Proposal and testing of a ‘risk-based’ framework for assessing reduction in consent monitoring requirements of closed landfills.

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Abstract

Legacy municipal landfills, closed before the implementation of contemporary engineering standards, represent a persistent potential source of long-term environmental contamination, particularly in groundwater-dependent regions. In New Zealand, numerous such sites, constructed without engineered liners or leachate management systems, may continue to impact groundwater resources decades after closure. This thesis investigates the current environmental risk posed by seven closed, unlined landfills within the Central Hawke's Bay District and evaluates the ongoing justification for their groundwater monitoring regimes.

To enable consistent and defensible assessment across multiple heterogeneous sites, a risk-based decision-making flowchart was developed and applied. This flowchart integrates the Source–Pathway–Receptor (S-P-R) model with a semi-quantitative, multi-evidence assessment of hydrogeological setting, monitoring data, and receptor sensitivity. Risk classification is determined through converging lines of evidence, including statistical exceedance analysis (using UPL95 thresholds), inter-well comparisons (where possible), and evaluation of long-term contaminant trends in key monitoring parameters. Linear regression was applied to detect potential long-term trends in water quality, while t-tests assessed whether the mean concentrations of monitored parameters were significantly above or below nominated thresholds. All analyses were conducted on raw data, acknowledging potential variability introduced by analytical methods and seasonal influences, and any apparent outliers were carefully considered in context rather than automatically excluded.

The application of this framework indicates that the majority of the assessed landfills exhibit stable or declining contaminant concentrations. This is consistent with their progression into late methanogenic or maturation phases, where leachate strength naturally attenuates. Key conservative indicators, such as chloride and potassium, were consistently within or close to the background range, while ammoniacal nitrogen and organic parameters (TOC, VFA, UV254) generally show either no trend or weakening signatures. Notably, some minor fluctuations and isolated higher measurements were observed at a few sites, likely reflecting short-term environmental variability, laboratory differences, or seasonal influences, but these did not represent sustained or significant changes in groundwater quality.

Hydrogeological context is a critical moderating factor; several sites benefit from advantageous conditions such as river-adjacent dilution or hydraulic gradients that direct flow away from sensitive receptors. For example, downgradient monitoring bores at Waipawa, Tikokino, and Takapau are located over 2 km from the landfill footprint, which limits the potential for contaminant transport. In contrast, the Tamumu site exhibits shorter transport pathways and closer downgradient receptors, which correspond with detectable elevations, including total ammoniacal nitrogen exceeding the threshold limits and slightly elevated concentrations of TOC and UV254. Despite these elevated indicators, the concentrations remain below thresholds of human health concern, and the potential impact is partially mitigated by natural barriers such as the Tukituki River. In some cases, results are likely to reflect location and nearby landuse factors. For example, higher than usual conductivity, ammoniacal nitrogen, and TOC are present at Porangahau closed landfill, consistent with soil chemistry in a coastal location.

Across the seven landfills, the monitoring evidence can be broadly categorised into two behavioural groups. Six landfills (Waipawa, Waipukurau, Takapau, Tikokino, Ongaonga, and Porangahau) demonstrated low-risk characteristics, including no measurable downgradient enrichment, stable or declining concentrations, and chemical profiles consistent with natural background groundwater. Temporal trends were weak or absent, and t-tests consistently showed significant negative values, confirming compliance with nominated thresholds and reinforcing the conclusion that these sites exert minimal influence on surrounding groundwater quality. Tamumu represents a medium risk site, this classification is supported by upward trends in select parameters, shorter pathway distances, and the presence of downgradient receptors within 100 m.

The analysis of these seven case-study landfills highlights the effectiveness of the risk-based decision framework for synthesising disparate monitoring data into coherent, scientifically grounded risk assessments. By explicitly considering source characteristics, pathways, receptor alignment, and hydrogeological context, the framework allows for transparent classification of sites, supports the rationalisation of monitoring where appropriate, and provides defensible guidance for environmental management and regulatory compliance. Overall, the results demonstrate that, while some minor site-specific variations exist, the dominant pattern across Central Hawke's Bay's legacy municipal landfills is one of long-term stability and generally low environmental risk, with the exception of a single site (Tamumu) at medium-risk profile.

Keywords: Legacy municipal landfills, Environmental risk assessment, Risk-based decision making framework, Source–pathway–receptor model

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List of Abbreviations

Abbreviation	Full Term
BOD	Biochemical Oxygen Demand
CHB	Central Hawke's Bay
DOM	Dissolved Organic Matter
EC	Electrical Conductivity
MfE	Ministry for the Environment
NES-CS	National Environmental Standard for Assessing and Managing Contaminants in Soil
NES-FW	National Environmental Standards for Freshwater
NPS-FM	National Policy Statement for Freshwater Management
NZDWS	New Zealand Drinking Water Standards
RMA	Resource Management Act 1991
RRMP	Hawke's Bay Regional Resource Management Plan
S-P-R	Source–Pathway–Receptor
TAN	Total Ammoniacal Nitrogen
TOC	Total Organic Carbon
UPL95%	95% Upper Prediction Limit
UV254	Ultraviolet Absorbance at 254 nm
VFA	Volatile Fatty Acids
WWTP	Wastewater Treatment Plant

1. Introduction

1.1 Overview

Across New Zealand, many closed municipal landfills were established before the adoption of modern engineering and environmental protection standards. These older sites typically lack basal liners, leachate collection systems, engineered capping, and long-term containment measures that are now considered fundamental to landfill design. As a result, they continue to generate leachate for decades after closure, creating persistent risks to groundwater, surface water, and surrounding environments (Mukherjee, 2015). For regional authorities, understanding the long-term behaviour of these legacy landfills is essential to ensuring public health and environmental protection, particularly as monitoring networks age and land uses surrounding closed sites evolve (Raco, 2010).

The Central Hawke's Bay District contains seven closed landfills that operated between the 1950s and 1990s, well before contemporary engineering and environmental controls were introduced (Nanda, 2021). All these landfills at Waipawa, Waipukurau, Takapau, Tamumu, Tikokino, Ongaonga, and Porangahau were constructed without engineered containment systems. Their locations vary from inland valley floors to coastal margins, and many sit close to rivers, streams, or shallow groundwater systems. These characteristics mean that each site has a unique hydrogeological setting and therefore a different potential to affect surrounding environments. While monitoring has been undertaken for many years, the degree of ongoing risk remains uncertain, and some monitoring networks no longer reflect current site conditions, changes in land use, or present-day regulatory expectations.

The core problem addressed in this thesis is the long-term environmental risk posed by unlined closed landfills and, specifically, their potential to continue influencing groundwater quality long after disposal activities have ceased (Vaverková, 2019). Because the quality of monitoring data, site characterisation, and hydrogeological understanding varies across sites, there is a need for a consistent, transparent, and scientifically defensible method for evaluating risk and determining whether ongoing monitoring remains necessary.

The purpose of this study is to assess the environmental risk associated with the seven closed landfills in Central Hawke's Bay and to determine whether current monitoring requirements are scientifically justified, environmentally appropriate, and aligned with legal and policy frameworks. The study integrates hydrogeological assessment, historical groundwater monitoring data, and regulatory analysis to create an evidence-based foundation for future decision-making.

1.2 Research context

1.2.1 Historical landfill design and the evolution of sanitary landfills

The development of landfill design in New Zealand has followed global trends, shifting from unregulated waste dumping toward engineered containment systems. Prior to the 1970s, most rural and small-town disposal sites functioned primarily as open dumps or simple burial pits, lacking engineered infrastructure such as liners, leachate collection systems, or gas management controls (Kjeldsen, 2002). These early landfills often permitted uncontrolled infiltration of precipitation and subsequent leachate migration into underlying soils and groundwater. The absence of environmental monitoring and operational control further exacerbated long-term risks, leaving legacy sites vulnerable to persistent contamination.

The transition to sanitary landfill principles was driven by increasing regulatory oversight and environmental awareness during the late twentieth century. Modern sanitary landfills incorporate engineered liners, leachate collection systems, gas extraction, and impermeable caps to isolate waste and limit migration of contaminants (MfE, 2001). In contrast, older landfills, such as those examined in the Central Hawke's Bay District, were constructed prior to these regulatory standards. Consequently, these sites remain unlined, with no engineered leachate or gas control, and rely primarily on natural attenuation processes for contaminant reduction. Understanding the long-term behaviour of such legacy landfills is essential to inform environmental management and monitoring strategies

1.2.2 Environmental risks of unlined closed landfills

Unlined landfills present enduring environmental risks, primarily due to leachate migration into soil and groundwater. Leachate from municipal waste contains a complex mixture of dissolved organic matter, nitrogen species, heavy metals, and conservative ions such as chloride, along with emerging contaminants (Siddiqua et al., 2022). In the absence of engineered barriers, these contaminants can infiltrate subsurface environments and, in some cases, reach surface water systems. The risks associated with unlined landfills are predominantly chronic rather than acute, often persisting for decades after closure (Vaverková, 2019). Certain contaminants, such as ammoniacal nitrogen, chloride, and refractory organics [1], remain mobile over long periods, underscoring the need for sustained monitoring.

The environmental consequences of leachate migration extend to both human and ecological receptors. Groundwater used for drinking water may be affected if contaminants reach abstraction points, while surface-water bodies and ecological

[1] Refractory organics are persistent, non-biodegradable organic pollutants, such as aromatic compounds and halogenated substances.

habitats may experience chronic nutrient or metal loading. These risks are compounded by variable hydrogeological conditions, including shallow aquifers, permeable soils, or proximity to rivers and streams, which can facilitate contaminant transport (Christensen, 2001). Despite natural attenuation processes and the decline in waste biodegradability over time, legacy unlined landfills remain potential sources of long-term environmental impact, warranting careful assessment and monitoring.

1.2.3 Leachate composition and long-term biochemical phases (>20 years)

The biochemical evolution of landfill waste governs leachate composition and contaminant mobility. Classical landfill decomposition follows a sequence of phases: aerobic, acidogenic, methanogenic, and maturation (Kulikowska & Klimiuk, 2008). Each phase is characterised by distinct microbial processes, chemical signatures, and contaminant profiles. Due to the ages of the landfills examined in this study, the aerobic and acidogenic conditions would have long finished, and only the later phases are relevant.

Methanogenic Phase

During the methanogenic stage, syntrophic microbial communities convert volatile fatty acids (VFAs) into methane and carbon dioxide. This phase is typically accompanied by a rise in pH toward neutral levels and a sharp decline in biochemical oxygen demand (BOD) and VFAs. Ammoniacal nitrogen becomes the dominant contaminant, derived from the anaerobic mineralisation of proteins. The methanogenic phase represents the most prolonged and stable period of landfill evolution, often lasting several decades (Mor, 2006).

Maturation / Late Methanogenic Phase (>20 years)

Landfills that have been closed for over two decades generally transition into the late methanogenic or maturation phase. In this stage, biodegradable organic matter is substantially reduced, VFAs are very low or absent, and total organic carbon (TOC) declines while humic and refractory organics persist. Ammoniacal nitrogen remains detectable but typically shows gradual attenuation, and conservative ions such as chloride and potassium stabilize at low to moderate concentrations. This chemical profile reflects the weakened leachate strength and reduces environmental threat associated with older, stabilised landfills (Kulikowska & Klimiuk, 2008).

Key chemical indicators expected in older landfills are summarised in Table 1.

Table 1: Key Chemical Indicators

Indicator	Expected Behaviour (>20 years)	Interpretation
pH	Neutral–alkaline	Stabilised methanogenic environment
Electrical Conductivity	Moderate, declining	Reduced ionic load from aged leachate
Chloride	Low–moderate, conservative	Indicator of historical leachate strength
Potassium	Declining	Leaching of soluble fractions
TAN	Persistent; slow attenuation	Long-term risk indicator
TOC	Low	Reflects reduced organic content
VFAs	Very low/absent	Completion of acidogenic phase
UV254	Moderate	Presence of refractory humic substances

TAN -Total Ammoniacal Nitrogen, TOC - Total Organic Carbon, VFAs - Volatile Fatty Acids, UV254 - Ultraviolet Absorbance at 254 nm

1.2.4 Conceptual model of contaminant transport (source–pathway–receptor)

The source–pathway–receptor (S–P–R) framework provides a conceptual basis for evaluating environmental risk at closed landfills. In this context, the three components of this model are as follows.

- Source: Waste deposits in legacy landfills continue to generate leachate over decades. The magnitude of leachate production is influenced by residual organic matter, waste composition, and infiltration rates (Farquhar, 1989).
- Pathway[2]: Groundwater transport is the predominant mechanism for contaminant migration. Aquifer properties, hydraulic gradients, and proximity to surface water determine the strength and direction of the pathway. In the context of long-closed landfills, discharges to air are expected to usually be minor to negligible, and groundwater quality is the target of regulatory monitoring. The focus of this research is therefore on this pathway.
- Receptor: Sensitive receptors include drinking-water bores, surface water bodies, and ecological habitats. Exposure risk depends on the receptor's location relative to the plume and its intrinsic sensitivity. Many New Zealand drinking-water bores are distant or upgradient, reducing the likelihood of exposure.

[2] The influence of underlying geology and soil type on contaminant transport and attenuation is important, as these factors control permeability, adsorption capacity, and biodegradation potential. However, this aspect is not considered in the current research and is recommended as an area for future study.

Application of the S–P–R model allows for a systematic assessment of the likelihood and magnitude of contaminant transport, informing both monitoring network design and risk-based management strategies.

1.2.5 Relevance to drinking water supply protection

Groundwater protection is a critical driver for landfill monitoring, as contaminants such as ammoniacal nitrogen, nitrate, chloride, and refractory organics can compromise potability. Early detection of changes in water quality is essential to protect public health, particularly where abstraction wells are present. Monitoring wells positioned between the landfill and drinking-water supplies serve as an early-warning system, allowing intervention before concentrations approach guideline limits (Przydatek & Kanownik, 2019).

1.2.6 Relevance of geographic and site-specific features

The assessment of hydrogeological settings, inland, riparian, or coastal, informs monitoring strategies. For inland sites, the primary focus is subsurface groundwater plumes, while riparian and coastal sites require additional consideration of hyporheic zones (transition areas beneath and alongside riverbeds where groundwater and stream water mix), vertical profiling, and freshwater–saltwater interactions. Conservative tracers such as chloride and potassium provide reliable early-warning indicators of the presence of leachate, complemented by organic parameters that reflect ongoing leachate decomposition. Three general settings relevant to this thesis are inland locations, riparian sites, and coastal sites.

- **Inland Sites:** For landfills remote from surface water, the sole pathway is subsurface groundwater flow. Monitoring must focus on classic plume delineation to protect downgradient potable aquifers, with well networks designed to capture the core, fringe, and leading edge of the contaminant plume over long flow paths (Christensen, 2001; Huang, et al., 2024).
- **Riparian Sites:** Landfills adjacent to streams or rivers introduce a critical groundwater-surface water interface. Here, the hyporheic zone becomes the focal point. Monitoring must expand beyond sentinel wells to include transects perpendicular to the stream, quantifying the contaminant flux discharging into surface water to protect aquatic ecosystems and downstream water intakes (Lorah, Cozzarelli, & Böhlke, 2009; Milosevic, Thomsen, Juhler, Albrechtsen, & Bjerg, 2012).
- **Coastal Sites:** These present the highest complexity due to freshwater-saltwater interactions. The density of saline water can cause leachate plumes to sink and behave unpredictably, while tidal forces and changing redox conditions can remobilize metals. Monitoring networks must be denser, include deeper wells for

vertical profiling, and routinely analyse salinity indicators to deconvolve the complex geochemical signals (Brand, Spencer, O'shea, & Lindsay, 2017).

1.3 Legislative framework

1.3.1 Current framework

The management of legacy landfills in New Zealand is guided by multiple statutory and policy frameworks. Significant components of these are outlined below.

The **Resource Management Act 1991 (RMA)** provides the overarching statutory framework governing discharges, land use, and environmental effects (RMA, 1991). Several sections are directly relevant to the ongoing management of closed landfills:

- Section 15 prohibits the discharge of contaminants (such as leachate) into water or onto/into land in circumstances where it may enter water (including groundwater), unless expressly allowed by a regional plan, proposed plan, resource consent, or regulation. For closed landfills, this commonly requires a discharge permit because ongoing leachate migration from waste into underlying groundwater typically constitutes a prohibited activity under s15(1)(a) or (b).
- Section 17 imposes a general duty on every person to avoid, remedy, or mitigate any adverse effects on the environment arising from their activities, which applies to landfill operators or owners managing long-term groundwater contamination risks post-closure.
- Section 30(1)(f) assigns regional councils the function of controlling the discharge of contaminants into the environment (including to groundwater) to give effect to the RMA's purpose of sustainable management, thereby empowering them to regulate such discharges through plans and consents for closed landfills. Together, these provisions require ongoing management, monitoring, and often resource consents to address leachate impacts on groundwater from legacy landfills.
- Section 104 requires decision-makers to consider the actual and potential effects of an activity on the environment, relevant planning documents, and other matters such as the efficient use of natural resources. When evaluating monitoring changes for closed landfills, councils must rely on robust evidence to demonstrate that effects on groundwater and surface water remain "no more than minor".
- Section 105 requires consideration of alternative methods of discharge and the nature of the discharge and receiving environment. Although closed landfills no longer actively discharge managed leachate, they remain ongoing sources of contaminant release, meaning monitoring must continue unless evidence demonstrates decreasing or negligible effects.
- Section 107 prohibits granting discharge permits that may result in significant adverse effects such as contamination, objectionable colour, or toxicity, unless

exceptions apply. While closed landfills pre-date modern permitting, any proposed changes to monitoring must demonstrate that undetected discharges will not breach Section 107 thresholds.

- Section 127 enables consent holders to apply for changes to existing conditions, such as groundwater monitoring requirements, if the change does not alter the activity's fundamental character, scale, or intensity. This provision is central to the present assessment: any reduction or rationalisation of monitoring must be supported by scientific evidence demonstrating that leachate effects are stable, declining, or negligible. Councils must show that any amendment continues to meet the purpose of the RMA and protects relevant environmental values.

National Environmental Standards

The **National Environmental Standards for Freshwater (NES-FW)**: establish minimum requirements for protecting freshwater ecosystems, including provisions for water quality and the management of activities that may affect rivers, wetlands, and groundwater (NES-FW, 2020). However, as the closed landfills were consented prior to the introduction of the NES-FW, these regulations are unlikely to be triggered unless remedial works are undertaken, such as earthworks or vegetation clearance in close proximity to wetlands. Therefore, monitoring near surface water bodies at Waipawa, Waipukurau, and Porangahau remains important, the direct applicability of the NES-FW to these sites is limited under current conditions .

The **National Environmental Standard for Assessing and Managing Contaminants in Soil (NES-CS[3])**: are a set of regulations that apply when assessing risks to human health from contaminants in soil, in the context of land-use consent applications that involve subdivision, earthworks or building on potentially contaminated (NES-CS, 2011). Although not directly governing groundwater monitoring, the NES-CS influences landfill management by requiring councils to understand legacy contamination, ensure that closed landfills do not create exposure pathways, and maintain adequate site characterisation information.

Until recently, the **National Policy Statement for Freshwater Management (NPS-FM 2020[4])**: required councils to manage freshwater in a way that prioritises: the health and well-being of water bodies and ecosystems, the health needs of people (such as drinking water), and the ability of people and communities to provide for their social, economic, and cultural well-being (NPS-FM, 2020). Under this hierarchy, groundwater protection was a first-order priority, reinforcing the need for monitoring networks capable of

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[3] The NES-CS regulations would only be triggered if there is a proposal to disturb or subdivide a landfill. This may occur if remedial works are required and the cap or fill is to be disturbed during that process.
[4] The NPS-FM may be relevant to closed landfills where ongoing monitoring is required and trigger limits need to be reviewed to ensure that the national bottom line attributes for various contaminants (as listed in Appendix 2A) are met in downstream freshwater bodies.

identifying contaminant migration from closed landfills before drinking water supplies are affected. The NPS-FM 2020 also required councils to adopt a precautionary approach when effects are uncertain, further emphasising the need for strong evidence before monitoring reductions are approved.

Hawke's Bay Regional Resource Management Plan (RRMP)

The Hawke's Bay Regional Resource Management Plan (RRMP) provides an integrated framework for managing resource use in the Hawkes Bay region and consists of both a Regional Policy Statement (RPS) and Regional Rules (RRMP, 2006). The plan was notified in April 2000 and became fully operative on 28 August 2006. As such it sets the regional framework for any new changes to conditions applying to resource consents associated with the closed landfills that are the subject of this study. For proposals to change monitoring conditions, a full reassessment (of these consents granted before the RRMP became operative) is not necessary. This is because the relevant provisions of earlier policies and plans (see Section 1.4.2) were considered at the time of the substantive decision on these consent applications, the proposed changes relate to water quality monitoring only.

However, the RRMP is still clearly relevant to consider, because any changes should not have the effect of altering the substantive consent objectives or causing a worsening of environmental outcome. Relevant sections include section 5.2 (Groundwater Quality) which aims to maintain or improve groundwater quality, and section 5.4 (Surface Water Resources) which seeks to avoid, remedy, or mitigate adverse effects on surface water quality and quantity.

These policies align with RMA sections 104 and 127 by requiring that any monitoring changes be justified through technical assessment demonstrating reduced risk, limited pathway strength, or absence of sensitive receptors.

1.3.2 Legal basis for changing monitoring conditions

Because closed landfills remain potential long-term contaminant sources, councils must be able to demonstrate that monitoring changes:

- Are scientifically defensible,
- Do not increase environmental risk, and
- Are consistent with all applicable legislation and policy.

Under section 127 of the RMA, changes cannot be approved solely on the basis of reduced operational needs or cost efficiency. Instead, the decision must be grounded in evidence showing:

- Contaminant trends are stable or declining;
- Pathways to receptors are weak, absent, or fully characterised;

- Effects remain “no more than minor” under section 104;
- The receiving environment meets NES-FW and NPS-FM requirements;
- Changes remain consistent with the RRMP policy direction.

Thus, the legislative framework operates alongside the hydrogeological and chemical assessments presented in this thesis, ensuring that both scientific evidence and statutory obligations are met before any modification to monitoring conditions is considered.

1.3.3 Future changes

Legislation governing resource management in New Zealand, and local council structures, are current undergoing significant changes. On 9 Dec 2025, the New Zealand government introduced the Planning Bill, and the Natural Environmental Bill. When enacted, these will replace the RMA (MfE, 2025). Concurrently, changes have and will be undertaken with national direction instruments, from national policy statements to national standards. Legislation has also been introduced to manage transitional arrangements. Under these arrangements, most resource consents will be extended until 2031 (MfE, 2025).

