbed by the water in the sampling zone is proportional to the concentration of nitrate in the water.

[1] 'Nitrate-nitrogen' (NO_3–N) refers to the nitrogen portion of the total nitrate (NO_3) in a sample. Some countries tend to express nitrate results and guidelines as total nitrate concentrations; thus, care must be taken when comparing results with international guidelines. For example, the World Health Organization's drinking water standard for nitrate concentration (50 mg/L) corresponds to 11.3 mg/L when expressed as nitrate-nitrogen concentration. Thus, care should be taken when interpreting results from the laboratory.

An ISE nitrate sensor operates on the principle that nitrate variation in the water will affect the electric potential of a solution in the probe. This change is then transmitted to the meter, which converts the electric signal to a scale that is read in millivolts. The millivolts are then converted to nitrate concentration. The accuracy of the electrode can be affected by high concentrations of chloride or bicarbonate ions in the sample water, and fluctuating pH.

The nitrate concentration data, measured by either type of sensor, are recorded and stored in a datalogger. The datalogger may then transmit the data to a central location (such as a base station PC), or data can be manually downloaded during a site visit.

The pros and cons of high-frequency nitrate monitoring

Because high-frequency monitoring can generate a significantly larger sample of nitrate time series data than discrete sampling, the probability of detecting statistically significant changes within a given period is higher. This may mean that the effectiveness of nitrate loss management actions can be determined more quickly and/or with more certainty. The magnitude of this potential time saving/certainty improvement can be constrained by serial correlation and the periodicity of background variability (aka “noise”) at the monitoring site, however. The *Groundwater quality monitoring design guide for management of diffuse nitrate pollution* provides more information on these limitations.

High-frequency data can also yield new insights into nitrate leaching processes and the local hydrological system. For example, a sensor installed by Environment Canterbury in north Canterbury showed large spikes in nitrate concentrations following rainfall, which provided important insights into lag times, the effect of land use intensification and concentration variance.

Environment Canterbury staff concluded that nitrate sensor data *would ideally be utilised for looking at timing of responses to various drivers, general trends and load modelling rather than checking compliance with limits or concentration thresholds*[2].

Nitrate sensors require routine maintenance and validation sampling site visits. The frequency of these visits depends on whether the sensor is fitted with an auto-cleaner to remove biofilm and/or sediment accumulation, the biochemistry of the well and the type of sensor. Quarterly site visits are generally recommended as a minimum, with more frequent visits likely to be required for sensors without auto-cleaners. Environment Canterbury has installed five nitrate sensors in groundwater monitoring wells and developed a validation sampling and site inspection schedule which comprises bi-monthly or monthly visits (depending on the sensor type) for maintenance (sensor cleaning) and routine validation samples[3].

[2] *Environment Canterbury Memo 15/09/22: Environment Canterbury continuous nitrate data, current site details, project scopes and disclaimers around the data.*

[3] *https://niwa.co.nz/publications/isu/instrument-systems-update-22-june-2018/time-for-a-closer-look-at-nitrates*

Some loggers provide quality assurance (QA) indicators based on readings from additional sensors (e.g. turbidity). Although these QA readings are generally used to identify unreliable data within the record, the readings could be used to signal when a maintenance visit is required via a telemetry system, and thereby optimise the maintenance visit frequency.

The recommended site visit allowance for budgeting purposes is:

- Monthly to bi-monthly for loggers without auto-cleaners
- Quarterly for self-cleaning loggers

The visit frequency should be optimised based on:

- Validation sampling results. These should include down-hole logger readings before and after cleaning of loggers without auto-cleaners plus an out-of-hole logger reading in the water sample to be sent for laboratory testing[4]. If logger and laboratory samples are sufficiently similar over a 12- to 18-month period, visit frequency could be reduced to quarterly. Conversely, divergence between laboratory and logger results could signal the need for more frequent site visits.
- QA readings for loggers with telemetry and a QA data feed.

How much will it cost?

The cost of carrying out a groundwater nitrate monitoring programme based on discrete sampling will depend on how often samples are collected, the location of the site and laboratory charges for analysing the samples. Some approximate costs for a single sampling occasion, based on average results from a survey of regional councils in 2022, are shown in the table. Using these estimates, monthly monitoring of nitrate in one well will cost about $2,700 per year. Travel time can be a significant proportion of the cost and hence localised monitoring (e.g. by a catchment group) may be less expensive.

	Cost per sampling occasion
Laboratory testing for nitrate-nitrogen	$12
Staff time, vehicle expenses, field equipment	$212
Total per sampling occasion	**$224**

The cost of operating a high-frequency nitrate monitoring site will depend on several factors, such as:

- The type of sensor chosen
- The location and accessibility of the monitoring well and frequency of maintenance visits
- Whether the data are electronically transmitted (telemetered) or manually downloaded.

The purchase cost of a mid-range nitrate sensor is in the range $15–20,000. Installation costs are likely to be relatively low for groundwater sites, in the order of $500, assuming telemetry is not included.

Ongoing costs would be in the order of $1,600 for the first year, assuming bi-monthly maintenance and validation sampling visits and allowing for some additional data processing and QA time. This could potentially be reduced to $1,000 per year if the first year of data show that maintenance and validation sampling visits can be decreased to quarterly without compromising the integrity of the data.

[4] This is important because the sampling process can draw groundwater from a different part of the formation and therefore result in a different nitrate concentration relative to the natural well throughflow under static conditions.

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Research Team | *Te hunga i whai wāhi mai*

PROGRAMME LEADER, IMPLEMENTATION LEAD
Olivier Ausseil
Aquanet Consulting

TE AO MĀORI LEAD
Joanne Clapcott
Ngāti Porou
Cawthron Institute

SCIENCE LEAD
David Hamilton
Griffith University

MATHEMATICS LEAD
Alasdair Noble
AgResearch

PHASE 1 CO-LEAD
Rogier Westerhoff
GNS Science

WEBAPP DEVELOPER
Mike Kittridge
Headwaters Hydrology