1.4 Sites selected for case study

1.4.1 Basic site information of selected landfills

The study area (Figure 1) encompasses eight closed landfills within the region, each presenting distinct characteristics and environmental contexts.

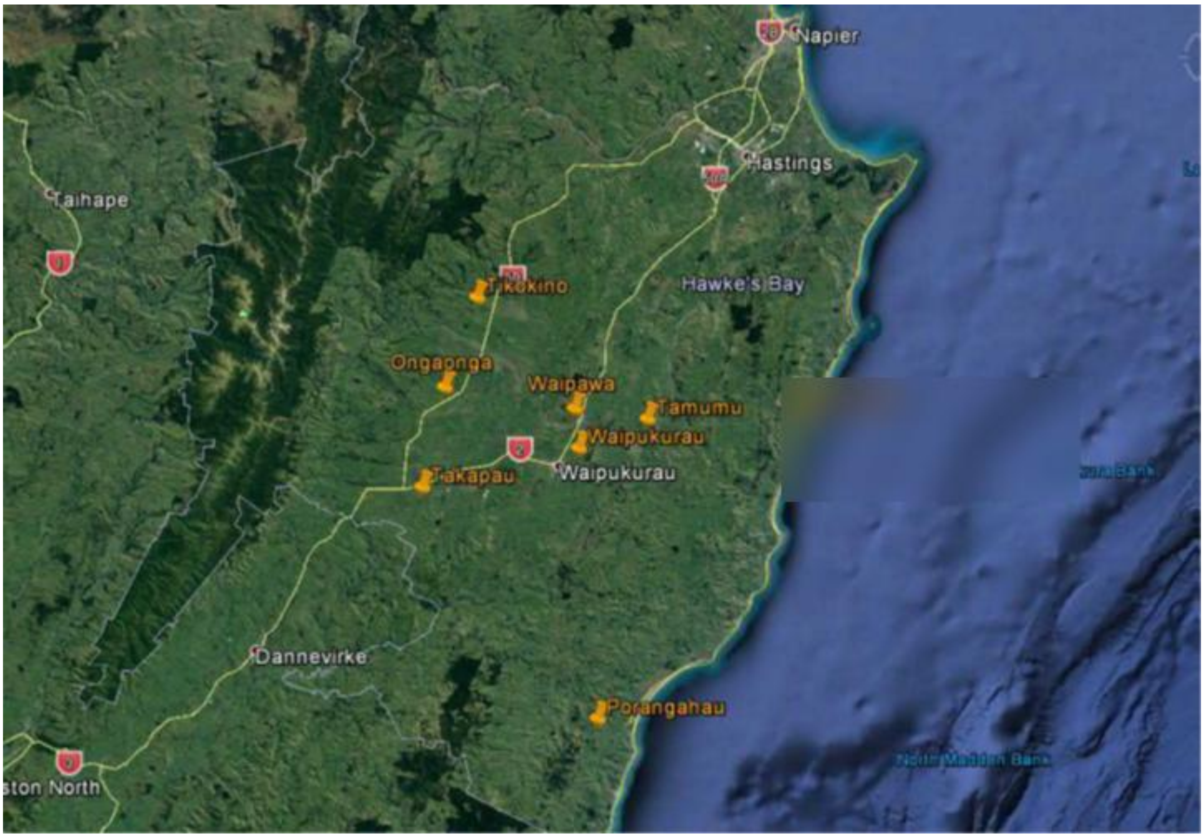


Figure 1: GIS map - Central Hawkes Bay Closed Landfills

The landfills were used for the disposal of municipal waste generated by nearby small communities and were active from around the 1950s to the 1990s.

None of the sites were constructed with liner systems, and the cover materials typically consist of locally sourced fill. These covers were not installed to meet current standards for low-permeability capping systems, and no leachate or landfill gas control measures are in place. These site characteristics are indicative of typical landfill practices during that period.

The Waipawa closed landfill is a 1.4 hectare site situated on Tikokino Road, adjacent to Waipawa Township. It is bordered by rural land to the west, residential areas to the north, sports fields to the east, and lies approximately 50 metres from the Waipawa River to the south. The vegetated site forms a mound on the river's northern terrace, with topography sloping toward the river and a boundary drain, minimising rainwater infiltration. A refuse transfer station operates at its eastern end.

The Waipukurau closed landfill is a 1.0 hectare, triangular-shaped site located on Mt Herbert Road, roughly 1.5 km from Waipukurau township and 50 metres from the south bank of the Tukituki River. It is neighboured by a transfer station and wastewater treatment plant to the west, with rural land to the south. A public walkway has been established along its northern boundary. Approximately 8.3 km east of Waipawa, the Tamumu closed landfill is a larger 2.0-hectare site on River Road. It operated from 1956 to 1991, features a 2-3 metre engineered cap, and is situated 200 metres east of the Tukituki River with a small stream on its northern side.

Several smaller, historically rural community dumps are also included. The Ongaonga closed landfill (0.7 hectare) on Blackburn Road, operational from 1972 to 1994, is located 1.6 km north of the Tukituki River. The Takapau closed landfill (0.7 hectare), now used as a transfer station, lies 600 metres south of Takapau township and is concerningly proximate, within 20 metres, to the Porangahau Stream, where flow is restricted by culverts. The Tikokino closed landfill is a 0.3 hectare site on Holden Road, 2 km west of Tikokino, manifesting as a defined mound in pastoral land approximately 75 metres from an ephemeral stream.

One coastal site completes the inventory: the Porangahau closed landfill (0.6 hectare) on Keppel Street is situated about 50 metres from the tidally influenced Porangahau River, on a terrace likely above groundwater level.

1.4.2 Site information, resource consents and groundwater monitoring requirements

Legal descriptions, addresses, areas, consent authorisations, and an overview of groundwater monitoring requirements for the seven closed landfill sites are provided in Tables 2 and 3.

Table 2: Legal descriptions, addresses, areas, resource consent authorisations, and an overview of groundwater monitoring requirements for the seven closed landfill sites.

Sites	Authorisation Number	Site address	Legal description	Site area (hectares)	Comments
Waipawa	DP950138 AUTH-107831-01 AUTH-107831-04	Tikokino Road, Waipawa	Lt 1 DP 16272 and Pt DP 2404	1.4	Four bores monitored for groundwater levels. Upstream/Downstream bores are confirmed; two bores are monitored for water quality.
Waipukurau	DP950134 AUTH-107815-01 AUTH-107815-04	Mt Herbert Rd, Waipukurau	Lt 1 DP 9735	1.0	Upstream/Downstream bores are confirmed; only downgradient bore was tested as other has been destroyed.
Takapau	DP950137 AUTH-107826-01 AUTH-107826-04	Paulsen Road, Takapau	Otawhao A3 Sec 49A No.2 Blk	0.7	Three bores monitored for groundwater levels. Downstream bore is confirmed and monitored for water quality.
Tamumu	DP950133 AUTH-107810-01 AUTH-107810-04	River Road, Tamumu	Lt 1 DP 12148	2.0	Two bores monitored for groundwater levels. Downstream bore is confirmed and monitored for water quality.
Tikokino	DP950132 AUTH-107805-01 AUTH-107805-04	Holden Road, Tiokino	Sec 1 SO 2365	0.3	Three bores monitored for groundwater levels. Upstream/Downstream bores are not confirmed, so all three are monitored for water quality.
Ongaonga	DP950130 AUTH-107795-01 AUTH-107795-04	Blackburn Road, Ongaonga	Sec 4 Blk XI Ruataniwha SD	0.7	Three bores monitored for groundwater levels. Downstream bore is confirmed and monitored for water quality.
Porangahau	DP950131 AUTH-107800-01 AUTH-107800-04	Keppel Street, Porangahau	Sec 7 Blk XII Porangahau SD	0.6	Two bores monitored for groundwater levels. Downstream bore is confirmed and monitored for water quality.

Table 3: The summary of the groundwater monitoring for all sites

Closed landfill sites	Monitoring bore required	Sampling regimes
Waipawa Closed Landfill Tikokino Closed Landfill	One bore upstream of the landfill, and at least one downstream of the landfill are proposed for monitoring purposes.	Groundwater sampling of group one parameters shall occur in February, May, August and November of each year. Sampling of group two parameters (includes group one) shall be carried out every third year.
Waipukurau Closed Landfill Takapau Closed Landfill Tamumu Closed Landfill Ongaonga Closed Landfill Porangahau Closed Landfill	One bore downstream of the landfill shall be constructed for monitoring purposes.	Groundwater sampling of group one parameters shall occur in February, May, August and November of each year. Sampling of group two parameters (includes group one) shall be carried out every third year.

The resource consents (or 'authorisations,' Table 2) were granted in 2004 and were for a term of 30 years. These are held by Central Hawke's Bay District Council, and the authorising agency was Hawke's Bay Regional Council. As outlined in Section 1.3, the consents were granted pursuant to the Resource Management Act 1991 and relevant provisions of the Hawke's Bay Regional Council's planning instruments. At that time, these included Rule 6.1 of the Proposed Regional Water Resources Plan (November 1996), Rule 20 of the Regional Air Plan (January 1998), Rule 4.4.1.2 of the Regional Waste and Hazardous Substances Plan (April 1995), and Rule 40 of the Proposed Regional Resource Management Plan (June 2001).

1.4.3 Suitability of the current monitoring parameters

The suite of parameters required by the current resource consent (pH, electrical conductivity (EC), chloride, dissolved potassium, total ammoniacal-N, nitrite-N, nitrate-N, total organic carbon (TOC), and volatile fatty acids (VFAs)) forms a practical core for detecting and interpreting landfill leachate influence. Collectively, these parameters capture the dominant geochemical and biochemical indicators associated with leachate, including ionic strength, major conservative tracers, nitrogen transformation pathways, and organic loading. This combination is consistent with international and New Zealand guidance for closed landfill monitoring programmes (Environment Agency, 2004; MfE, 2001; SWANA, 1991).

Based on the NZ Drinking Water Standards, the acceptable range for pH is 7.0 – 8.5.[5] Impacted at landfill plumes: can extend to acidic ($\leqslant 6$) or alkaline ($\geqslant 9$) locally, depending

[5] Note: it is understood that a pH range of 6.5-9.0 from ANZECC (2000) for protection of aquatic ecosystems is used by at least some other New Zealand regional councils in similar closed landfill consents. In these (Hawkes Bay Regional Council) consents, it appears that the NZ Drinking Water Standard range of pH 7.0–8.5 was selected at the time consents were issued. This would be consistent with the main risk being managed being the potential use of groundwater as drinking-water.

on leachate chemistry and redox reactions (reports case studies show pH commonly between 6.40 – 7.80 in leachate-impacted wells (Negi, 2020).

The inclusion of EC and chloride offers robust early-warning capability, as both respond sensitively to the presence of dissolved inorganic ions and behave conservatively in aquifer systems. pH provides essential context for interpreting contaminant behaviour, especially ammonia toxicity, metal mobility, and microbial processes. Dissolved potassium serves as a useful, though less conservative, supplementary indicator that strengthens the chemical fingerprint of leachate when interpreted alongside chloride and EC (Aleksandra, 2020).

The nitrogen suite (total ammoniacal-N, nitrite-N, and nitrate-N) is particularly valuable because it reveals both contamination and redox conditions. Elevated ammoniacal-N is one of the most persistent indicators of landfill impact, while shifts toward nitrite or nitrate reflect changes in oxygen availability and the progression of natural attenuation processes. These parameters therefore support the assessment of both current risk and long-term stabilisation of the landfill system (Koda, 2015; Environment Agency, 2004).

The organic indicators TOC and VFA provide insight into the biochemical state of waste decomposition and the organic strength of leachate. TOC helps quantify overall organic loading and potential oxygen demand, while VFA concentrations reveal whether portions of the landfill remain in the acidogenic stage or have transitioned to methanogenesis. Their inclusion strengthens the monitoring programme's ability to determine landfill stability and evaluate longer-term risk profiles (Negi, 2020; MfE., 2023).

1.5 Research questions

As noted in Section 1.1, the purpose of this study is to assess the environmental risk associated with the seven closed landfills in Central Hawke's Bay and to determine whether current monitoring requirements are scientifically justified, environmentally appropriate, and aligned with legal and policy frameworks.

Accordingly, this thesis addresses the following research questions:

1. What is the current level of environmental risk associated with each of the seven closed unlined landfills in Central Hawke's Bay?
2. Can a standardised, defensible framework be developed to classify risk and guide future monitoring requirements?
3. Which sites require continued monitoring, and where can monitoring be reduced or rationalised based on available evidence?

In a local regulatory context, the third research question anticipates the situation where the consent holder (in this case, Central Hawke's Bay District Council) may apply to the consenting authority (Hawkes Bay Regional Council) to reduce current groundwater monitoring requirements.

In addressing these three questions, the thesis provides an integrated assessment of the Waipawa, Waipukurau, Takapau, Tamumu, Tikokino, Ongaonga, and Porangahau closed landfills and presents a decision-making framework that supports consistent, transparent, and proportionate environmental management across the district.

This research is significant for two main reasons. First, it provides a systematic evaluation of long-term groundwater risks from legacy unlined landfills in a region where groundwater resources support agriculture, domestic use, and ecological values. Second, it develops a risk-based decision-making framework that can be applied consistently across multiple closed sites.

2. Methodology and analytical framework

2.1 Overview

In order to undertake this analysis, it was first necessary to develop a structured, evidence-based methodology to evaluate the environmental risk posed by seven closed municipal landfills within Central Hawke's Bay.

Any decision-making flowchart would need to integrate the source–pathway–receptor paradigm, principles of natural attenuation, and quantitative monitoring evidence, aligning with internationally recognised contaminated-site assessment frameworks (Christensen, 2001). Key elements are as follows:

- **Source**: the assessment considered the landfill age, waste composition, and expected leachate strength during late methanogenic or maturation phases (Kjeldsen, 2002).
- **Containment**: the assessment evaluates both engineered and natural barriers. As all seven landfills lack engineered liners and leachate collection systems, containment relies primarily on natural features.
- **Pathway**: assesses the potential for contaminants to migrate through groundwater. Key variables include hydraulic gradients, aquifer conductivity, and proximity to rivers or ephemeral streams. Underlying geology and soil types can have a significant influence on migration, for example by retarding or attenuating contaminant transport and migration.
- **Receptor**: Sensitive downgradient receptors, including water supplies, surface-water bodies, and ecological habitats, were identified. The evaluation considered distance, hydraulic connectivity, and potential exposure pathways, following risk-based groundwater protection principles.
- **Monitoring Evidence assessment**: contaminant trends (increasing, stable, or declining) were determined through analysis of key parameters, including pH, electrical conductivity (EC), chloride, potassium, ammoniacal nitrogen, nitrate-N,

total organic carbon (TOC), volatile fatty acids (VFAs), and ultra-violet light absorbance at 254 nm (UV254, a measure of dissolved organic compounds). Interpretation was guided by known leachate evolution patterns, anticipating stability or decline in older landfills (Mor, 2006).

- **Risk Outcome**: the final decision node synthesises all preceding evidence to assign a low, medium, or high environmental risk classification. This classification provides a defensible rationale for maintaining, modifying, or reducing site-specific monitoring regimes.

The analytical framework (set out in the sections following) synthesises hydrogeological assessment, monitoring data evaluation, statistical analysis, and a customised risk-based decision tool to ensure a consistent and defensible appraisal across all sites. The subsequent sections detail the methodological components used to classify environmental risk and to determine the scientific and legal justification for ongoing monitoring programmes.

2.2 The decision-making flowchart and its components

2.2.1 The flowchart

The decision-making flowchart that was both developed and applied in this work is shown in Figure 2 (the overall flowchart) and Figure 3 (which provides further detail about the monitoring evidence assessment component). Specific aspects of the decision system are provided in the sections that follow.

Figure 2: Proposed Decision-making flowchart 1

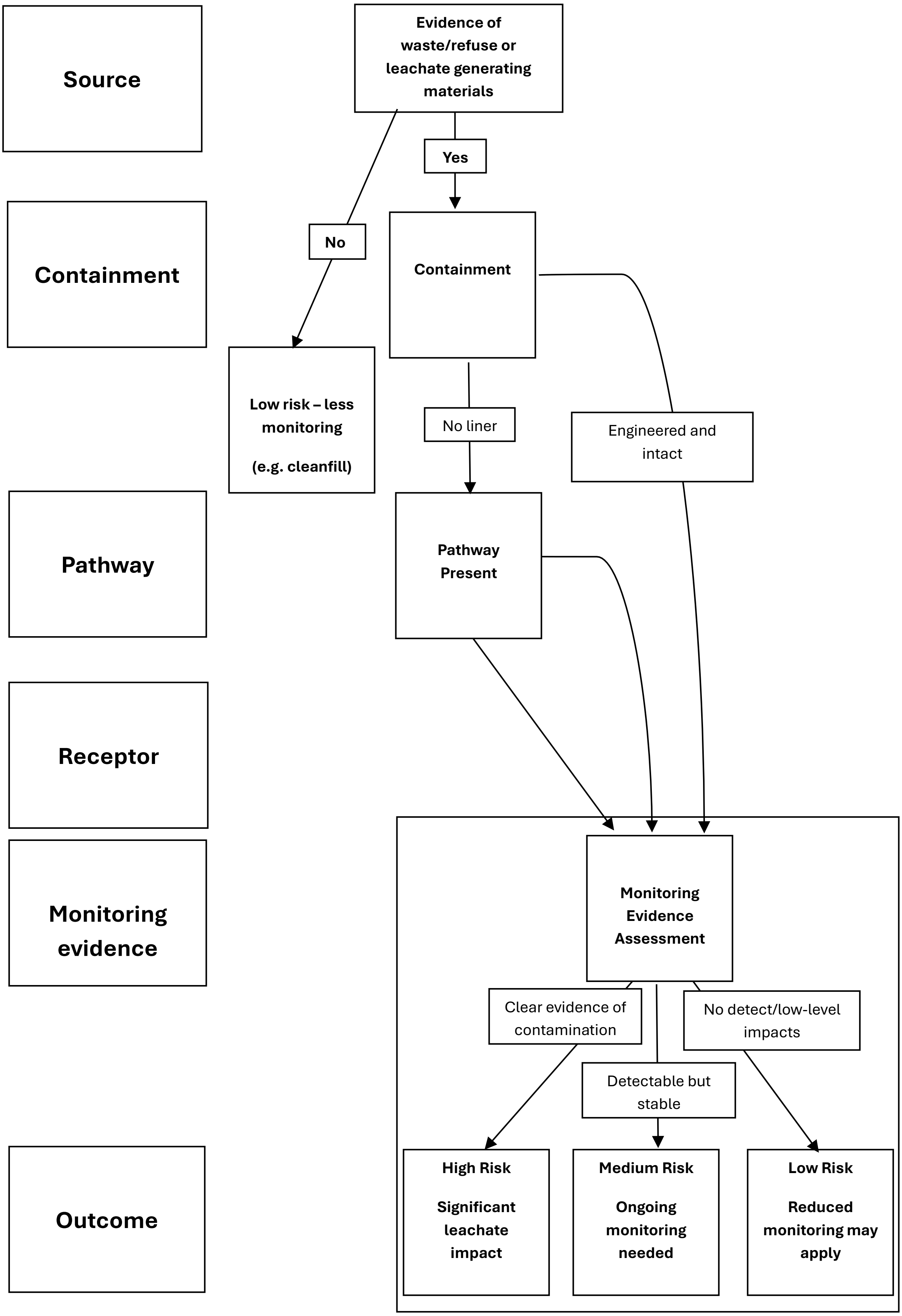
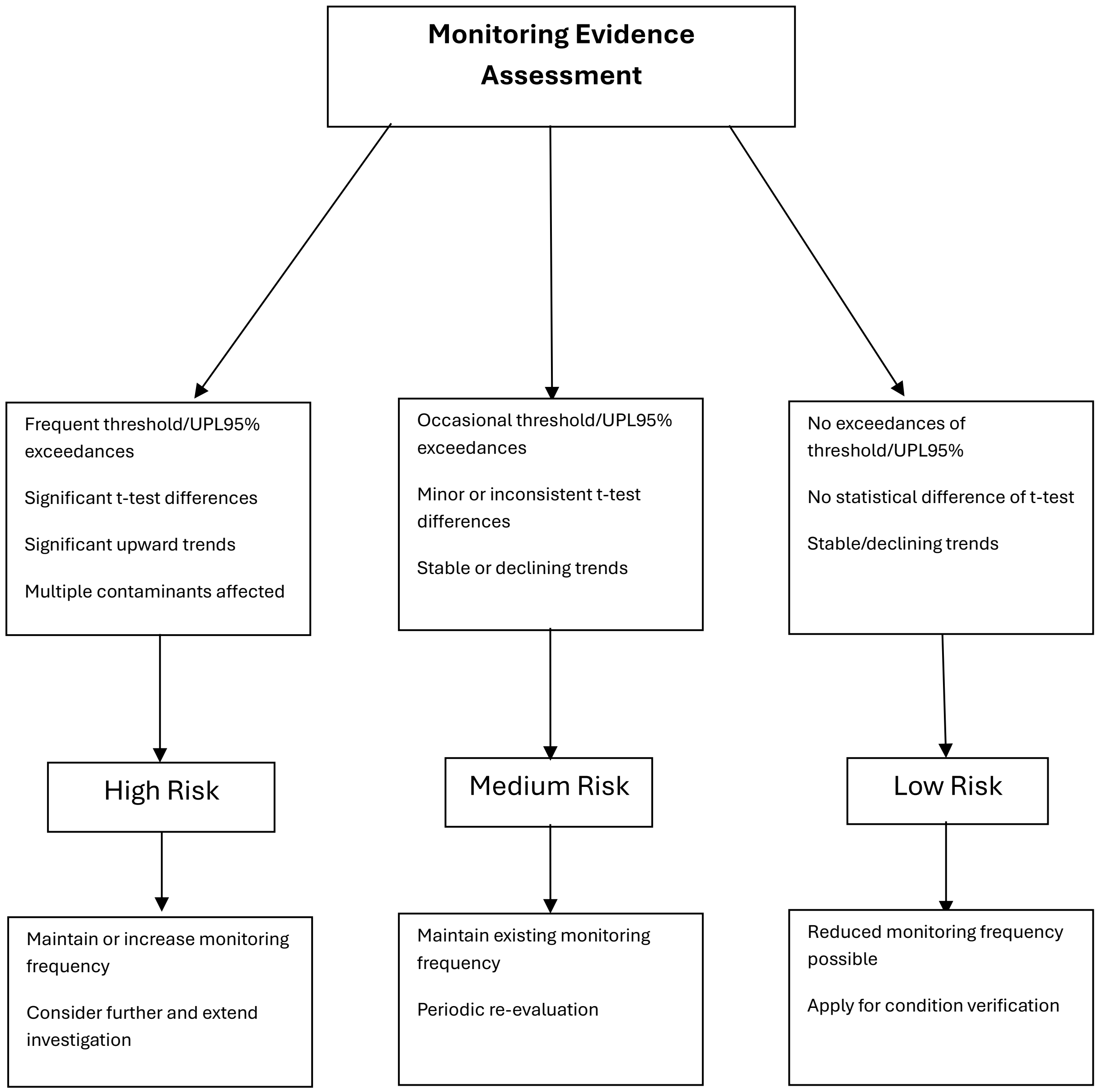


Figure 3: Proposed Decision-making flowchart 2



2.2.2 Source-pathway-receptor framing

As noted in Section 2.1, a risk-based methodology for evaluating closed landfills necessitates the systematic integration of four core elements: contaminant sources, containment integrity, migration pathways, and sensitive receptors. The decision framework, presented as a flowchart (Figures 2 and 3), provides a structured and transparent process for determining environmental risk levels and corresponding monitoring requirements. This section details each component of the framework, outlining the requisite monitoring evidence for a scientifically defensible risk classification (Oyebamiji, 2024; Environment Agency, 2025). The sequential logic of the framework progresses as follows: Source → Containment → Pathway → Receptor → Monitoring Evidence → Outcome.

Sources

The risk evaluation is initiated by confirming the presence of a waste source capable of generating leachate. The scope of this risk evaluation is limited to sites containing municipal waste, mixed refuse, or other leachate-generating materials. Sites with no waste present are classified as low risk, warranting minimal oversight. For locations containing municipal, industrial, or mixed refuse, leachate generation remains a persistent possibility through precipitation percolation and the ongoing biochemical degradation of residual organic matter, even in closed landfills (Christensen, 2001).

Containment

Engineered containment systems, including liners, caps, and leachate collection infrastructure, serve as primary barriers against contaminant migration. Their confirmed integrity significantly reduces environmental risk. However, many older landfills, particularly those closed prior to modern regulatory standards, lack such systems or possess barriers that may have degraded over time (El-Fadel, 1997). This is a common scenario in New Zealand, where numerous facilities were closed before comprehensive liner and leachate management requirements were established in the early 2000s. Where containment is absent, compromised, or uncertain, the assessment must proceed to evaluate potential migration pathways.

Pathways

This stage evaluates the physical mechanisms for contaminant migration from the source. Groundwater advection is the most prevalent pathway, with leachate infiltration and subsequent transport dictated by subsurface permeability and hydraulic gradients (Farquhar, 1989). Additional pathways include surface water runoff, preferential flow through fractures, and gas migration. The absence of a viable pathway, for instance, where impermeable geology isolates the waste or hydraulic gradients direct flow away from sensitive areas, results in a low-risk classification. Conversely, the identification of a functional pathway necessitates evaluation in conjunction with potential receptors.

Identification of sensitive receptors

Sensitive receptors include groundwater resources used for drinking water abstraction, ecologically significant surface water bodies, wetlands, stock watering, irrigation uses and private or community water supplies. The proximity and vulnerability of such receptors elevate the potential consequence of any contaminant release. Historical evidence demonstrates that closed, unlined landfills can impact groundwater quality decades after closure, particularly where shallow aquifers are utilised for supply (Slack, Gronow, & Voulvoulis, 2005). Therefore, risk classification must explicitly consider receptor vulnerability and the potential magnitude of effects if contaminants reach these environments.

2.2.3 Monitoring network design principles

The assessment included the analysis of groundwater quality monitoring to distinguish landfill impacts from natural hydrogeochemical variability.

Monitoring networks were critically evaluated based on the fundamental upgradient-downgradient logic. Upgradient bores were used to establish background groundwater quality, while downgradient bores were positioned to detect potential leachate influence, requiring proper alignment with groundwater flow direction for valid interpretation (Helsel, 2020). Background water quality was established using long-term data from upgradient locations. Where hydrogeological complexity introduced uncertainty, greater emphasis was placed on conservative leachate tracers, such as chloride and potassium (Christensen, 2001).

2.2.4 Statistical methods

Statistical thresholds were employed to objectively identify exceedances. In situations where upgradient monitoring bores are not available to establish background groundwater quality, an alternative statistical approach is required. In this context, the 95% Upper Prediction Limit (UPL95% was used to define the upper bound of expected background groundwater quality. The UPL95% is used in environmental monitoring to define a statistical upper bound that future observations are expected to stay below 95% of the time. UPL95% values were calculated from historical monitoring data for each parameter to represent baseline conditions.

$$\mathrm{UPL}_{95\%} = \bar{x} + t_{0.95,\,n-1} \cdot s \cdot \sqrt{1 + \frac{1}{n}}$$	$\bar{x}$	Sample mean
	s	Sample standard deviation
	n	Number of samples
	$t_{0.95,\,n-1}$	One-sided t-value (95%, df = n−1)

This approach accounts for natural background variability and provides a conservative indicator for identifying potential anthropogenic impacts on groundwater quality. Where available, consent trigger levels were also applied to identify concentrations warranting further investigation. Although monitoring bores are not drinking-water sources, New Zealand Drinking Water Standards were referenced to contextualise potential risk to any downgradient potable supplies. The New Zealand Drinking Water Standards (NZDWS) were revoked in 2018; however, they are referenced in this assessment to maintain continuity with historical groundwater monitoring data and to enable comparison with long-established numeric guideline values. The NZDWS have since been replaced by the Drinking Water Standards for New Zealand under the Water Services Act 2021, which continue to specify numeric drinking-water quality criteria for key parameters (Water Services Act, 2021). Accordingly, the NZDWS are referenced in this assessment for consistency with historical monitoring records. Further discussion about the selection of the trigger levels, and the values chosen for these landfills, is provided in Section 2.3

Quantitative analysis supported the evaluation of spatial differences, temporal trends, and threshold exceedances.

Independent sample t-tests were applied, where dataset assumptions were met, to identify statistically significant differences between upgradient (background) and downgradient concentrations (Helsel, 2020). Trend analysis via linear regression was used to assess long-term contaminant behaviour, with the slope and coefficient of determination (R^2) informing classifications of increasing, stable, or decreasing concentrations—a key indicator of landfill stabilisation (Kulikowska & Klimiuk, 2008). The threshold exceedance logic systematically recorded instances where concentrations surpassed UPL95% values or consent triggers, with repeated exceedances indicating a more persistent source strength.

When a source, pathway, and receptor are identified, monitoring data provide the primary evidence used to determine the final environmental risk classification for closed landfill sites. A multi-line-of-evidence approach is employed, integrating exceedance assessment, statistical testing, and trend analysis to determine whether groundwater quality is deteriorating, stable, or improving. The following expanded criteria refine the decision-making pathway for assigning a site to high, medium, or low-risk categories.

A defensible monitoring network must be purposefully designed to distinguish landfill signals from natural background variation. This requires:

- Upgradient wells to characterise natural background groundwater quality.
- Downgradient wells positioned hydraulically down-gradient of the waste source to detect potential impacts.

- Where true upgradient wells are unavailable, calculated statistical thresholds such as the UPL95% can be used to establish a background benchmark.

Statistical Comparison

- Prior to statistical analysis, the groundwater monitoring dataset was reviewed for data quality and internal consistency. The isolated anomalous results were retained in the analysis to avoid introducing subjective bias and to ensure that the full range of observed groundwater conditions was represented. Where short-term spikes were present, these were interpreted cautiously and discussed in the context of potential analytical uncertainty or transient environmental influences.
- Statistical comparison between monitoring results and nominated threshold values was undertaken using a two-tailed t-test. This test was applied to determine whether the mean concentration of each parameter, calculated across the data from 2016 to 2023for each bore, was statistically distinguishable from the relevant threshold value, and whether the monitoring mean was higher or lower than that threshold. This approach provides a consistent and transparent method for assessing overall compliance and identifying parameters that may warrant closer scrutiny. It is acknowledged that assessing the mean across the entire dataset may mask short-term or recent changes in groundwater quality; however, where the long-term mean remains well below threshold values, this also implies that more recent data are unlikely to exceed those thresholds. Where appropriate, recent trends and individual elevated results are therefore discussed separately alongside the statistical outcomes.
- For the site, only a single downgradient monitoring bore is available. Utilizing one-sample t-tests, downgradient concentrations were compared against regulatory trigger values or UPL95% thresholds. The two-sample t-tests were used where comparisons between two independent datasets were available.
- Temporal trends were assessed using simple linear least-squares regression applied to the raw (non-transformed) monitoring data. Regression equations and coefficients of determination (R^2) are presented to provide an indication of the direction and strength of any apparent temporal trends. More advanced trend-testing methods, such as the Seasonal Mann–Kendall test, were not applied, as there was no consistent evidence of seasonal variability in the groundwater monitoring data. In cases where variability was observed, the timing and magnitude of changes did not display repeatable seasonal patterns and were more likely attributable to factors such as natural hydrochemical variability, changes in sampling conditions, or analytical differences between laboratories. Consistent with practical experience in long-term monitoring programmes, some apparent outliers may therefore reflect laboratory-related variability rather than true changes in groundwater quality.

- For the purpose of calculating means and other summary statistics for each record, any non-detects were treated as zero.

2.2.5 Risk outcomes

Risk categories were assigned using a structured interpretation matrix:

- High Risk was assigned where clear downgradient impacts, repeated threshold exceedances, strong contaminant pathways, or sensitive receptors were present.
- Medium Risk was indicated by moderate or intermittent exceedances, ambiguous hydrogeological pathways, or localised receptor sensitivity.
- Low Risk was concluded for sites with stable or declining trends, minimal exceedances, weak pathways, and distant or absent sensitive receptors.

Further detail of evidential and related factors considered for each of these are as follows.

High Risk – A site is classified as high risk when multiple, independent lines of monitoring evidence indicate that landfill leachate is exerting a measurable and sustained influence on downgradient groundwater quality. This classification reflects a convergence of statistical, chemical, and hydrogeological indicators demonstrating active and potentially progressing contamination. Supporting evidence could include:

- Frequent threshold exceedances: downgradient monitoring reveals repeated exceedances of the UPL95% or other thresholds, derived from background conditions. Exceedances across multiple leachate indicators (e.g., electrical conductivity, chloride, ammoniacal nitrogen) suggest a broad chemical impact consistent with persistent leachate influence beyond natural hydrochemical variability (US EPA, 2009).
- Significant statistical differences: robust inter-well comparisons, such as single-sample t-tests, show statistically significant elevations ($p \leqslant 0.05$) in contaminant concentrations in downgradient wells relative to background. This provides quantitative evidence of a spatially structured leachate impact (Przydatek & Kanownik, 2021; (Magda, 2015).
- Clear upward temporal trends: trend analysis demonstrates statistically significant monotonic increases or positive linear regression slopes with meaningful coefficients of determination (R^2) for key leachate indicators. This upward trajectory across multiple monitoring periods signals an intensifying or migrating contaminant plume.

<u>Risk implication</u>: sites classified as high risk would require immediate management attention. Monitoring frequency should be maintained or increased to characterise plume behaviour, and engineering or operational interventions may be necessary to mitigate contamination. This classification typically triggers regulatory scrutiny and may necessitate formal remedial action planning.

Medium Risk – applies where monitoring data confirm a detectable leachate influence, but the system exhibits stability, attenuation, or controlled conditions. Contaminant levels may reflect a legacy impact rather than an actively expanding plume. Supporting evidence could include:

- Occasional threshold exceedances: exceedances of nominated trigger values or UPL95% occur intermittently but do not form a persistent or escalating pattern. Most parameters remain within the range of natural background variability.
- Small or inconsistent statistical differences: while statistically significant differences for some contaminants may be present, the associated effect sizes are small, indicating limited practical environmental significance (Huang, et al., 2024).
- Stable or declining temporal trends: trend analysis shows non-significant ($p > 0.05$) or negative slopes, reflecting stable equilibrium conditions or effective natural attenuation processes.

Risk implication: for medium-risk sites, monitoring should continue at the current frequency to verify ongoing stability. Periodic reassessment is recommended to detect any future deterioration. Proactive engineering interventions are generally not required unless monitoring trends indicate a transition to a higher-risk state.

Low Risk – supporting evidence could include:

- No threshold exceedances: all monitored parameters remain below site-specific UPL_{95} values or relevant guideline thresholds.
- No significant inter-well differences: statistical testing reveals no meaningful difference ($p > 0.05$) between upgradient and downgradient groundwater quality.
- Stable or declining trends: trend analyses yield non-significant or negative slopes across all key parameters, indicating stable or improving conditions.
- Intact containment systems: site inspections confirm the integrity of caps, covers, or other containment features, supporting the interpretation of effective source isolation.

Risk implication: low-risk sites may qualify for a reduced monitoring frequency (and/or a reduced set of parameters measured), provided a periodic verification sampling programme is maintained to detect any future change. Reduced monitoring is contingent upon stable hydrogeological conditions and the absence of new potential risk sources.

2.3 Selection of trigger levels

Trigger levels for groundwater quality assessment were established in accordance with individual resource consent conditions, which mandate the use of site-specific thresholds to detect meaningful changes in groundwater chemistry. Where relevant values are available, NZDWS were adopted as the primary human health benchmark, consistent with the hierarchical selection of guideline values for water quality protection in New Zealand. Although the NZDWS were revoked in 2018, they are referenced in this assessment to maintain continuity with historical groundwater monitoring datasets and long-established numeric guideline values. The NZDWS have since been replaced by the (Water Services (Drinking Water Standards for New Zealand) Regulations, 2022) which continue to specify numeric drinking-water quality criteria for key parameters. Accordingly, trigger levels adopted in this study are consistent with the replacement regulations. For example, the drinking-water standard for nitrate of 11.3 mg/L as NO_3-N (Water Services (Drinking Water Standards for New Zealand) Regulations 2022) was used as a threshold in this work (see Table 4). For parameters not addressed by the drinking-water standards, site-specific trigger levels were derived from historical groundwater monitoring data, using the UPL95% of the long-term record.

For site-specific thresholds, trigger values for each monitoring well were calculated as the historical mean plus three standard deviations (mean + 3 SD) of the dataset. This approach quantifies the natural variability inherent in baseline groundwater quality and places the threshold at a level that is statistically unlikely to be exceeded by random fluctuations alone, thereby providing an early warning of potential anthropogenic influence beyond natural background conditions. The selection of mean + 3 SD is supported by statistical monitoring practice, where control limits based on three standard deviations approximate a 99.7 % confidence interval for normally distributed data, making it a conservative threshold for identifying deviations from baseline conditions (Gove, Oates, & Archibald, 2015; WasteMINZ, 2023). This method is frequently used in environmental trigger-setting frameworks to balance sensitivity to change with avoidance of false positives that may arise from normal temporal variability.

Overall for this study, trigger values for chloride, total ammoniacal nitrogen, nitrite, and nitrate were based on the relevant NZDWS health-based or aesthetic guideline values where available. All other monitoring parameters, which lack specific NZDWS limits, were assessed against site-specific trigger levels calculated as mean + 3 SD. This combined approach ensures that human health benchmarks are applied where appropriate, while site-specific statistical thresholds provide robust detection of anomalous changes in groundwater quality consistent with consent requirements.

Table 4: Selected Trigger levels for each landfill sites

Upgradient Well	Downgradient Well

	Unit	Waipawa		Waipukurau	Tamumu	Ongaonga	Porangahau	Tikokino			Takapau
Well number		**5686**	**5501**	**5503**	**5504**	**5356**	**5661**	**4228**	**4227**	**5662**	**5660**
Monitoring parameters											
pH		8.6	7.1	7.8	7.3	7.7	7.7	7.7	7.4	7.3	7.5
Electrical Conductivity	**µS/cm**	283.6	121	60.3	73.2	26.7	174.1	40.5	25.6	106.8	72.3
Chloride	**mg/L**	250	250	250	250	250	250	250	250	250	250
Total Ammoniacal Nitrogen	**mg/L**	0.41	0.41	0.41	0.41	0.41	0.41	0.41	0.41	0.41	0.41
Nitrite-N	**mg/L**	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9
Nitrate-N	**mg/L**	11.3	11.3	11.3	11.3	11.3	11.3	11.3	11.3	11.3	11.3
Nitrate-N + Nitrite-N	**mg/L**	4.6	1.7	3.6	1.9	5.6	0.9	1.3	14.4	5.2	5.9
Dissolved Potassium	**mg/L**	5.4	13.2	4.3	12.7	4.8	5.3	4.4	3.3	3.4	22.4
Total Organic Carbon (TOC)	**mg/L**	28.7	20.7	9.2	21.9	11.5	19.2	15.7	7.6	19.5	12.8
Volatile Fatty Acids (VFA)	**mg/L**	-	-	14.8	-	16.9	8.6	-	15.9	12.9	28.7
Absorbance at 254 nm		0.7	0.4	0.1	0.3	0.5	0.1	1.7	0.2	0.8	0.7

pH, Electrical Conductivity, Dissolved Potassium, Total Organic Carbon, Volatile Fatty Acids and Absorbance at 254nm - Site specific trigger level

Chloride and Total Ammoniacal Nitrogen - Drinking Water Aesthetic values (Note the TAN value of 0.41 mg/L originated from the Australian Guidelines for Drinking Water – Groundwater Sampling and Analysis: A Field Guide, and was historically used by CHBDC and HBRC (Sundaram, Wallace, Feitz, & Caritat, 2009)**.)**

Nitrite-N and Nitrate-N - Drinking Water Health values

3. Results and discussion

3.1 Preliminary risk characterisation

Based on the information presented in Sections 1.2.6 and 1.4, a first-level risk matrix was developed (Table 5) to evaluate and justify the level of environmental risk associated with each closed landfill site. The risk assessment is primarily based on the spatial relationship between the contamination source and the nearest potential receptors, including the closest surface waterbody and the nearest potable water supply.

Distances between the landfill source and these receptors were used as key risk determinants, reflecting the likelihood of contaminant migration and potential exposure. The resulting risk rankings derived from the matrix are consistent with, and directly support, the outcomes of the decision-making flowchart applied in this study. This alignment demonstrates that the risk matrix provides a coherent and defensible framework for classifying site-specific risks and informing subsequent monitoring and management decisions.

All seven closed landfill sites were assessed as Low Risk overall. Drinking water bores near the sites range from under 500 meters to about 4 kilometres away. In most cases, these bores are situated upstream, perpendicular to, or offset from the main groundwater flow. Even where a drinking water bore is relatively close downstream, such as at Tamamu, major surface water features like the Tukituki River provide significant dilution and act as a hydraulic barrier, greatly reducing the potential for landfill impacts.

While surface water is often found within 100 metres of the landfills, historical groundwater data consistently shows good quality. Any observed variations are largely due to natural factors, like river interaction or nearby farming, not landfill contamination. No site shows persistent or systematic groundwater deterioration.

Overall, the combination of adequate distance to sensitive receptors, favourable groundwater flow directions, and consistently clean historical monitoring data supports a Low-Risk rating for every site. The likelihood of any adverse effect on downstream groundwater or surface water is low. The likelihood of adverse impacts on downstream groundwater users or surface waters is minimal.

However, this level of assessment is preliminary. It does not preclude the possibility that a detailed assessment of monitoring data (Section 3.2) and application of the proposed framework (Section 3.4) could result in different outcomes for one or more of the landfills.

Table 5: Risk Matrix

Low Risk	Moderate Risk	Significant Risk

Site name	Upstream bore	Downstream bore	Distance to nearest drinking water bore	Nearest Downstream Bore (all types)	Nearest Surface Water	Nearest Downstream Surface Water	Historical Groundwater Quality
Waipawa	5686	5501	2.3 km	2.3 km	< 100 m	< 100 m	variable but still consistently good results
Waipukurau	-	5503	< 1000 m	3.3 km	< 100 m	> 500 m	consistently good
Ongaonga	-	5356	3 km	> 500 m	600 m	600 m	consistently good
Porangahau	-	5661	4 km	< 1 km	< 100 m	< 100 m	consistently good
Tikokino	4228	5662 & 4227	2.3 km	2.2 km	< 100 m	< 100 m	consistently good
Tamamu	-	5504	< 500 m	< 500 m	< 100 m	< 100 m	variable but still consistently good results
Takapau	-	5660	1 km	3.4 km	< 100 m	< 200 m	consistently good

3.2 Monitoring data analysis

3.2.1 Waipawa Closed Landfill

Figure 4: GIS map - Waipawa Closed Landfill

Groundwater at the Waipawa Closed Landfill is monitored at two wells: bore 5686, located west of the landfill, is interpreted as the upgradient reference well, and bore 5501, located east of the site, is considered the primary downgradient monitoring point. This interpretation is based on the regional groundwater flow direction, which is inferred to be from west to east across the site, and the relative positions of the bores to the landfill footprint and the Waipawa River. The additional wells (5146 and 5147) were installed for initial site characterisation but are not part of the current routine monitoring programme. The analysis integrates temporal trend behaviour with statistical comparisons against nominated threshold values to evaluate the magnitude and direction of potential effects.

Time-series monitoring results for the nine targeted variables are shown in Figures 5 and 6. Means, standard deviations, and t-tests results are provided in Tables 7 and 8. For each variable, the t-tests was used to determine whether there was a statistical difference between the mean of monitoring results and the specified threshold value. Statistically significant ($p<0.05$) negative t-values denote cases where the mean of the monitoring results is distinguishably less than the threshold. Linear regression equations and their associated statistical measures are provided in Table 6.

Table 6: trend equation and R^2 value for monitoring wells - Waipawa Closed landfill. Well 5686 is upgradient, and well 5501 is downgradient.

	Well 5686		Well 5501	
	Equation	R^2 value	Equation	R^2 value
pH	y = 0.011x + 8.07	R^2 = 0.432	y = 0.0005x + 6.5815	R^2 = 0.0004
Electrical Conductivity (EC)	y = 0.1077x + 168	R^2 = 0.001	y = 1.2512x + 50.541	R^2 = 0.3517
Dissolved Potassium	y = -0.0144x + 3.4542	R^2 = 0.0175	y = 0.0555x + 7.1734	R^2 = 0.0682
Chloride	y = 2.63x + 141.04	R^2 = 0.0883	y = 0.0427x + 36.571	R^2 = 0.0082
Total Ammoniacal-N	y = -0.0052x + 0.7203	R^2 = 0.0148	y = -0.004x + 0.9881	R^2 = 0.0069
Nitrite-N	y = -0.0006x + 0.1081	R^2 = 0.0033	y = 0.0002x - 0.0009	R^2 = 0.2157
Nitrate-N	y = -0.0637x + 2.213	R^2 = 0.2832	y = 0.0051x + 0.0101	R^2 = 0.0457
Nitrate-N + Nitrite-N				
Total Organic Carbon (TOC)	y = 0.0745x + 3.7207	R^2 = 0.0161	y = 0.1239x + 5.6774	R^2 = 0.0566
Volatile Fatty Acids (VFA)	-	-	-	-
Absorbance at 254 nm	y = -0.0005x + 0.1208	R^2 = 0.0015	y = -0.0007x + 0.1956	R^2 = 0.0184

Table 7: Summary statistics and t-test results for Well No. 5686, the upgradient well. The t-tests were between the mean of the monitoring results, and the specified threshold value.

	Unit	n	Mean	Standard Deviation	Threshold	t-value	p-value
pH		27	8.23	0.14	8.6	-14.04	<0.0001
Electrical Conductivity	µS/cm	27	170	28.9	283.6	-20.47	<0.0001
Dissolved Potassium	mg/L	27	3.24	0.95	5.4	-11.87	<0.0001
Chloride	mg/L	27	181	77	250	-4.68	0.0001
Total Ammoniacal-N	mg/L	27	0.78	0.36	0.41	5.41	<0.0001
Nitrite-N	mg/L	27	0.10	0.08	0.90	-49.01	<0.0001
Nitrate-N	mg/L	27	1.25	1.04	11.3	-50.09	<0.0001
Total Organic Carbon (TOC)	mg/L	27	7.26	4.61	28.7	-24.15	<0.0001
Volatile Fatty Acids (VFA)	mg/L	27	-	-	-	-	-
Absorbance at 254 nm		27	0.11	0.10	0.7	-28.03	<0.0001

Table 8: Summary statistics and t-test results for Well No. 5501, the downgradient well. The t-tests were between the mean of the monitoring results, and the specified threshold value.

	Unit	n	Mean	Standard Deviation	Threshold	t-value	p-value
pH		28	6.59	0.23	7.1	-12.40	<0.0001
Electrical Conductivity	µS/cm	28	69.1	18.2	121	-15.10	<0.0001
Dissolved Potassium	mg/L	28	8.00	1.83	13.2	-14.99	<0.0001
Chloride	mg/L	28	34	10.05	250	-113.62	<0.0001
Total Ammoniacal-N	mg/L	28	0.93	0.41	0.41	6.64	<0.0001
Nitrite-N	mg/L	28	0.01	0.02	0.90	-213.38	<0.0001
Nitrate-N	mg/L	28	0.22	0.58	11.3	-101.23	<0.0001
Total Organic Carbon (TOC)	mg/L	28	7.79	4.32	20.7	-15.81	<0.0001
Volatile Fatty Acids (VFA)	mg/L	28	-	-	-	-	-
Absorbance at 254 nm		28	0.19	0.04	0.4	-19.68	<0.0001

Figure 5: Monitoring data for Well No: 5686 (upgradient):

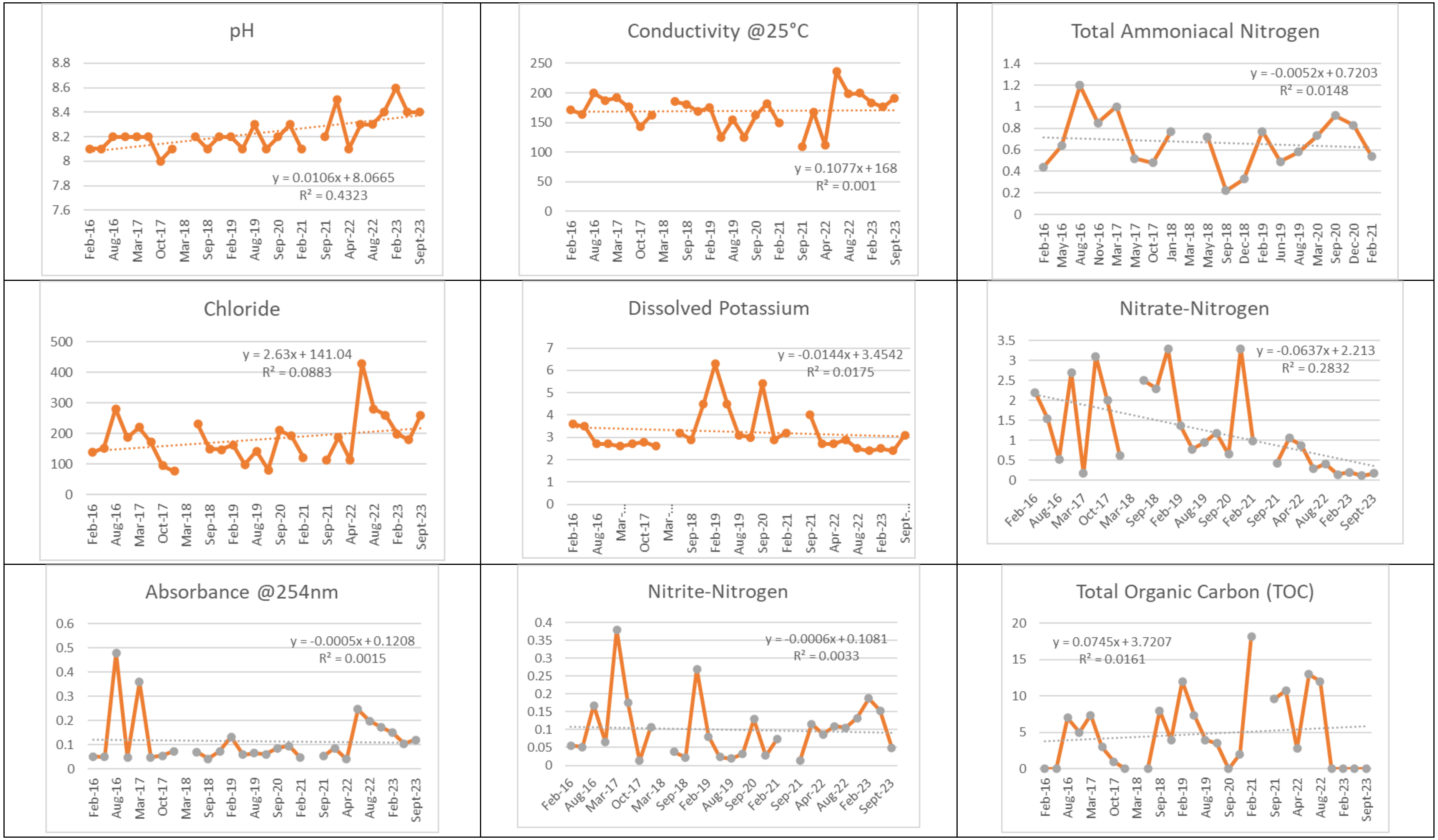


Figure 6: Monitoring data for Well No: 5501 (downgradient):

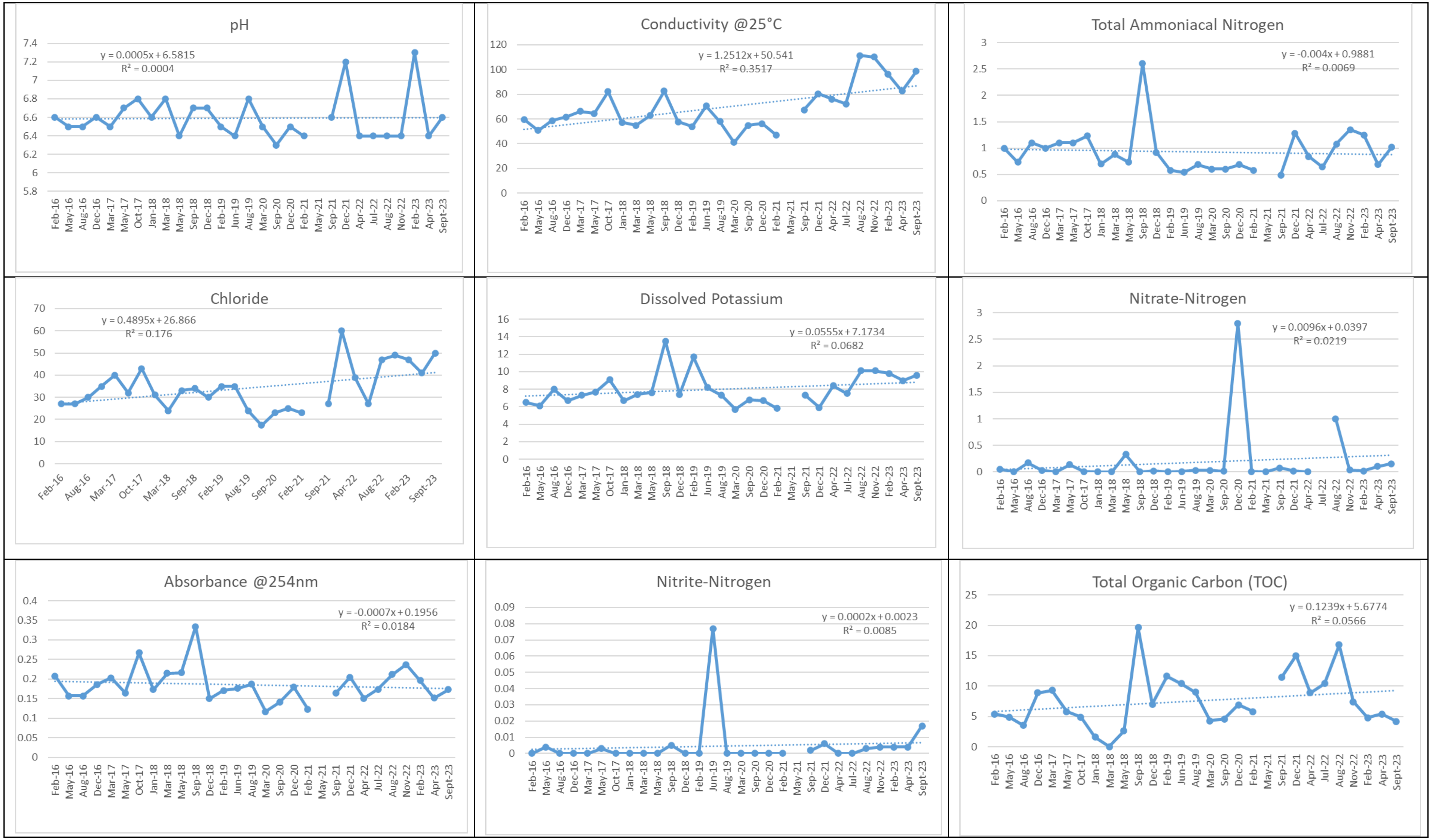
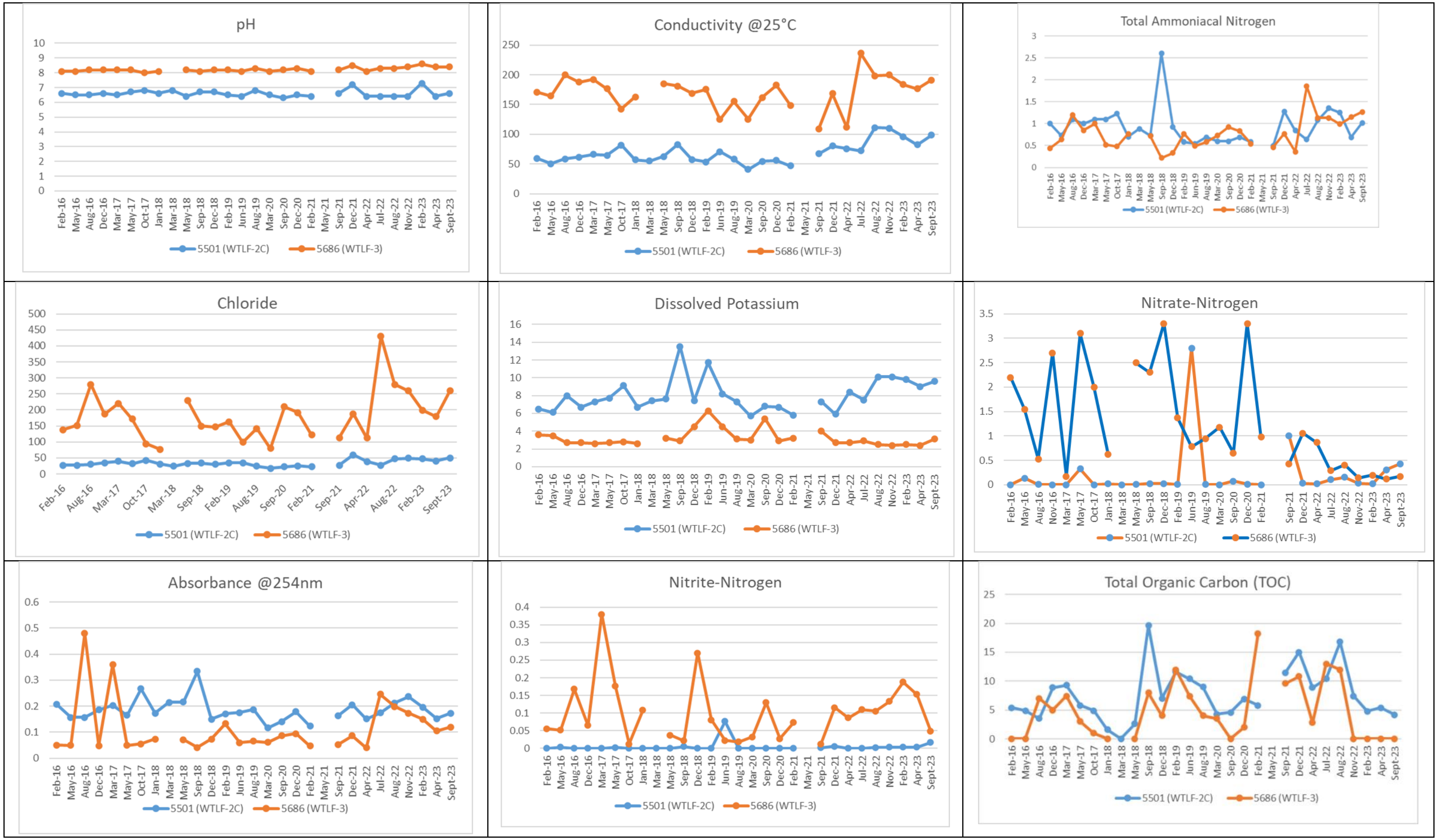


Figure 7: Comparing data - 5686 (WTLF-3) and 5501 (WTLF-2C):



For pH, the upgradient well shows a modest increasing trend ($y = 0.011x + 8.07$; $R^2 = 0.432$, $p<0.001$) (Figure 5), reflecting a reasonably strong upward trajectory over time. In contrast, the downgradient bore (Figure 6) exhibits no evidence of any trend ($R^2 = 0.0004$), indicating no meaningful temporal change. Mean pH at the upgradient well falls within the aesthetic drinking water range of 7.00 - 8.50, while the downgradient well is slightly below this range. Both wells remain well below the threshold value of pH 8.6, supported by strongly negative t-values (upgradient: $t = -14.04$, $p<0.0001$; downgradient: $t = -12.40$, $p<0.0001$). Overall, the upgradient pH is higher than the Central Hawke's Bay groundwater background concentration, whereas the downgradient pH is lower, consistent with minimal leachate influence. The reason for an upward trend in the upgradient well is unknown, but may possibly relate to longer term land-use factors working through the groundwater system, such as the use of agricultural lime

Electrical conductivity (EC) shows a marked contrast between the two monitoring locations. At the upgradient reference bore (5686), EC remains effectively constant over the monitoring period ($p<0.0001$), with a mean value of 170 µS/cm (Table 7). In contrast, the downgradient bore (5501) displays a substantially lower mean EC of 69.1 µS/cm (Table 8), approximately 2.5 times lower than that observed at the upgradient site. This pattern is atypical for a landfill monitoring setting, where elevated EC would ordinarily be expected downgradient if leachate were influencing groundwater quality. The linear regression analysis of the EC record at bore 5501 indicates a possible upward trend ($y = 1.25x + 50.54$; $R^2 = 0.352$, $p<0.0001$). However, interpretation of this trend requires careful consideration of the full temporal context shown in Figure 6. Over the majority of the monitoring record, EC values at this location remain relatively stable and clustered within a narrow range, with no clear directional change. If the record had ended in February 2021, no upward trend would be evident, and even with a modest extension of the dataset beyond this point, EC behaviour would still be characterised as stable. The apparent trend identified by regression analysis is driven primarily by a sequence of higher EC values recorded since August 2022, comprising the most recent five to nine data points. In this context, EC behaviour at the downgradient bore is more appropriately described as indicating a possible upward trend emerging in the later part of the record, rather than a consistent or well-established long-term trend.

The consistently higher EC observed at the upgradient reference bore suggests that factors unrelated to landfill leachate, such as agricultural land use, regional groundwater chemistry, or natural geological variability, are likely to exert a stronger influence on EC at this site.[6] Importantly, mean EC values at both monitoring locations remain well below their respective threshold values (283.6 µS/cm for the upgradient bore and 121 µS/cm for the downgradient bore). This is further supported by strongly negative one-sample t-test

[6] It should be noted that another possibility may be that despite its council classification, this bore may not be genuinely upgradient. For the purpose of this thesis discussion, it is assumed to be upgradient.

results comparing monitoring means with threshold values (t = -20.47, $p<0.0001$ for the upgradient bore; t = -15.10, $p<0.0001$ for the downgradient bore).

Chloride concentrations at both wells (means 181 mg/L upgradient, 34 mg/L downgradient)) are higher than the CHB background value of 10.3 mg/L. Once again, the upgradient well shows the higher of the two values. The time-series plots (Figures 5 and 6) with chloride values at both wells exhibiting moderate variability around relatively stable long-term means rather than sustained directional change. Importantly, mean chloride concentrations at both monitoring locations remain comfortably below the aesthetic threshold value of 250 mg/L ($p=0.038$ upgradient, and $p<0.0001$ downgradient). The linear regression analysis indicates that the least-squares trend lines for chloride at both monitoring locations are positive; however, neither trend is statistically significant. For the paired dataset (n=27), the coefficient of determination ($R^2= 0.0883$) corresponds to a two-tailed p-value of 0.132, which exceeds the conventional significance threshold of $p=0.05$. Even when testing specifically for a monotonic upward trend using a one-tailed test, the p-value remains above the significance threshold ($p=0.07$). Therefore, while the fitted regression lines slope upward at both locations, the data do not provide reliable statistical evidence of a temporal increase in chloride concentrations.

Dissolved potassium occurs at low concentrations and shows no statistically significant temporal pattern. Mean concentrations at both locations remain well below their respective thresholds (5.4 mg/L upgradient and 13.2 mg/L downgradient). The t-tests further confirm this, with significant negative t-values at both the upgradient (t = -11.87; $p<0.0001$) and downgradient wells (t = -14.99; $p<0.0001$). Interestingly, unlike pH, EC, or chloride, downgradient concentrations of potassium are consistently higher than the upgradient values (Figure 7).

The three forms of nitrogen included in the monitoring are nitrate-N (oxidation state +5, and commonly the dominant form), nitrite-N (an unstable intermediate oxidation level of +3, usually present only in trace amounts), and ammoniacal-N (NH4+ in which nitrogen has an oxidation state of -3). Ammoniacal-N is regarded as most toxic to aquatic organisms, and can be a marker of landfill leachate. In normal soils, it is typically much less abundant than nitrate, because it is quickly nitrified under aerobic conditions. These three forms of nitrogen will be considered in oxidation order.

Following the usual pattern, nitrate-N shows the highest concentrations of the three forms of nitrogen, though in absolute terms these are not very high. As for pH, EC, chloride, mean nitrate-N is higher at the upgradient well (1.25 mg/L) than the downgradient well (0.22 mg/L). This adds further support for the concept that the upgradient well is influenced by land-use factors that are unrelated to the landfill. Agricultural production is a well-known diffuse source of nitrate in rural groundwater. No obvious trends are evident in the data records (Figures 5 and 6), but some apparent outliers exist, which point to the possibility of laboratory error on some sampling days.

Mean nitrate levels 1.25 mg/L upgradient and 0.22 mg/L downgradient are well below the drinking-water health guideline of 11.3 mg/L, supported by a highly significant t-value ($t = -50.09$, $p<0.0001$; $t = -101.23$, $p<0.0001$).

As expected, nitrite-N remains very low at both wells, and essentially presents as a subset of nitrate (Figure 7), so that the higher mean levels are in the upgradient well. No time-series trends are evident, and all values sit comfortably below their respective thresholds.

Total ammoniacal-N (TAN) also shows no significant change at either bore, with similar mean concentrations of 0.78 mg/L at 5686 (upgradient) and 0.93 mg/L at 5501 (downgradient), and no statistical difference between these two means. However, in both cases, TAN exceeds the nominated threshold of 0.41 mg/L, even for the upgradient well (Well 5686 $t = 5.4$, $p<0.0001$; Well 5501 $t = 6.64$, $p<0.0001$) (Tables 7 and 8). The similarity between the two records and presence in the upgradient well points to natural biogeochemical processes, and/or other land-use factors, rather than landfill influence, as the likely source of the higher TAN.

TOC and A254 remain consistently low, with no indication of organic leachate influence or any temporal trends (Figure 7). The average TOC concentrations (7.26 mg/L upgradient and 7.29 downgradient) and sit well below their respective thresholds of 28.7 mg/L and 20.7 mg/L (Tables 7 and 8), with t-tests providing strong ($p<0.0001$) confirmation of compliance. In addition, a two-sample (Welch) t-test was undertaken to assess whether mean TOC concentrations differed between the upgradient reference bore (mean = 7.26 mg/L, $n = 27$) and the downgradient bore (mean = 7.29 mg/L, $n = 28$). The difference between means was negligible (-0.03 mg/L) and not statistically significant ($p\approx0.98$), indicating no evidence of downgradient TOC enrichment

Absorbance at 254 nm (A254) (mean values 0.11 mg/L upgradient and 0.19 mg/L downgradient) also remains comfortably beneath their respective aesthetic thresholds (0.7 and 0.4 mg/L). Two interesting aspects of these TOC and Absorbance records are that:

- TOC results appear to be reasonably well correlated (upgradient and downgradient, allowing for missing sampling days), with a possible time delay between upgradient and downgradient results in some parts of the record (Figure 9). This matching suggests that TOC is dominated by regional groundwater conditions. Following heavy rainfall, or seasonal changes in rainfall, TOC in groundwater could increase through more soluble organic matter being leached and decrease through groundwater dilution – so the measured result may reflect a balance between such competing factors. Similar leaching and dilution effects may be behind much of the variability in other measured parameters, such as chloride and potassium.

- Despite this result for TOC, on most (25 of the 27) sampling events, A254 readings were higher in the downgradient well than the upgradient well. This is interesting because both TOC and A254 are measures of dissolved organic matter (DOM) in water. The main difference is that A254 is particularly sensitive to the presence of aromatic compounds, and compounds with unsaturated carbon bonds. The slightly higher downgradient A254 readings on most days might therefore reflect a minor influence from landfill leachate, which is carrying some additional forms of DOM, not present in the upgradient well.

Across the full suite of parameters, groundwater quality at the downgradient well remains consistently low across all indicators, with no parameter exhibiting systematic enrichment relative to upgradient levels. Conservative ion chemistry does not show the characteristic pattern of downgradient elevation typically associated with leachate migration (Slack, Gronow, & Voulvoulis, 2005). Nitrogen species other than TAN are generally low, stable, and internally consistent, and organic indicators such as TOC and A254 remain well below their nominated thresholds. The absence of higher values downgradient indicates a regional hydrochemical influence rather than an active leachate plume. Taken together, these lines of evidence demonstrate that the Waipawa Closed Landfill is exerting minimal to negligible influence on downgradient groundwater quality. Under current conditions, there is no indication of deteriorating water quality or of a developing contaminant plume, and the site therefore presents a low environmental risk.

3.2.2 Waipukurau Closed Landfill

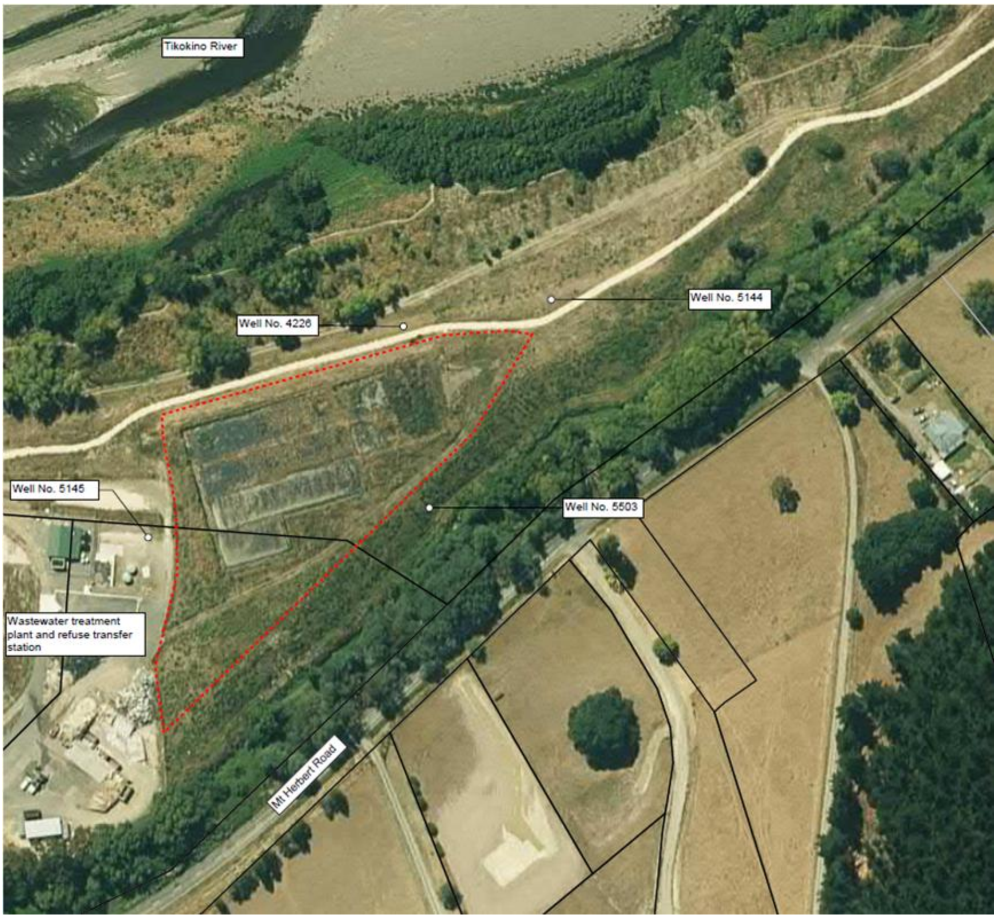


Figure 8: GIS map - Waipukurau Closed Landfill

Groundwater conditions at the Waipukurau Closed Landfill were assessed using long-term monitoring data from a single downgradient bore (Well 5503). No upgradient monitoring bore is available at this site to provide a direct background reference. A potentially significant feature of this site to bear in mind for interpretation is that Well 5503 may also pick up some influence from a wastewater treatment plant and refuse transfer station (Figure 8). Therefore, UPL95% values were adopted as the baseline for comparative assessment of groundwater quality. Historical monitoring bores at the site are no longer available: Well 5145 was lost or buried during the expansion of the adjacent wastewater treatment plant, while Wells 4226 and 5144 have not been monitored for decades and were therefore excluded from the current assessment.

Time-series monitoring results for the ten targeted variables are shown in Figure 9. Means, standard deviations, t-tests results are provided in Table 10. For each variable, the t-tests were used to determine whether there was a statistical difference between the mean of monitoring results and the specified threshold value. Statistically significant ($p<0.05$) negative t-values denote cases where the mean of the monitoring results is distinguishably less than the threshold. Linear regression equations and their associated statistical measures are provided in Table 9.

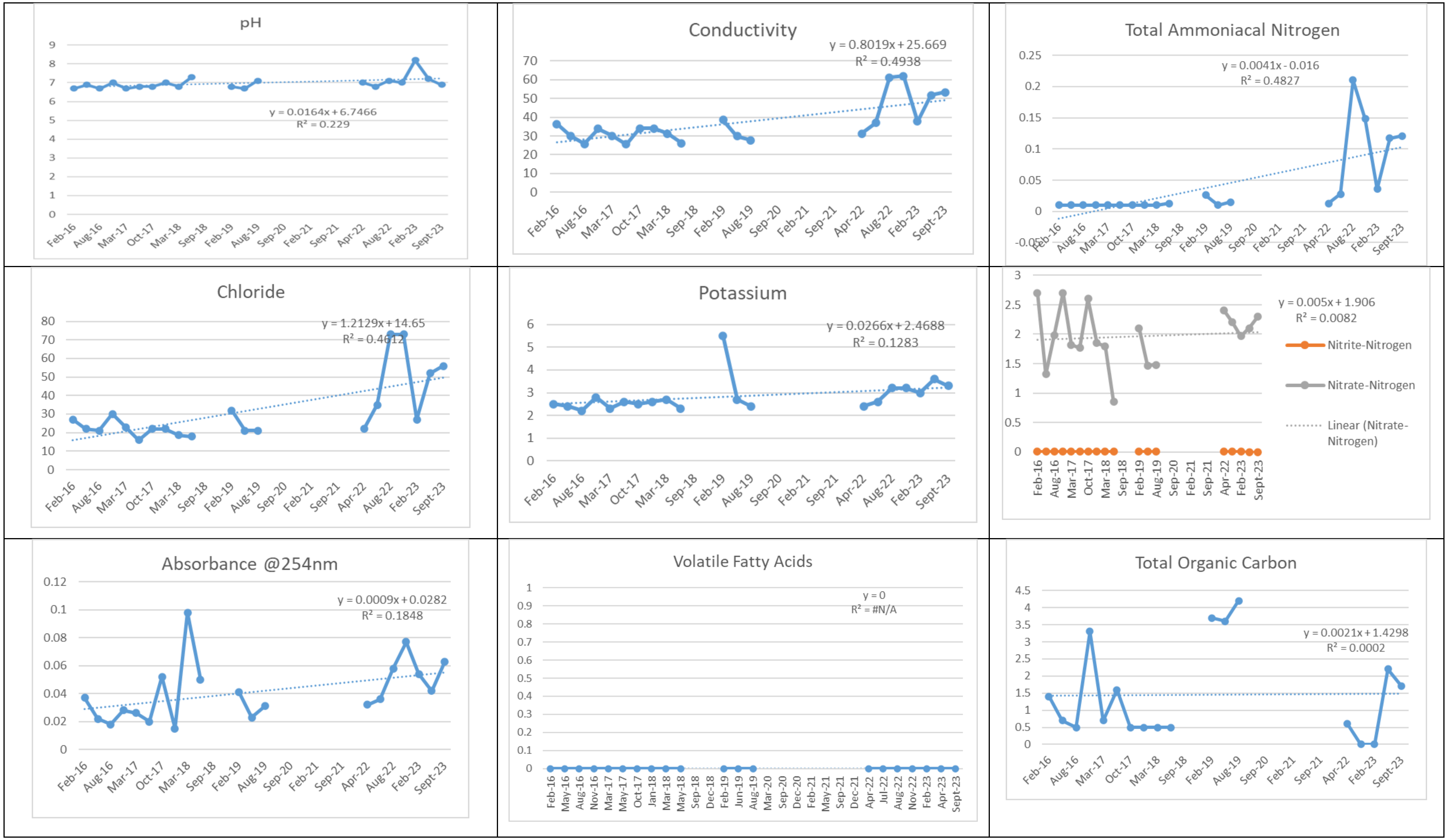
Table 9: Trend equation and R^2 value for monitoring wells - Waipukurau Closed landfill. Well 5503 is downgradient.

	Well 5503	
	Equation	R^2 value
pH	y = 0.0164x + 6.7466	R^2 = 0.229
Electrical Conductivity (EC)	y = 0.8019x + 25.669	R^2 = 0.4938
Dissolved Potassium	y = 0.0266x + 2.4688	R^2 = 0.1283
Chloride	y = 1.2129x + 14.65	R^2 = 0.4612
Total Ammoniacal-N	y = 0.0041x – 0.016	R^2 = 0.4827
Nitrite-N		
Nitrate-N	y = 0.005x + 1.906	R^2 = 0.0082
Nitrate-N + Nitrite-N		
Total Organic Carbon (TOC)	y = 0.0021x + 1.4298	R^2 = 0.0002
Volatile Fatty Acids (VFA)	-	-
Absorbance at 254 nm	y = 0.0009x + 0.0282	R^2 = 0.1848

Table 10: Summary statistics and t-test results for Well No. 5503, the downgradient well and UPL(95%) values. The t-tests were between the mean of the monitoring results, and the specified threshold value.

	Units	n	Mean	Standard Deviation	Threshold	t-value	p-value	UPL(95%)
pH		20	6.98	0.34	7.8	-10.30	<0.0001	7.65
Electrical Conductivity (EC)	µS/cm	20	36.86	11.24	60.3	-9.34	<0.0001	59.44
Dissolved Potassium	mg/L	20	2.84	0.73	4.3	-9.06	<0.0001	4.31
Chloride	mg/L	20	31.57	17.59	250	-55.54	<0.0001	66.92
Total Ammoniacal-N	mg/L	20	0.04	0.06	0.41	-28.31	<0.0001	0.16
Nitrite-N	mg/L	20	-	-	-	-	-	0.00
Nitrate-N	mg/L	20	2.00	0.47	11.3	-87.83	<0.0001	2.95
Total Organic Carbon (TOC)	mg/L	20	1.84	1.54	9.2	-21.47	<0.0001	4.92
Volatile Fatty Acids (VFA)	mg/L	20	-	-	-	-	-	-
Absorbance at 254 nm		20	0.04	0.02	0.1	-15.94	<0.0001	0.08

Figure 9: Monitoring data for Well No: 5503 (downgradient):



For pH, the downgradient monitoring bore at Waipukurau (Well 5503) shows an apparently increasing ($p<0.03$) trend over time, but with a slope that is close to zero (0.016) and one recent outlier Figure 9). Mean pH at this well is 6.98, which (within measurement error) is essentially at the lower bound of the aesthetic drinking water range (7.00 - 8.50). The mean pH is also below both the site-specific threshold value of 7.8 and the UPL(95%) of 7.57, as confirmed by a strongly negative t-value ($t = -10.30$, $p<0.0001$; Table 10). Overall, pH conditions at the downgradient bore are close to neutral and stable. The observed gradual increase in pH, if genuine, may reflect long-term geochemical stabilisation of groundwater associated with the advanced age of the landfill rather than any ongoing leachate influence.

For EC, the linear regression indicates an upward trend ($y = 0.802x + 25.7$; $p<0.001$) (Figure 9), showing a likely rise in EC over the monitoring period. The mean EC concentration at this well is 36.86 µS/cm (Table 10), which is below both the UPL(95%) value of 56.77 µS/cm and the site-specific threshold of 60.3 µS/cm, supported by strongly negative t-value ($t = -9.34$, $p<0.0001$). The linear regression analysis indicates an upward trend, but there is some noise in the record; this increase is not uniform across the entire monitoring period and may be influenced by higher values observed in the more recent samples (Figure 9). Importantly, even with this recent elevation, EC concentrations remain well below levels typically associated with landfill leachate impacts.

Chloride concentrations also show an increasing trend over time. The linear regression indicates an upward trajectory ($y = 1.213x + 14.65$; $R^2 = 0.461$) (Figure 9). As with EC, this trend is influenced primarily by higher values recorded in the more recent monitoring period rather than representing a consistent increase across the full dataset. The similarity of both the patterns and slopes of EC and chloride (0.8 for EC, and 1.2 for chloride, mean value 1.0) suggests that changes in EC readings may be driven by changes in dissolved chloride (Cl^-) ions. Chloride is both abundant as a major ion, and mobile. Although the landfill is a possible source, to see an increase in chloride and EC at this stage of its lifespan would seem inconsistent with known landfill chemistry. It is possible that the increases in EC and conductivity are instead more likely to be attributable to activities at the wastewater treatment plant and refuse transfer station. Importantly, the mean chloride concentration is 31.57 mg/L (Table 10) and remains below the aesthetic drinking water guideline of 250 mg/L and below the UPL(95%) value of 62.74 mg/L, supported by strongly negative t-value ($t = -55.54$, $p<0.0001$) (Table 10). However, assuming the trend continues (an increase of +1.2 mg/L in the average interval between sampling events, or 109 mg/L over 91 months, which is equivalent to 14 mg/L/yr), it would take about 13 years from 2023 to increase from 65 mg/L to 250 mg/L. The apparent trends in EC and chloride are therefore worth further investigation to determine: (a) whether they are genuine, and (b) if an area of the water treatment station and refuse transfer site is acting as a new upgradient source. Although the influence of landfill leachate on the EC and chloride results cannot be ruled out, the landfill's age suggests it is less likely to contribute new EC or chloride at this phase in its lifespan, compared with an active source.

Dissolved potassium, the linear regression shows no evidence of a trend (y = 0.027x + 2.469; R^2 = 0.128, p>0.1) (Figure 9). The mean potassium concentration at this well is 2.84 mg/L (Table 10), which remains well below both the threshold value of 4.3 mg/L and the UPL(95%) value of 4.14 mg/L, supported by a strongly negative t-value (t = -9.06, p<0.0001).

Nitrate-N shows the highest concentrations among the nitrogen species at this site. Mean nitrate-N at this well is 2.00 mg/L (Table 10) and shows no temporal trend (y = 0.005x + 1.91; R^2 = 0.008) (Figure 9). Nitrate-N concentrations remain well below the drinking-water health guideline of 11.3 mg/L and the UPL(95%) value of 2.83 mg/L, supported by a strongly negative t-value (t = -87.83, p<0.0001) (Table 10).

As expected, nitrite-N concentrations remain extremely low throughout the monitoring record, with no detectable values reported (Figure 9). No time-series trends are evident, and all values sit at or below the detection limit.

The regression analysis indicates an upward trend for TAN; however, this increase is not uniform across the entire monitoring period and appears to be influenced primarily by higher values observed in the more recent samples (y = 0.004x − 0.02; R^2 = 0.483) (Figure 9). The mean TAN concentration is 0.04 mg/L (Table 10), which is significantly below both the site-specific threshold of 0.41 mg/L and the UPL(95%) value of 0.14 mg/L, supported by a strongly negative t-value (t = -28.31, p<0.0001).

TOC and A254 remain consistently low at the Waipukurau Closed Landfill. TOC shows no meaningful trend through time (y = 0.002x + 1.43; R^2 = 0.0002), with a mean concentration of 1.84 mg/L (Table 10). This value sits well below both the site-specific threshold of 9.2 mg/L and the UPL(95%) value of 4.56 mg/L, supported by a strongly negative t-value (t = -21.47, p<0.0001). A254 also remains very low, showing only a possible weak upward trend (y = 0.0009x + 0.028, R^2 = 0.185, p = 0.06. The mean A254 value of 0.04 is below both the site-specific threshold of 0.1 and the UPL(95%) value of 0.08, supported by a strongly negative t-test result (t = -15.94, p<0.0001).

Across the full suite of indicators, the downgradient well consistently shows low concentrations. For those parameters showing no or limited evidence of any temporal trend, minor fluctuations occur over time. These are consistent with natural hydrochemical variability and broader catchment-scale influences, rather than landfill impact. The two exceptions are the similar upward trends shown by EC and chloride, which are out of keeping with both the other parameters, and the age of the landfill (which other parameters remain consistent with). Taken together, these lines of evidence demonstrate that the Waipukurau Closed Landfill is exerting minimal to negligible influence on downgradient groundwater quality, but a more recent nearby source may be starting to have an impact. Under current conditions, there is no indication that the landfill is responsible for deteriorating water quality or for a developing contaminant plume, and the site therefore presents a low environmental risk.

However, further investigation is warranted about the potential impact of sources that could be associated with either the waste-water treatment plant or refuse transfer station. The increase in EC and chloride are unexpected, and worth attention because these parameters will be among the earliest indicators of a developing plume, which is likely to involve other contaminants progressing at slower speeds. It is also possible that the recent uptick in ammoniacal nitrogen (TAN) is genuine and linked to the same source.

3.2.3 Takapau Closed Landfill

Figure 10: GIS map - Takapau Closed Landfill

Groundwater conditions at the Takapau Closed Landfill were evaluated using long-term monitoring data from a single downgradient monitoring bore (Well 5660). No upgradient bore is available at this site to provide a direct representation of background groundwater quality. Therefore, UPL(95%) values were adopted as the baseline for comparative assessment. Historical monitoring bores (Well 5659 and Well 4222) were previously installed at this site but are no longer monitored.

Time-series monitoring results for the 10 targeted variables are shown in Figure 11. Means, standard deviations, t-tests results are provided in Table 12. For each variable, the t-tests were used to determine whether there was a statistical difference between the mean of monitoring results and the specified threshold value. Statistically significant ($p<0.05$) negative t-values denote cases where the mean of the monitoring results is distinguishably less than the threshold. Linear regression equations and their associated statistical measures are provided in Table 11.

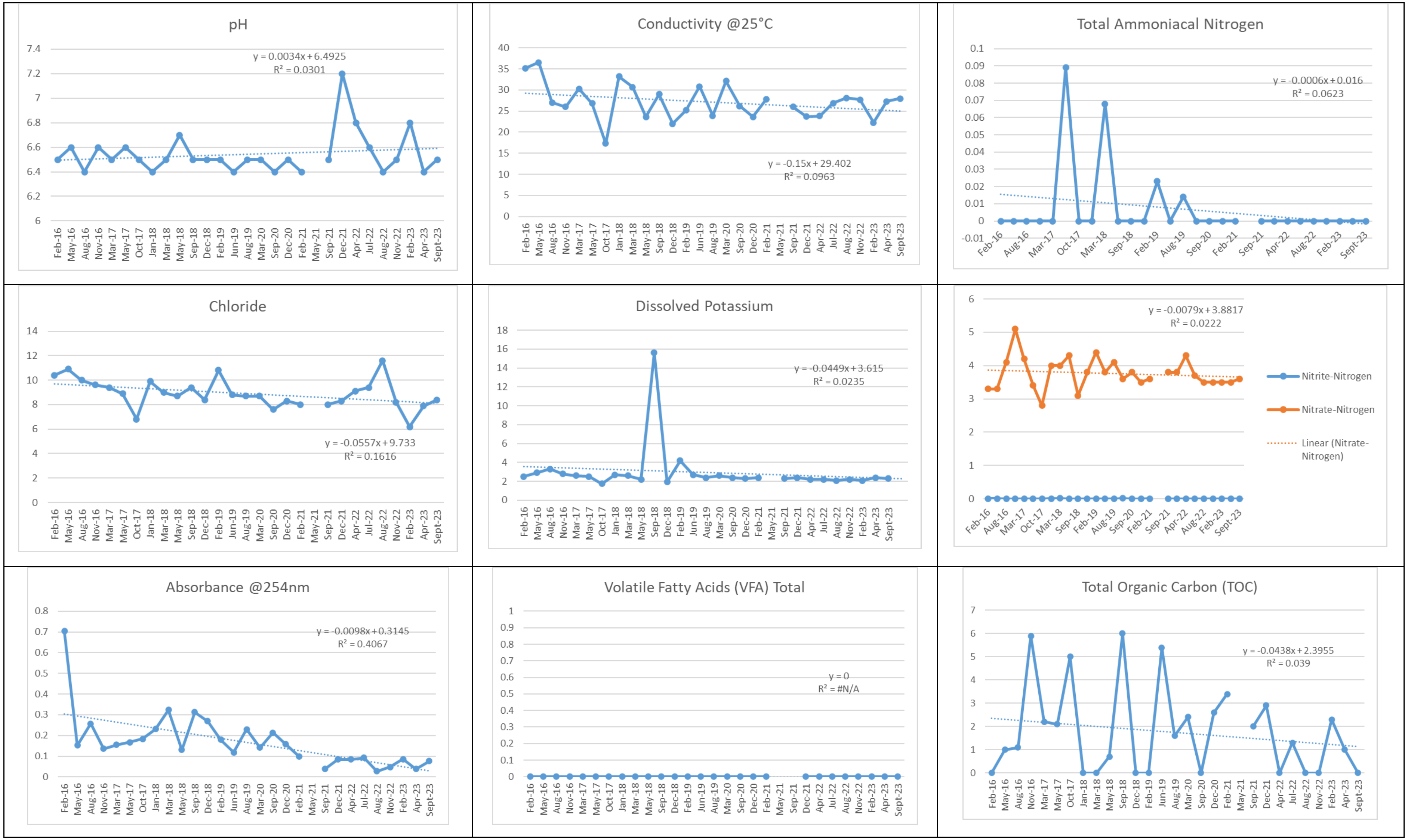
Table 11: trend equation and R^2 value for monitoring wells - Takapau Closed landfill. Well 5660 is downgradient.

	Well	5660
	Equation	R^2 value
pH	y = 0.0034x + 6.4925	R^2 = 0.0301
Electrical Conductivity (EC)	y = -0.15x + 29.402	R^2 = 0.0963
Dissolved Potassium	y = -0.0449x + 3.615	R^2 = 0.0235
Chloride	y = -0.0557x + 9.733	R^2 = 0.1616
Total Ammoniacal-N	y = -0.0006x + 0.016	R^2 = 0.0623
Nitrite-N	-	-
Nitrate-N	y = -0.0079x + 3.8817	R^2 = 0.0222
Nitrate-N + Nitrite-N		
Total Organic Carbon (TOC)	y = -0.0438x + 2.3955	R^2 = 0.039
Volatile Fatty Acids (VFA)	-	-
Absorbance at 254 nm	y = -0.0098x + 0.3145	R^2 = 0.4067

Table 12: Summary statistics and t-test results for Well No. 5660, the downgradient well and UPL(95%) values. The t-tests were between the mean of the monitoring results, and the specified threshold value

	Unit	n	Mean	Standard Deviation	Threshold	t-value	p-value	UPL(95%)
pH		28	6.54	0.17	7.5	-29.38	<0.0001	6.84
Electrical Conductivity (EC)	µS/cm	28	27.18	4.17	72.3	-57.28	<0.0001	34.40
Dissolved Potassium	mg/L	28	2.95	2.25	22.4	-40.83	<0.0001	7.32
Chloride	mg/L	28	8.91	1.19	250	-1068.4	<0.0001	10.98
Total Ammoniacal-N	mg/L	28	0.05	0.04	0.41	-53.32	<0.0001	0.11
Nitrite-N	mg/L	28	-	-	-	-	-	0.01
Nitrate-N	mg/L	28	3.76	0.46	11.3	-87.06	<0.0001	4.56
Total Organic Carbon (TOC)	mg/L	28	2.72	1.73	12.8	-30.86	<0.0001	5.72
Volatile Fatty Acids (VFA)	mg/L	28	-	-	-	-	-	-
Absorbance at 254 nm		28	0.17	0.13	0.7	-20.40	<0.0001	0.40

Figure 11: Monitoring data - Takapau - 5660 (downgradient):



For pH, values remain tightly clustered within a narrow range (6.4 - 6.8) with a single late-stage spike to 7.2 Figure 11, with only minor short-term fluctuations and no sustained directional change. The linear regression confirms the absence of a meaningful temporal trend, with a slope indistinguishable from zero ($y = 0.003x + 6.493$, $p>0.3$), indicating that pH has remained effectively stable through time. There was an isolated fluctuation observed in the December 2021 sample (pH = 7.2); however, pH returned to within the typical range by April 2022. This isolated spike is likely to reflect short-term environmental influences or analytical variability rather than a persistent change in groundwater chemistry. The mean pH at this well is 6.54 (Table 12), which is below the lower bound of the aesthetic drinking water range (7.00 - 8.50). Mean pH is substantially below both the specified threshold value of 7.5 and the UPL(95%) value of 6.84, supported by a strongly negative t-value ($t = -29.38$, $p<0.0001$). The pH values at the downgradient bore are mildly acidic but stable throughout the time.

EC remains effectively constant over the monitoring period (Figure 11). Measured values fluctuate within a relatively narrow range (21 µS/cm - 34 µS/cm), with minor short-term variability (May 2016 and October 2017) but no consistent directional change through time. The linear regression indicates a weak negative trend ($y = -0.15x + 29.40$) which fails to reach statistical significance ($R^2 = 0.096$, $p=0.11$), confirming that there is no meaningful long-term change in EC. The mean EC at this well is 27.2 µS/cm (Table 12), which is substantially below the site-specific threshold value of 72.3 µS/cm and the UPL(95%) value of 34.4 µS/cm, supported by a strongly negative t-value ($t = -57.28$, $p<0.0001$). The EC values at the downgradient bore remain consistently low and stable.

Dissolved potassium concentrations have remained low and generally consistent over the monitoring period, with a short-term fluctuation observed in September 2018 (Figure 11). As no cause for this fluctuation was identified, it is considered likely to be an analytical anomaly. The linear regression provides no evidence of a temporal trend ($y =-0.045x + 3.62$, $R^2 = 0.024$, $p = 0.43$), confirming that potassium concentrations have remained effectively stable over time. The mean dissolved potassium concentration at this well is 2.95 mg/L (Table 12) and remains well below both the site-specific threshold value of 22.4 mg/L and the UPL(95%) value of 7.32 mg/L, supported by a strongly negative t-value ($t = -40.83$, $p<0.0001$).

Chloride concentrations show a slight downtrend over time (Figure 11), with values fluctuating within a narrow range (8 mg/L - 10 mg/L), with occasional short-term increases and decreases. The linear regression indicates a weak negative trend ($y = -0.0557x + 9.73$, $R^2 = 0.162$, $p = 0.03$), suggesting that temporal changes in chloride are significant ($p<0.05$) but gradual. To the extent that this record may include some chloride from landfill leachate, a downward trend is in the preferred direction. The mean chloride concentration at this well is 8.91 mg/L (Table 12), which is well below the aesthetic drinking water guideline value of 250 mg/L and the UPL(95%) value of 10.98 mg/L. This is strongly supported by the t-value ($t = -1068.4$, $p<0.0001$). The chloride concentrations remain low, stable, with a weak decline trend and well within threshold limits.

TAN also shows no significant change over time, with a mean concentration of 0.05 mg/L at the downgradient well. This is well below the drinking-water guideline threshold of 0.41 mg/L and the UPL(95%) value of 0.11 mg/L ($t = -53.32$; $p<0.001$) (Table 12). The linear regression shows no evidence of a trend ($y = -0.0006x + 0.016$, $R^2 = 0.062$, $p = 0.2$) (Table 11). TAN concentrations remained negligible throughout the monitoring period, except for fluctuations observed in May 2017, March 2018, February 2019, and August 2019.

No time-series trend is observed for nitrite-N, and concentrations remain negligible throughout the monitoring period.

Nitrate-N shows the highest concentrations of the three forms of nitrogen, the mean nitrate-N at the downgradient well 5660 is 3.76 mg/L, well below the drinking-water guideline of 11.3 mg/L and UPL(95%) value of 4.56 mg/L ($t = -87.06$, $p<0.001$). There are no obvious trends in the time-series data (Figure 11), and the linear regression indicates essentially no trend over the monitoring period ($y = -0.008x + 3.882$; $R^2 = 0.022$, $p = 0.45$). Nitrate-N concentrations showed fluctuations in the early years but have remained stable since late 2022, with all values below threshold limits.

TOC and A254 concentrations remain consistently low. Linear regression indicates no significant trend for TOC ($y = -0.044x + 2.396$; $R^2 = 0.039$, $p = 0.28$), confirming that concentrations have remained effectively stable over the monitoring period. The mean TOC concentration is 2.72 mg/L (Table 12), which is well below both the site-specific threshold value of 12.8 mg/L and the UPL(95%) of 5.72 mg/L, supported by a strongly negative t-value ($t = -30.86$, $p<0.0001$). In contrast with TOC, A254 shows a weak but significant declining trend over time ($y = -0.0098x + 0.3145$; $R^2 = 0.4067$, $p<0.0003$), which (if genuine) may reflect a decrease in the proportion of dissolved aromatic compounds in groundwater, as seen for Waipawa closed landfill (Section 3.2.1). The mean value of 0.17 (Table 12) is below both the site-specific threshold of 0.7 and the UPL(95%) of 0.40, supported by a strongly negative t-value ($t = -20.40$, $p<0.0001$).

Groundwater quality at the downgradient bore at Takapau (Well 5660) remains consistently low across all measured parameters, with no indication of enrichment from the landfill. Concentrations of chloride ($p<0.03$) and A254 ($p<0.0003$) have both decreased over time, consistent with a decrease or tailing off of residual landfill influence. Variability is observed in some parameters, but spikes are not sustained. Taken together, these lines of evidence demonstrate that the Takapau Closed Landfill is exerting minimal to negligible influence on downgradient groundwater quality.

3.2.4 Tamumu Closed Landfill

Figure 12: GIS map - Tamumu Closed Landfill

Groundwater quality at the Tamumu Closed Landfill was assessed using results from the single downgradient monitoring bore (5504), with UPL(95%) values adopted as the baseline for comparative assessment of groundwater quality. Historical monitoring bores (Well 5355 and Well 4478) at this site are no longer monitored.

Time-series monitoring results for the 10 targeted variables are shown in Figure 13. Means, standard deviations, t-tests results are provided in Tables 13 and 14. For each variable, the t-tests were used to determine whether there was a statistical difference between the mean of monitoring results and the specified threshold value. Statistically significant ($p<0.05$) negative t-values denote cases where the mean of the monitoring results is distinguishably less than the threshold. Linear regression equations describing temporal trends, together with their associated statistical measures, are provided in Table 13.

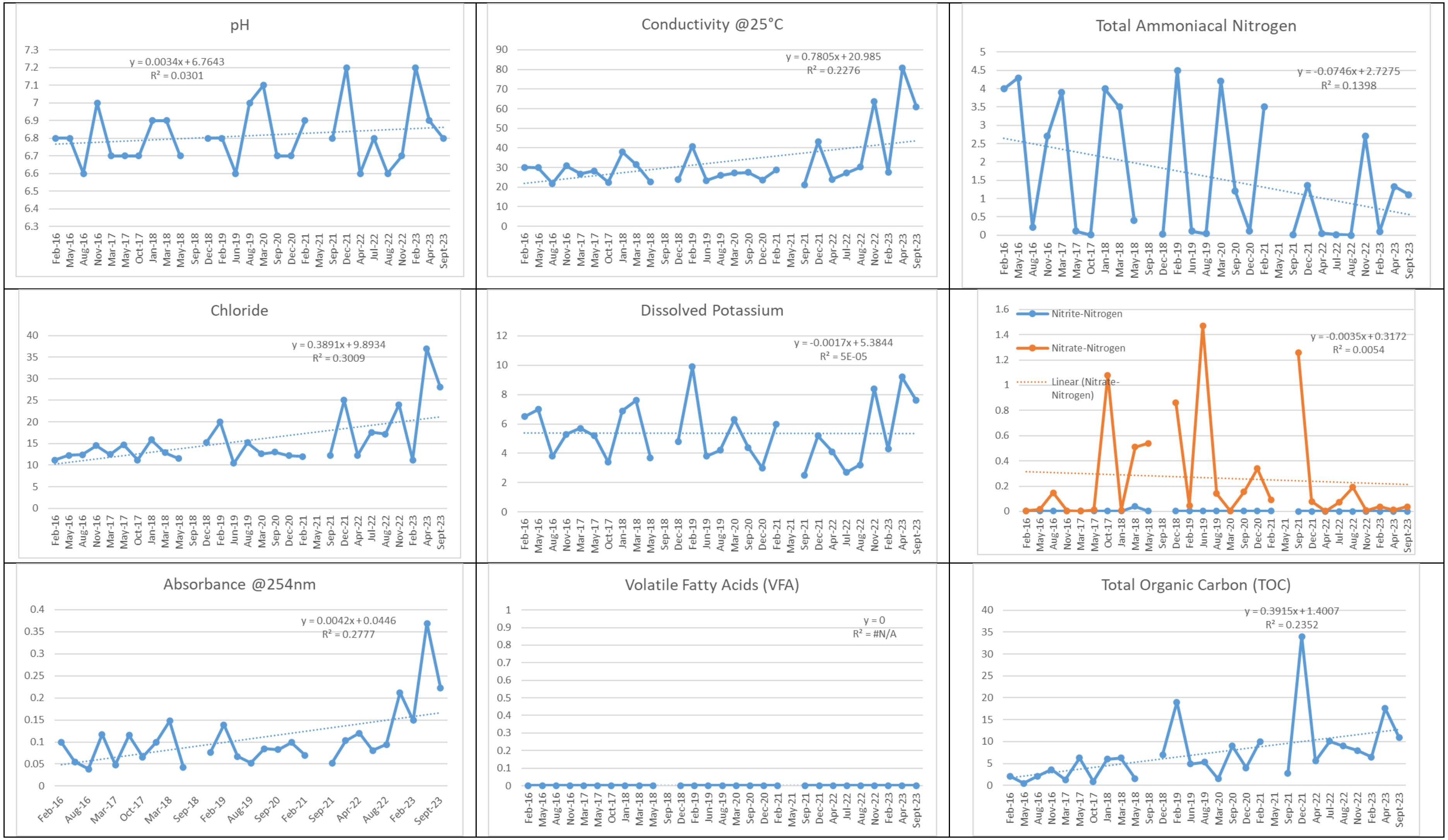
Table 13: trend equation and R^2 value for monitoring wells - Tamumu Closed landfill. Well 5504 is downgradient.

	Well	5504
	Equation	R^2 value
pH	y = 0.0034x + 6.7643	R^2 = 0.0301
Electrical Conductivity (EC)	y = 0.7805x + 20.985	R^2 = 0.2276
Dissolved Potassium	y = -0.0017x + 5.3844	R^2 = 5E-05
Chloride	y = 0.3891x + 9.8934	R^2 = 0.3009
Total Ammoniacal-N	y = -0.0746x + 2.7275	R^2 = 0.1398
Nitrite-N	-	-
Nitrate-N	y = -0.0035x + 0.3172	R^2 = 0.0054
Nitrate-N + Nitrite-N		
Total Organic Carbon (TOC)	y = 0.3915x + 1.4007	R^2 = 0.2352
Volatile Fatty Acids (VFA)	-	-
Absorbance at 254 nm	y = 0.0042x + 0.0446	R^2 = 0.2777

Table 14: Summary statistics and t-test results for Well No. 5504, the downgradient well and UPL(95%) values. The t-tests were between the mean of the monitoring results, and the specified threshold value

	unit	n	Mean	Standard Deviation	Threshold	t-value	p-value	UPL(95%)
pH		27	6.81	0.17	7.3	-13.30	<0.0001	7.11
Electrical Conductivity (EC)	μS/cm	27	32.66	14.31	73.2	-14.70	<0.0001	57.52
Dissolved Potassium	mg/L	27	5.36	2.01	12.7	-19.11	<0.0001	8.84
Chloride	mg/L	27	15.71	6.20	250	-196.23	<0.0001	26.49
Total Ammoniacal-N	mg/L	27	1.67	1.75	0.41	3.76	<0.0001	4.71
Nitrite-N	mg/L	27	0.00	0.01	0.09	-507.54	-	0.02
Nitrate-N	mg/L	27	0.26	0.42	11.3	-136.90	<0.0001	0.99
Total Organic Carbon (TOC)	mg/L	27	7.26	7.06	21.9	-10.74	<0.0001	19.52
Volatile Fatty Acids (VFA)	mg/L	27	-	-	-	-	-	-
Absorbance at 254 nm		27	0.11	0.07	0.3	-12.82	<0.0001	0.23

Figure 13: Monitoring data - Tamumu - 5504 (downgradient):



For pH, the downgradient monitoring bore at the Tamumu Closed Landfill (Well 5504) shows systematic variation over the monitoring period within a range of 6.6 - 7.2 (Figure 13). Linear regression shows no significant trend overall ($y = 0.003x + 6.76$; $R^2 = 0.030$), though higher than usual pH readings were recorded at two of the more recent sampling events (Dec 21 and Feb 23) (Figure 13). The mean pH at this well is 6.81 (Table 14), which is slightly below the lower bound of the aesthetic drinking water range (7.00 - 8.50). Mean pH is also below the site-specific threshold value of 7.3 and the UPL(95%) value of 7.11, supported by a strongly negative t-value ($t = -13.30$, $p<0.0001$).

For EC, the linear regression indicates an upward trend ($y = 0.781x + 20.99$; $R^2 = 0.228$, $p=0.012$) (Figure 13). Visual inspection shows that the EC concentration was historically stable, but has started showing more variability since November 2022. The reasons for this are unclear. The mean EC concentration at this well is 32.66 µS/cm (Table 14), which is low relative to both the UPL(95%) value of 57.52 µS/cm and the site-specific threshold of 73.2 µS/cm, supported by a strongly negative t-value ($t = -14.70$, $p<0.0001$). Importantly, even with this recent elevation, EC concentrations remain well below the nominated threshold limits.

Like EC, chloride concentrations show an overall increasing tendency through recent elevation since December 2021. The linear regression indicates an upward trend ($y = 0.389x + 9.89$; $R^2 = 0.301$, $p=0.003$) (Figure 13). As expected, the chloride record also correlates with the EC record. The mean chloride concentration is 15.71 mg/L (Table 14), which remains significantly below the aesthetic drinking water guideline of 250 mg/L and well below the UPL(95%) value of 26.49 mg/L, supported by an extremely negative t-value ($t = -196.23$, $p<0.0001$).

Dissolved potassium shows no statistical pattern through the monitoring time, with the data fluctuating within a range of 2 mg/L - 10 mg/L. The linear regression indicates essentially no trend ($y = -0.002x + 5.38$; $R^2 = 0.00005$) (Figure 13). The mean potassium concentration at this well is 5.36 mg/L (Table 14), which remains well below both the site-specific threshold value of 12.7 mg/L and the UPL(95%) value of 8.84 mg/L, supported by a strongly negative t-value ($t = -19.11$, $p<0.0001$).

TAN shows a downwards trend with historical fluctuations (Figure 13). The linear regression indicates a weak declining trend ($y = -0.075x + 2.73$; $R^2 = 0.140$, $p=0.055$). The mean TAN concentration is 1.67 mg/L (Table 14), which exceeds the aesthetic threshold value of 0.41 mg/L, but is lower than the UPL(95%) value of 4.71 mg/L, as confirmed by a positive t-value ($t = 3.76$, $p<0.0001$) (Table 14). It is a concern that TAN concentrations exceed the threshold limit, even though they remain below the UPL (95%) value. This suggests that the anaerobic decomposition of organic material (especially nitrogen-rich proteins) is influencing groundwater quality and may be the result of landfill leachate or geochemical changes in the surrounding environment.

Nitrite-N concentrations remain extremely low throughout the monitoring record, with values typically at or below the analytical detection limit (Figure 13).

Nitrate-N shows the highest concentrations among the nitrogen species at Tamumu site. The linear regression indicates a weak declining trend with historical fluctuations stable since November 2022 ($y = -0.004x + 0.32$; $R^2 = 0.005$) (Figure 13). Mean nitrate-N at this Well is 0.26 mg/L (Table 14), which is comfortably below both the nominated threshold of 11.3 mg/L and UPL(95%) value of 0.99 mg/L, supported by a strongly negative t-value ($t = -136.90$, $p<0.0001$).

The linear regression shows an upward trend of TOC concentrations throughout the time ($y = 0.392x + 1.401$; $R^2 = 0.235$, $p=0.01$) (Figure 13). The mean TOC concentration is 7.26 mg/L (Table 14), which remains well below both the site-specific threshold of 21.9 mg/L and the UPL(95%) value of 19.52 mg/L, supported by a strongly negative t-value ($t = -10.74$, $p<0.0001$). The linear regression for A254 shows a weak upward trend ($y = 0.0042x + 0.045$; $R^2 = 0.28$, $p=0.005$) (Figure 13). The mean A254 value of 0.11 is comfortably below both the site-specific threshold of 0.3 and the UPL(95%) value of 0.23, supported by a strongly negative t-value ($t = -12.82$, $p<0.0001$). Overall, a notable upward trend is observed for both TOC and A254. However, as the values remain well below the nominated threshold limits, continued monitoring is recommended without immediate concerns.

Across the full suite of monitored parameters at the Tamumu site, with the exception of TAN concentrations exceeding the applicable threshold limit, all other parameters remain well below their respective thresholds and UPL (95%) values. This evidence indicates that the Tamumu Closed Landfill is exerting a minimal to negligible influence on downgradient groundwater quality and, under current conditions, presents a low environmental risk. However, continuing monitoring is recommended because of the upward trends in organic compounds (TOC and absorbance), and recent apparent spikes in EC and chloride (which contributes to EC).

3.2.5 Tikokino Closed Landfill

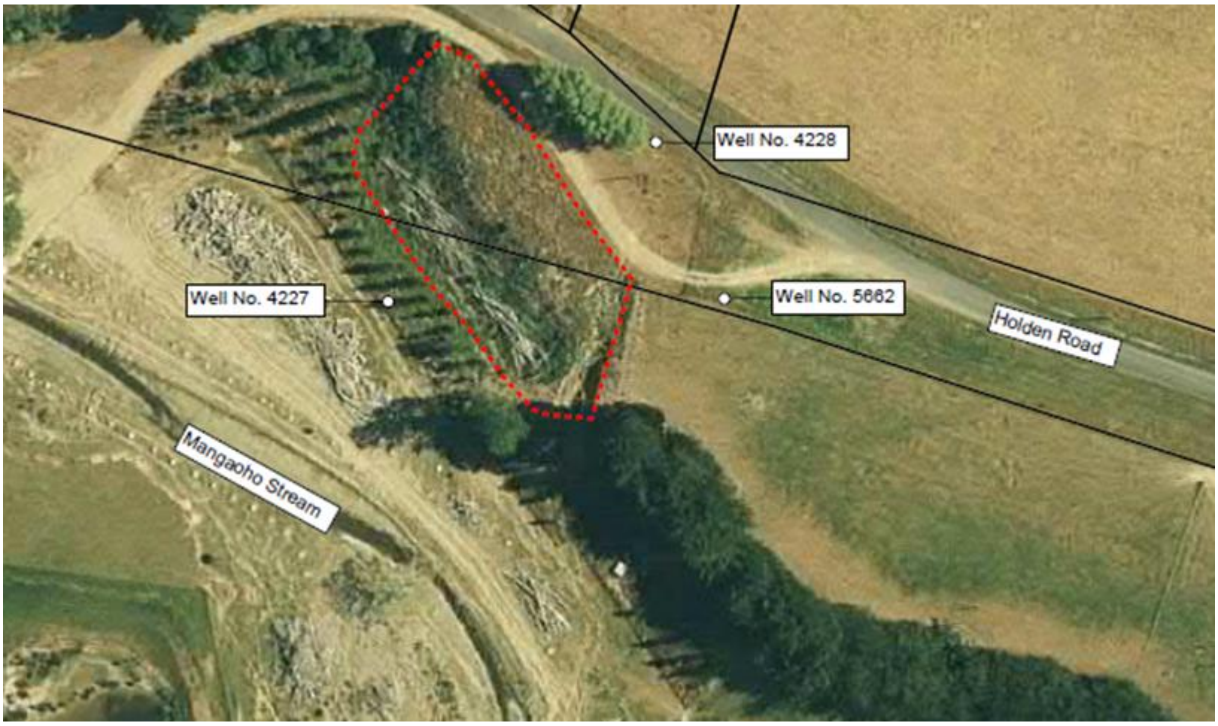


Figure 14: GIS map - Tikokino Closed Landfill - enclosed by the dotted red line indicates the extent of the former disposal area

Groundwater at the Tikokino Closed Landfill is monitored at three wells, upgradient bore (Well 4228) and downgradient bores (Well 5662 and Well 4227). This interpretation is based on the inferred regional groundwater flow direction across the site and the relative spatial positions of the monitoring bores with respect to the landfill boundary. The use of both upgradient and downgradient wells allows for direct comparison of groundwater quality and supports discrimination between landfill-related effects and broader background or catchment-scale influences.

Time-series monitoring results for the 9 targeted groundwater quality variables at the Tikokino Closed Landfill are shown in Figures 15–18. Summary statistics, including means, standard deviations, and t-test results, are provided in Tables 16–18 for the upgradient bore (Well 4228) and the two downgradient bores (Wells 5662 and 4227). For each variable, t-tests were undertaken to determine whether the mean of the monitoring results differs significantly from the specified threshold value. Statistically significant ($p < 0.05$) negative t-values denote cases where the monitoring mean is distinguishably lower than the threshold, indicating compliance. The linear regression equations describing temporal trends are provided in Table 15.

Table 15: trend equation and R^2 value for monitoring wells - Tikokino Closed landfill. Well 4228 is upgradient, and well 5662 and 4227 are downgradient.

	Well 4228		Well 5662		Well 4227	
	Equation	R^2 value	Equation	R^2 value	Equation	R^2 value
pH	y = -0.0141x + 7.0429	R^2 = 0.1558	y = 0.0113x + 6.5714	R^2 = 0.1729	y = -0.0086x + 6.8543	R^2 = 0.0603
Electrical Conductivity (EC)	y = -0.0044x + 24.287	R^2 = 6E-05	y = 0.1362x + 33.907	R^2 = 0.0011	y = -0.0436x + 13.489	R^2 = 0.0023
Dissolved Potassium	y = -0.0157x + 2.1666	R^2 = 0.0122	y = -0.0064x + 2.4316	R^2 = 0.0102	y = -0.0196x + 1.7481	R^2 = 0.0208
Chloride	y = 0.0011x + 11.276	R^2 = 2E-05	y = 0.431x + 4.1545	R^2 = 0.0595	y = -0.0157x + 6.0003	R^2 = 0.0054
Total Ammoniacal-N	y = -0.0002x + 0.0124	R^2 = 0.0099	y = -0.0002x + 0.0346	R^2 = 0.0099	y = -0.0003x + 0.0096	R^2 = 0.0224
Nitrite-N	y = 0.0001x +0.0028	R^2 = 0.0077	y = 0.0008x +0.0221	R^2 = 0.0175	y = -3E-06x +0.0002	R^2 = 0.0014
Nitrate-N	y = -0.0258x + 0.7984	R^2 = 0.5239	y = -0.0087x + 3.4321	R^2 = 0.0122	y = -0.0248x + 1.902	R^2 = 0.0016
Total Organic Carbon (TOC)	y = 0.1x + 0.5145	R^2 = 0.1856	y = 0.0033x + 2.9512	R^2 = 6E-05	y = -0.0172x + 2.2871	R^2 = 0.0134
Volatile Fatty Acids (VFA)	-	-	-	-	y = -0.02x + 1.004	R^2 = 0.0534
Absorbance at 254 nm	y = -0.0022x + 0.0685	R^2 = 0.1996	y = -0.0045x + 0.2898	R^2 = 0.0782	y = 0.0011x + 0.0416	R^2 = 0.0252

Table 16: Summary statistics and t-test results for Well No. 4228, the upgradient well. The t-tests were between the mean of the monitoring results, and the specified threshold value.

	Unit	n	Mean	Standard Deviation	Threshold	t-value	p-value
pH		28	6.85	0.29	7.7	-14.92	<0.0001
Electrical Conductivity (EC)	µS/cm	28	24.22	4.86	40.5	-17.76	<0.0001
Dissolved Potassium	mg/L	28	1.93	1.22	4.4	-10.83	<0.0001
Chloride	mg/L	28	11.29	2.05	250	-614.72	<0.0001
Total Ammoniacal-N	mg/L	28	0.04	0.02	0.41	-82.76	<0.0001
Nitrite-N	mg/L	28	0.02	0.02	0.90	-258.16	<0.0001
Nitrate-N	mg/L	28	0.45	0.29	11.3	-195.13	<0.0001
Total Organic Carbon (TOC)	mg/L	28	2.94	1.75	15.7	-38.52	<0.0001
Volatile Fatty Acids (VFA)	mg/L	28	-	-	-	-	-
Absorbance at 254 nm		28	0.04	0.04	1.7	-206.39	<0.0001

Table 17: Summary statistics and t-test results for Well No. 5662, one of the two downgradient wells. The t-tests were between the mean of the monitoring results, and the specified threshold value.

	Unit	n	Mean	Standard Deviation	Threshold	t-value	p-value
pH		19	6.74	0.23	7.3	-9.60	<0.0001
Electrical Conductivity (EC)	µS/cm	19	35.93	36.14	106.8	-8.54	<0.0001
Dissolved Potassium	mg/L	19	2.34	0.55	3.4	-8.68	<0.0001
Chloride	mg/L	19	10.54	15.22	250	-68.56	<0.0001
Total Ammoniacal-N	mg/L	19	0.03	0.01	0.41	-127.04	<0.0001
Nitrite-N	mg/L	19	-	-	-	-	-
Nitrate-N	mg/L	19	3.30	0.67	11.3	-51.54	<0.0001
Total Organic Carbon (TOC)	mg/L	19	2.19	1.31	19.5	-57.93	<0.0001
Volatile Fatty Acids (VFA)	mg/L	19	-	-	-	-	-
Absorbance at 254 nm		19	0.06	0.07	0.8	-47.88	<0.0001

Table 18: Summary statistics and t-test results for Well No. 4227, the other downgradient well. The t-tests were between the mean of the monitoring results, and the specified threshold value.

	Unit	n	Mean	Standard Deviation	Threshold	t-value	p-value
pH		28	6.75	0.32	7.4	-10.18	<0.0001
Electrical Conductivity (EC)	µS/cm	28	12.95	8.39	25.6	-7.96	<0.0001
Dissolved Potassium	mg/L	28	1.50	1.25	3.3	-7.58	<0.0001
Chloride	mg/L	28	5.81	1.95	250	-661.18	<0.0001
Total Ammoniacal-N	mg/L	28	0.03	0.03	0.41	-58.30	<0.0001
Nitrite-N	mg/L	28	-	-	-	-	-
Nitrate-N	mg/L	28	1.59	5.67	11.3	-9.05	<0.0001
Total Organic Carbon (TOC)	mg/L	28	2.19	1.31	7.6	-21.74	<0.0001
Volatile Fatty Acids (VFA)	mg/L	28	-	-	-	-	-
Absorbance at 254 nm		28	0.06	0.07	0.2	-10.25	<0.0001

Figure 15: Monitoring data - Tikokino – Well No. 4228 (upgradient)

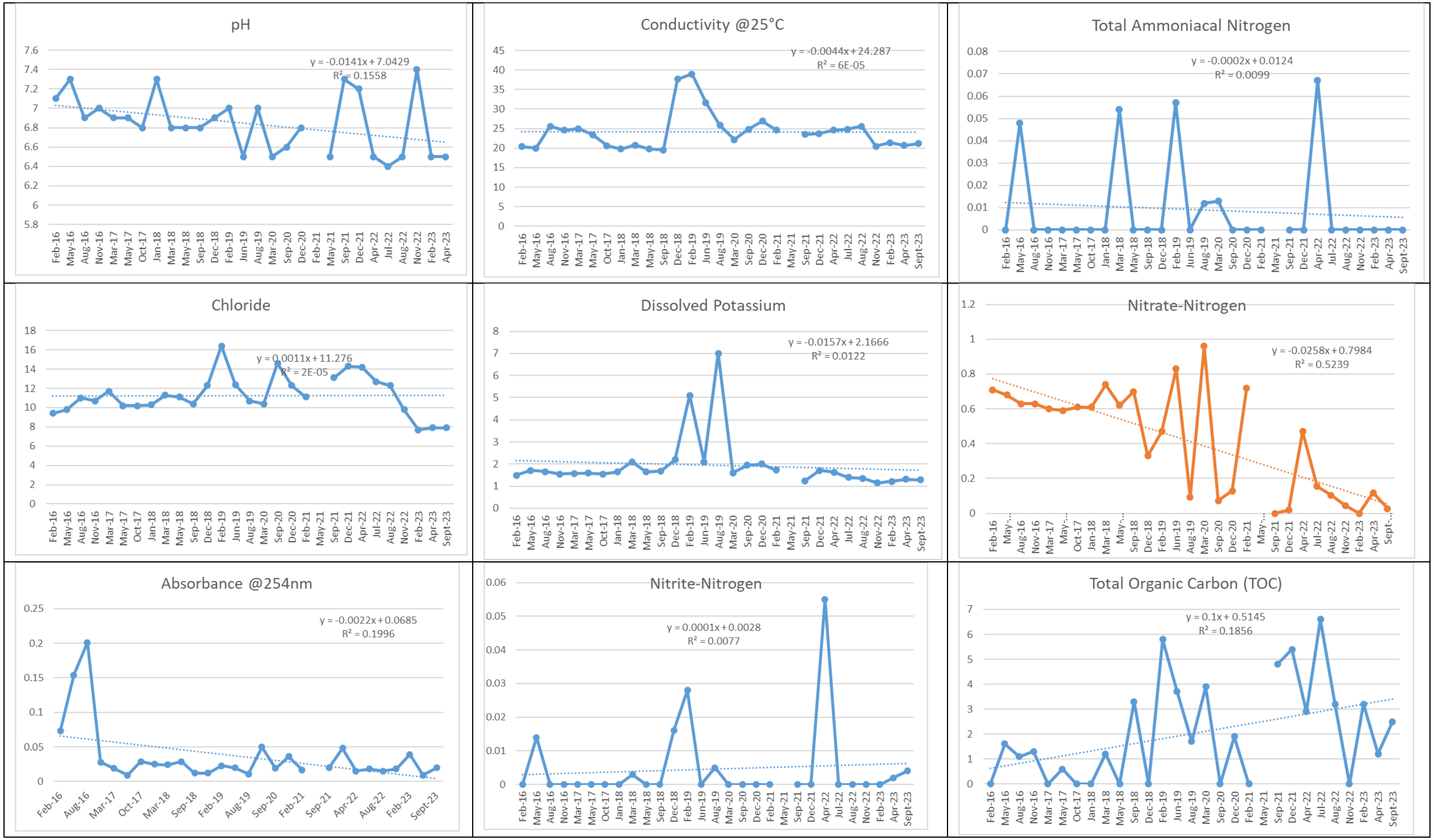


Figure 16: Monitoring data - Tikokino – Well No. 5662 (downgradient)

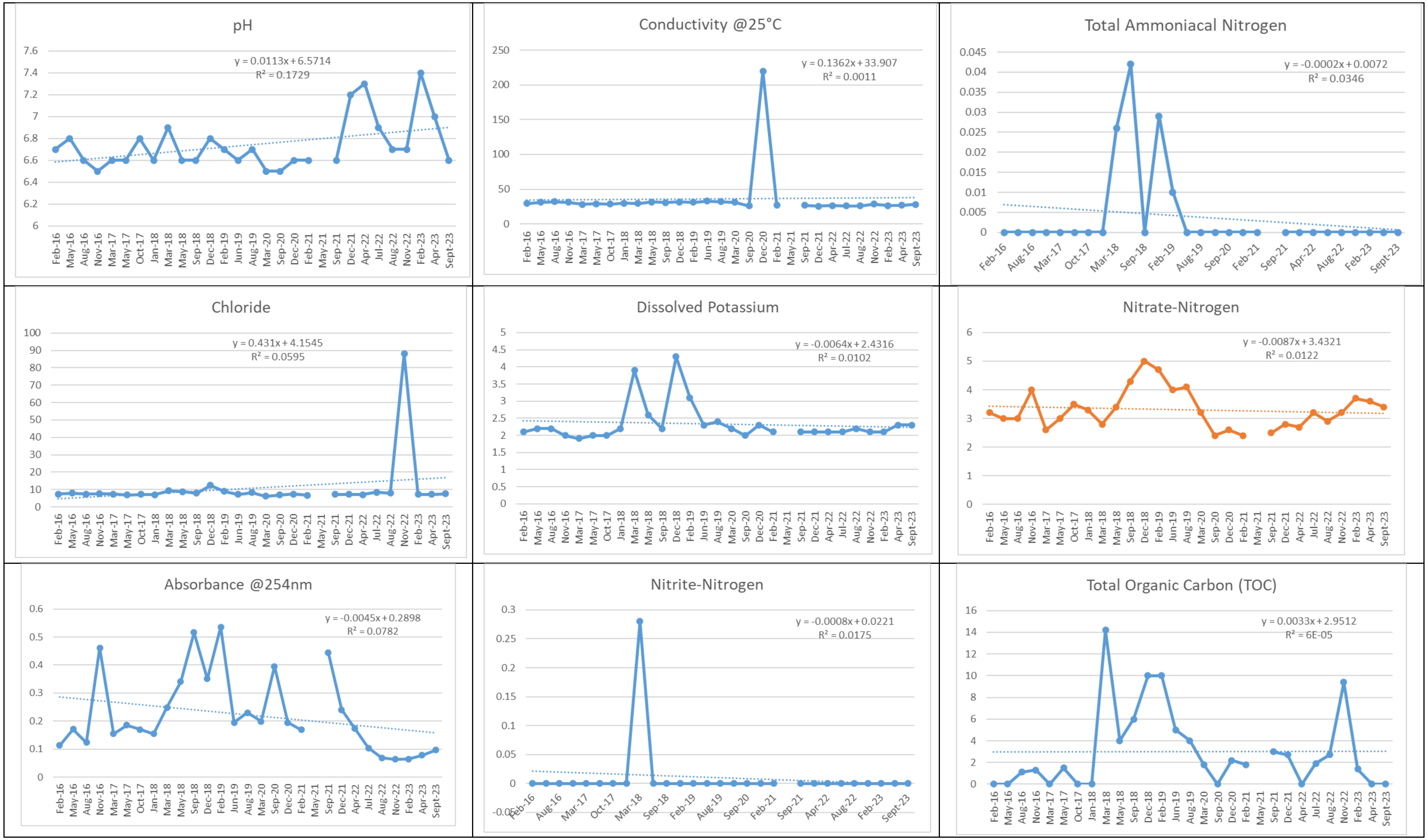


Figure 17: Monitoring data - Tikokino – Well No. 4227 (downgradient)

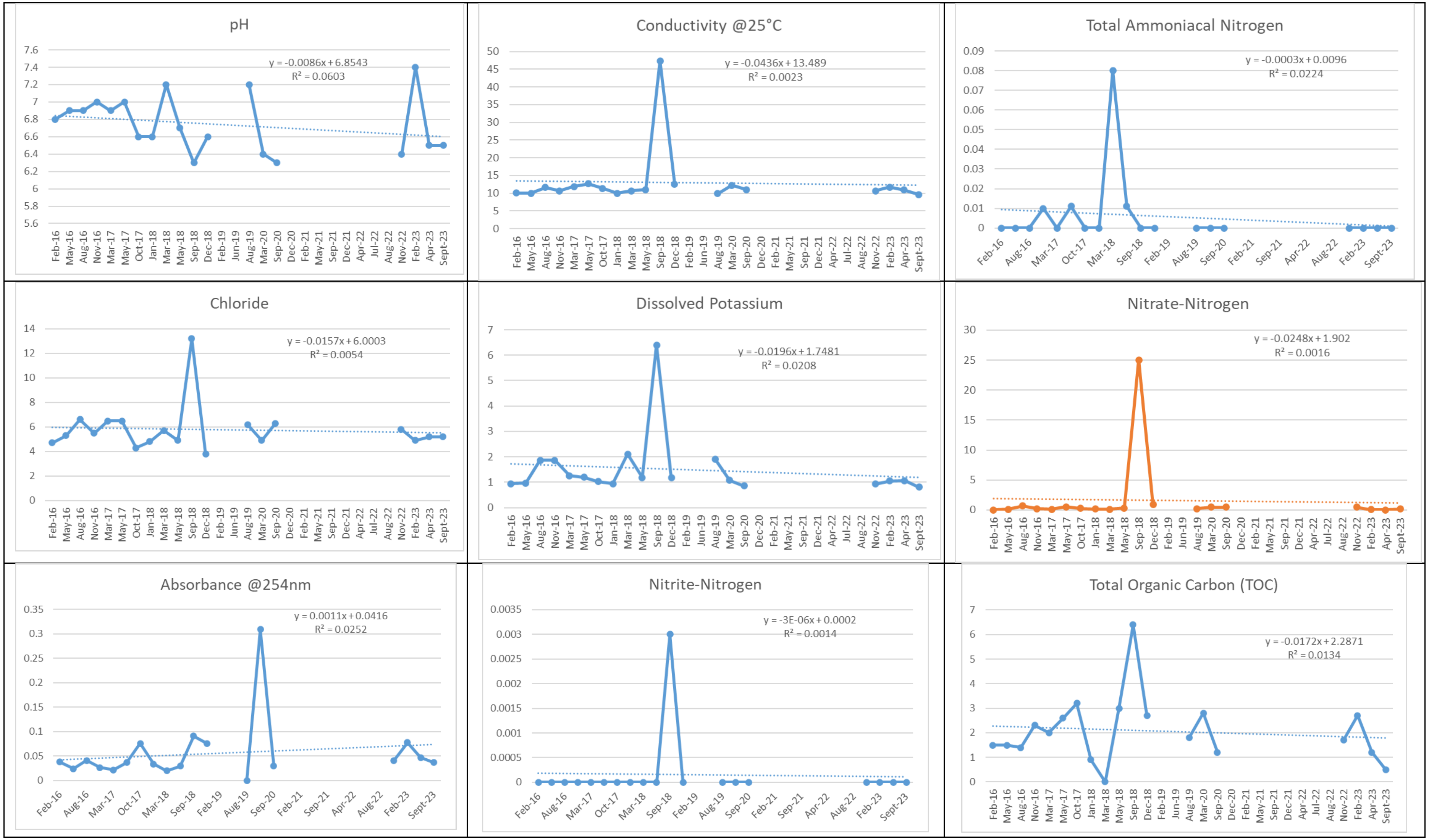
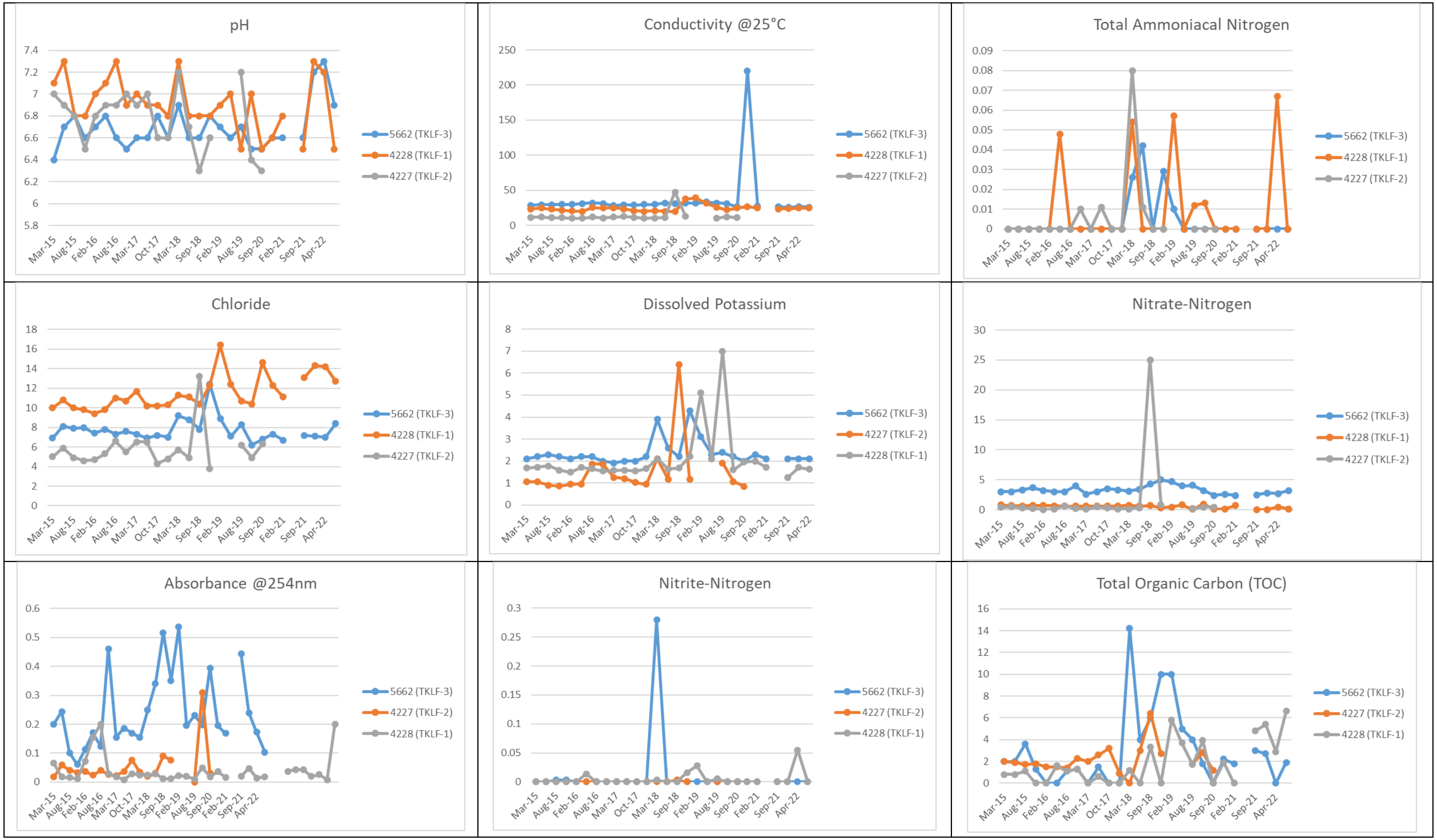


Figure 18: Comparison data – 4228, 5662 and 4227.



For pH, the upgradient well (Well 4228) shows a declining trend over time ($y = -0.014x + 7.043$; $R^2 = 0.156$, $p=0.04$) (Figure 15), indicating a gradual decrease in pH across the monitoring period. In contrast, the downgradient bores display differing but similarly weak temporal behaviour. Well 5662 shows an apparent weak increasing trend ($y = 0.011x + 6.57$; $R^2 = 0.173$, $p=0.03$) (Figure 16), while Well 4227 shows no trend ($y = -0.009x + 6.85$; $R^2 = 0.060$, $p>0.1$) (Figure 17). However, in all cases, the actual slopes are near zero, indicating that pH has remained stable over time, with no strong or sustained directional change at any monitoring location. Mean pH at the upgradient well is 6.85 (Table 16), which is slightly below the aesthetic drinking water range of 7.00 - 8.50. Mean pH values at the downgradient wells are similar, with values of 6.74 at Well 5662 (Table 17) and 6.75 at Well 4227 (Table 18), also below this range. All three wells remain well below their respective threshold values, supported by strongly negative t-test results (Well 4228: $t = -14.92$, $p<0.0001$; Well 5662: $t = -9.60$, $p<0.0001$; Well 4227: $t = -10.18$, $p<0.0001$). Overall, pH conditions are broadly similar across the upgradient and downgradient wells, with no evidence of pH elevation downgradient of the landfill.

EC, at the upgradient bore, EC remains effectively constant over the monitoring period, with linear regression indicating no meaningful temporal trend ($y = -0.004x + 24.29$; $R^2 = 0.00006$) (Figure 15). The mean EC at this well is 24.22 µS/cm and well below its threshold of 40.5 µS/cm (Table 16). At the downgradient locations, EC concentrations are similarly low overall, although modest differences are evident between the two bores. Well 5662 shows long term stability with one isolated peak in Dec 2020 ($R^2 = 0.0011$, $p>0.8$) (Figure 16). This, and a similar one-off peak for chloride in Nov 2022, may be analytical outliers, because EC and chloride would normally be expected to correlate. By contrast, Well 4227 ($R^2 = 0.0023$, $p>0.8$) also shows one isolated peak for EC in Sept 2018 (Figure 17). However, in this case, peaks also occur in Well 4227 on the same date for chloride, potassium, nitrate, nitrite, and TOC. This one-off elevation of 6 variables on the same sampling date suggests a genuine discharge event was being detected in Well 4227 in Sept 2018. The cause is unknown and could possibly relate to some type of sub-surface failure in the landfill that was only detected in that well, but such an event has not occurred a second time in the monitoring record. Setting aside the two possible analytical outliers in Well 5662, and the one-off Sept 18 event in Well 4227, the records for both wells indicate that EC has remained effectively stable through time. Mean EC at Well 5662 is 35.93 µS/cm, which is higher than both the upgradient well and Well 4227 (mean 12.95 µS/cm), but remains well below the site-specific threshold of 106.8 µS/cm (Table 17). Similarly, mean EC at Well 4227 is well below its threshold of 25.6 µS/cm (Table 18). These results are strongly supported by one-sample t-tests comparing monitoring means with threshold values, with strongly negative t-values at all three wells (Well 4228: $t = -17.76$, $p<0.0001$; Well 5662: $t = -8.54$, $p<0.0001$; Well 4227: $t = -7.96$, $p<0.0001$). Overall, EC concentrations are low relative to the nominated threshold limits, and t-tests strongly indicate that concentrations remain well below these thresholds ($p < 0.001$).

Notably, Well 4227, which is located closer to the Mangaoho Stream, shows the lowest EC concentrations, indicating that this well is not situated along the groundwater flow direction.

Chloride concentrations are low across all monitoring locations, with mean values of 11.29 mg/L at the upgradient reference bore (Well 4228), 10.54 mg/L at downgradient Well 5662, and 5.81 mg/L at downgradient Well 4227. These results show no consistent pattern of downgradient enrichment relative to the upgradient well (Tables 16–18). The upgradient Well 4228 records the highest chloride concentrations, while the more inland downgradient Well 5662 shows slightly lower values. This pattern is likely influenced by surrounding environmental factors, such as agricultural activities, rather than landfill-related effects.

The time-series plots (Figures 15–17) indicate that chloride values at all three wells are relatively long-term stable. The linear regression supports this interpretation, with no change (slopes indistinguishable from zero) at both the upgradient well ($y = 0.001x + 11.28$; $R^2 = 0.00002$) and downgradient Well 4227 ($y = -0.016x + 6.00$; $R^2 = 0.005$). A non-significant ($p=0.3$) upward slope indicated at downgradient Well 5662 ($y = 0.43x + 4.155$; $R^2 = 0.060$); however, this slope is an artefact of the single suspected outlier in November 2022 (Figure 16). When this outlier is excluded, the least-squares regression becomes $y = −0.0183x + 7.9352$ ($R^2 = 0.0166$), with a slope that is effectively indistinguishable from zero. Importantly, mean chloride concentrations at all monitoring locations remain comfortably below the aesthetic drinking-water guideline of 250 mg/L, supported by significant negative t-values (Well 4228: $t = -614.72$, $p<0.0001$; Well 5662: $t = -68.56$, $p<0.0001$; Well 4227: $t = -661.18$, $p<0.0001$) (Tables 16–18).

Dissolved potassium occurs at low concentrations across all monitoring wells and shows no statistically meaningful temporal trend. Isolated concentration peaks were observed at Well 4228 in February and August 2019, at Well 5662 in March and December 2018, and at Well 4227 in September 2018. No specific causes were identified for these isolated results, though, as noted above, the peak in September 2018 appears to have been associated with a genuine one-off event. Some of the others may reflect analytical error. Aside from these isolated peaks, dissolved potassium concentrations across all three monitoring wells remain consistently stable throughout the monitoring period. The linear regression analysis indicates no trends (slopes not significantly different from zero) at the upgradient bore (Well 4228: $y = -0.016x + 2.167$; $R^2 = 0.012$) and at the downgradient bores (Well 5662: $y = -0.006x + 2.43$; $R^2 = 0.010$; Well 4227: $y = -0.0196x + 1.748$; $R^2 = 0.0208$), confirming that potassium concentrations have remained effectively stable through time (Figures 15–17). Mean potassium concentrations are 1.93 mg/L at the upgradient bore, 2.34 mg/L at downgradient Well 5662, and 1.50 mg/L at downgradient Well 4227 (Tables 16–18). All mean values remain well below their respective threshold concentrations (4.4 mg/L for Well 4228, 3.4 mg/L for Well 5662, and 3.3 mg/L for Well

4227), supported by significant negative t-values (Well 4228: t = -10.83, p<0.0001; Well 5662: t = -8.68, p<0.0001; Well 4227: t = -7.58, p<0.0001).

TAN occurs at low concentrations across all wells and shows no statistically signficant temporal trends. Isolated concentration peaks were observed at Well 4228 in May 2016, May 2018, February and August 2019, February 2020 and April 2022, at Well 5662 in March 2018 and February 2019, and at Well 4227 in March 2018. No specific causes were identified for these isolated results, and some may reflect genuine seasonal or land use changes in subsurface conditions, whereas others may reflect analytical error. It is worth noting that many of the TAN and nitrite results are near or at the analytical detection limit, and this can make the low-level detections, which do occur seem more significant than they really are – on the scale of a graph. Aside from these isolated peaks, TAN concentrations across all three monitoring wells remain consistently stable throughout the monitoring period. The linear regression analysis show no trends with slopes indistinguishable from zero at the upgradient bore (y = -0.0002x + 0.012; R^2 = 0.010) and both downgradient bores (Well 5662: y = -0.0002x + 0.035; R^2 = 0.010; Well 4227: y = -0.0003x + 0.010; R^2 = 0.022), indicating stable concentrations through time (Figures 15 – 17). Mean TAN concentrations range from 0.03 to 0.04 mg/L across the monitoring network, remaining substantially below the nominated threshold value of 0.41 mg/L, supported by strongly by significant negative t-value at all wells (Well 4228: t = -82.76, p<0.0001; Well 5662: t = -127.04, p<0.0001; Well 4227: t = -58.30, p<0.0001).

Nitrite-N concentrations remain extremely low throughout the monitoring record. Isolated concentration peaks were observed at Well 4228 in March 2016, December 2018, February 2019 and April 2022, at Well 5662 in March 2018, and at Well 4227 in September 2018. When these outliers are excluded, the calculated trend slope is effectively indistinguishable from zero. Aside from these isolated peaks, Nitrite-N concentrations across all three monitoring wells remain consistently stable throughout the monitoring period. At the upgradient bore, mean nitrite-N is 0.02 mg/L, while at the downgradient wells, nitrite-N is either below detection limits or detected at very low levels (Tables 16–18). No discernible temporal trends are evident (Figures 15–17), and all values remain well below the nominated threshold.

Trend analysis indicates no consistent temporal increase in Nitrate-N at the downgradient wells, with no evidence for a trend over time at either Well 5662 (y = -0.009x + 3.432; R^2 = 0.012) and Well 4227 (y = -0.025x + 1.90; R^2 = 0.002) (Figures 16 and 17). In contrast, the upgradient bore exhibits a reasonably convincing downward trend (y = -0.026x + 0.804; R^2 = 0.524, p<0.0001) (Figure 15), which may relate to changes in upgradient land-use practices. For example, within pastoral farming systems, reductions in groundwater Nitrate-N may result from lower stocking densities (e.g. fewer cattle or a transition from dairy to sheep farming) and/or reduced application rates of nitrogen fertilisers. Importantly, mean nitrate-N concentrations at all wells (0.45 mg/L at Well 4228;

3.30 mg/L at Well 5662; 1.59 mg/L at Well 4227) remain well below the drinking-water health guideline of 11.3 mg/L, support by significant negative t-values (Well 4228: t = -195.13, $p<0.0001$; Well 5662: t = -51.54, $p<0.0001$; Well 4227: t = -9.05, $p<0.0001$).

TOC and A254 remain consistently low at all monitoring wells, with no indication of organic leachate influence or meaningful temporal trends (Figures 15–17). Linear regression indicates an upward trend at upgradient Well 4228, and no meaningful trend for both downgradient wells. Mean TOC concentrations are 2.94 mg/L at Well 4228, 2.95 mg/L at Well 5662, and 2.19 mg/L at Well 4227 (Tables 16–18), all well below their respective threshold values (15.7 mg/L, 19.5 mg/L, and 7.6 mg/L), strongly supported by t-values (Well 4228: t = -38.52, $p<0.0001$; Well 5662: t = -57.93, $p<0.0001$; Well 4227: t = -21.74, $p<0.0001$). The increase in TOC concentration at the upgradient Well 4228 is more likely influenced by surrounding environmental factors than by landfill leachate.

A254 also remains low across all three monitoring wells, with mean values of 0.04 at Well 4228, 0.06 at Well 5662, and 0.06 at Well 4227. All values are well below their respective aesthetic thresholds (1.7, 0.8, and 0.2), supported by strongly negative t-values (Well 4228: t = -206.39, $p<0.0001$; Well 5662: t = -47.88, $p<0.0001$; Well 4227: t = -10.25, $p<0.0001$). Linear regression shows only weak trends (R^2 = 0.1996 at Well 4228, 0.0782 at Well 5662, and 0.0252 at Well 4227). A254 readings are slightly higher in the downgradient wells on most sampling events, which may reflect the presence of additional aromatic or unsaturated compounds in the dissolved organic matter. However, these differences are small and do not indicate a sustained influence from landfill-derived leachate

Across the full suite of monitored parameters at the Tikokino Closed Landfill, temporal patterns remain weak, with only isolated fluctuations observed at all three monitoring wells. In September 2018, one of the two downgradient wells (Well 4227) picked up a transient peak in six variables (EC, chloride, potassium, nitrate, nitrite, and TOC) that could have possibly been caused by a subsurface event in the landfill, though this is speculative. Whatever the cause of that event, it has not happened a second time in the record to date, and the absolute values recorded were not extreme. For example, the September 2018 chloride peak in Well 4227 remained lower than at the upgradient Well 4228 (Figure 18).

Overall, the dataset for this landfill shows some isolated higher or lower measurements, rather than sustained directional change across the monitoring period. Allowing for probable analytical outliers, groundwater quality at the downgradient bores (Wells 5662 and 4227) is generally consistently below thresholds across all indicators. All values are well below their respective threshold limits.

3.2.6 Ongaonga Closed Landfill

Figure 19: GIS map - Ongaonga Closed Landfill

Groundwater quality at the Ongaonga Closed Landfill was assessed using results from the single downgradient monitoring bore (Well 5356), with UPL(95%) values adopted as the baseline for comparative assessment of groundwater quality. Historical monitoring bores (Well 4477 and Well 5502) at this site are no longer monitored.

Time-series monitoring results for the 9 targeted variables at the Ongaonga Closed Landfill are presented in Figure 20. Summary statistics, including means, standard deviations, and the results of one-sample t-tests are provided in Table 20. For each variable, the t-tests were used to determine whether there was a statistical difference between the mean of monitoring results and the specified threshold value. Statistically significant ($p<0.05$) negative t-values denote cases where the mean of the monitoring results is distinguishably less than the threshold. The linear regression equations and their associated statistical measures are provided in Table 19.

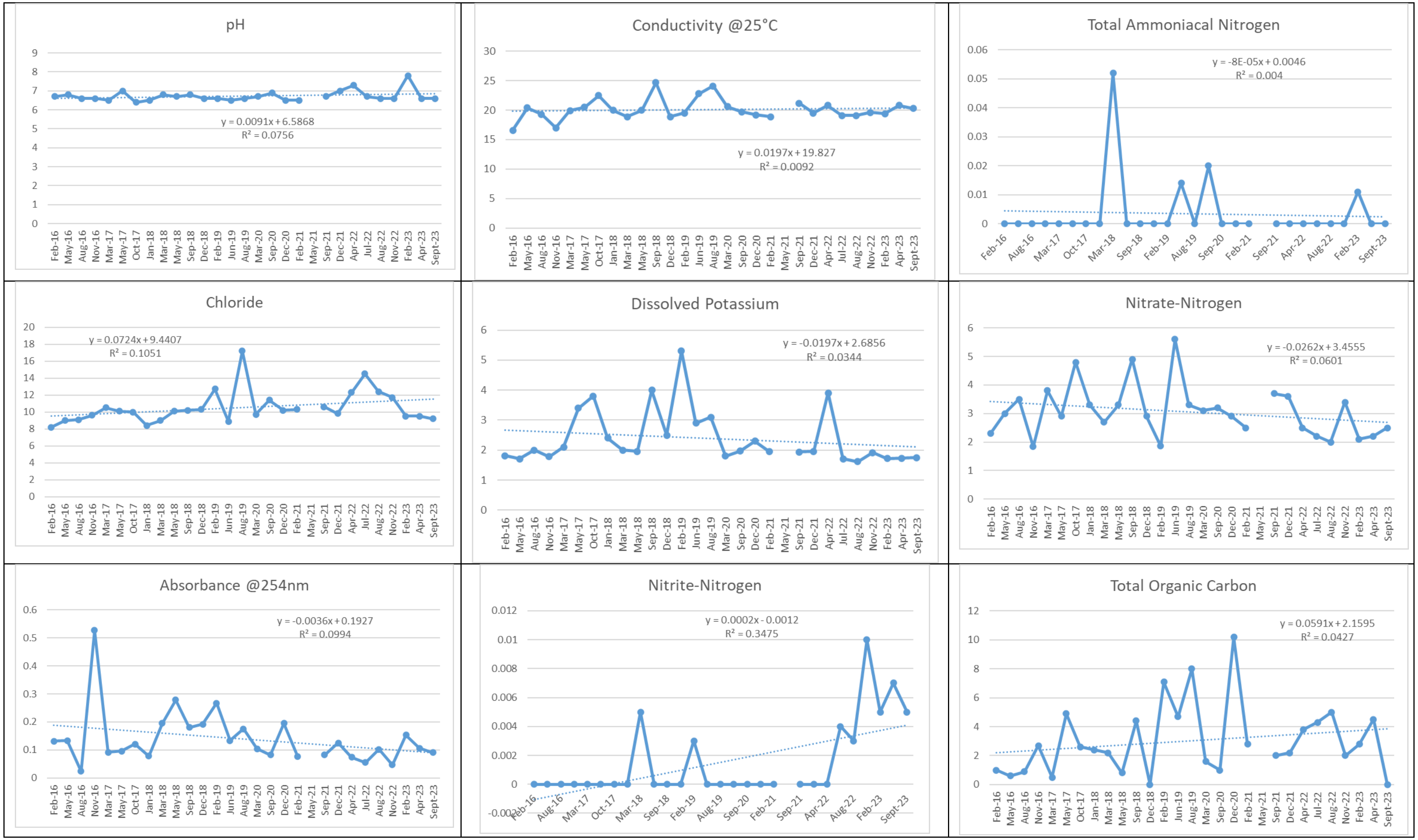
Table 19: Trend equation and R^2 value for monitoring wells - Ongaonga Closed landfill. Well 5356 is downgradient.

	Well	5356
	Equation	R^2 value
pH	y = 0.0091x + 6.5868	R^2 = 0.0756
Electrical Conductivity (EC)	y = 0.0197x + 19.827	R^2 = 0.0092
Dissolved Potassium	y = -0.0197x + 2.6856	R^2 = 0.0344
Chloride	y = 0.0724x + 9.4407	R^2 = 0.1051
Total Ammoniacal-N	y = -8E-05x + 0.0046	R^2 = 0.004
Nitrite-N	y = 0.0002x - 0.0012	R^2 = 0.3475
Nitrate-N	y = -0.0262x + 3.4555	R^2 = 0.0601
Total Organic Carbon (TOC)	y = 0.0591x + 2.1595	R^2 = 0.0427
Volatile Fatty Acids (VFA)	-	-
Absorbance at 254 nm	y = -0.0036x + 0.1927	R^2 = 0.0994

Table 20: Summary statistics and t-test results for Well No. 5356, the downgradient well. The t-tests were between the mean of the monitoring results, and the specified threshold value.

	Unit	n	Mean	Standard Deviation	Threshold	t-value	p-value	UPL(95%)
pH		28	6.72	0.29	7.7	-15.81	<0.0001	7.23
Electrical Conductivity (EC)	µS/cm	28	20.12	2.57	26.7	-15.88	<0.0001	24.57
Dissolved Potassium	mg/L	28	2.39	0.87	4.8	-15.89	<0.0001	3.90
Chloride	mg/L	28	10.51	1.67	250	-759.41	<0.0001	13.41
Total Ammoniacal-N	mg/L	28	0.02	0.01	0.41	-142.92	<0.0001	0.05
Nitrite-N	mg/L	28	0.01	0.00	0.09	-2234.84	-	0.01
Nitrate-N	mg/L	28	3.07	0.92	11.30	-47.26	<0.0001	4.67
Total Organic Carbon (TOC)	mg/L	28	3.27	2.59	11.5	-15.89	<0.0001	7.76
Volatile Fatty Acids (VFA)	mg/L	28	-	-	-	-	-	-
Absorbance at 254 nm		28	0.14	0.12	0.5	-16.32	<0.0001	0.35

Figure 20: Monitoring data Ongaonga Closed Landfill – 5356 (downgradient):



For pH, the downgradient monitoring bore at the Ongaonga Closed Landfill (Well 5356) shows no trend over time. The linear regression indicates a slight upward trajectory (y = 0.0091x + 6.587; R^2 = 0.0756) (Figure 20). The pH values remain consistently stable, with one isolated peak observed in Feb 2023. No specific causes were recorded for these peaks, and they may be attributable to analytical error or short-term disturbance from the surrounding environment. Mean pH at this well is 6.72 (Table 20), were slightly below the lower bound of the aesthetic drinking water range (7.00 - 8.50). The mean pH is also lower than the site-specific threshold value of 7.7 and the UPL(95%) of 7.23, supported by a strongly negative t-value (t = -15.81, p<0.0001).

The linear regression analysis shows a weak upward trend for EC over the monitoring period (y = 0.0197x + 19.827; R^2 = 0.0092) (Figure 20). EC concentrations appear consistently fluctuate in a range of 15 -20 µS/cm and have been stable since Sep 2021. The low R^2 value indicates that this apparent increase is not well developed and the peak value are caused by analytical error or short-term disturbance. The mean EC concentration at this Well is 20.12 µS/cm (Table 20), which is lower than the site-specific threshold value of 26.7 µS/cm and the UPL(95%) value of 24.57 µS/cm, supported by a strongly negative t-value (t = -15.88, p<0.0001).

Chloride contamination shows a steady trend over time. The linear regression indicates a very weak upward trajectory (y = 0.0724x + 9.441; R^2 = 0.1051) (Figure 20). The chloride concentration appeared consistently stable throughout the time, with two isolated peaks in August 2019 and July 2022. The mean chloride concentration is 10.51 mg/L (Table 20), which remains below the aesthetic drinking-water guideline of 250 mg/L and well below the UPL(95%) value of 13.41 mg/L, supported by a strong negative t-value (t = -759.41, p<0.0001).

Dissolved potassium concentrations at the downgradient monitoring bore (Well 5356) have remained low and effectively stable throughout the monitoring period (Figure 20). The linear regression indicates a weak decline trend (y = -0.0197x + 2.686; R^2 = 0.0344) (Figure 20). The mean dissolved potassium concentration at this well is 2.39 mg/L (Table 20), which is well below both the site-specific threshold value of 4.8 mg/L and the UPL(95%) of 3.90 mg/L, supported by a strong negative t-value (t = -15.89, p<0.0001).

TAN shows no trend over time (y = -8E-05x + 0.005; R^2 = 0.004) (Figure 20), with four isolated peaks. Mean TAN concentration at this well is 0.02 mg/L (Table 20), which is below the aesthetic threshold of 0.41 mg/L and the UPL(95%) of 0.05 mg/L, supported by a strong negative t-value (t = -142.92, p<0.0001).

The mean Nitrate-N concentration at this well is 3.07 mg/L (Table 20), and linear regression analysis shows a possible downward trend which does not reach statistical significance (y = -0.026x + 3.456; R^2 = 0.06, p=0 .2) (Figure 20). Nitrate-N concentrations remain well below the aesthetic drinking-water guideline of 11.3 mg/L and UPL(95%) value of 4.67 mg/L, supported by a strong negative t-value ($t = -47.26$, $p<0.0001$).

Nitrite-N concentrations have generally remained consistently low and stable, at or below the detection limit. From 2016 to April 2022, nitrite was only detected twice (Figure 20). In the more recent record (August 2022 to September 2023), it was detected on all six sampling occasions, with the highest reading recorded in December 2022. Nitrite is a reduction product of nitrate, usually present at a fraction of its concentration. In these records, mean nitrite (0.01 mg/L) is 300 times lower than mean nitrate (3.07 mg/L, Table 20). For these reasons, increases observed in the nitrite record are most likely to be caused by modest changes in subsurface oxidation-reduction conditions acting to moderate nitrate reduction, rather than the presence of a new source. Although in absolute terms, levels of nitrite remain low, the recent increase in detections does produce a slight upward trend for the overall record (Figure 20). Linear regression analysis indicates a weak upward temporal trend ($y = 0.0002x + 0.0012$; $R^2 = 0.348$). Importantly, the nitrite-N concentrations remain well below the nominated threshold limit.

TOC concentration remains relatively consistent at the Ongaonga Closed Landfill. TOC shows the appearance of a possible upward trend (Figure 20), but this does not reach statistical significance ($y = 0.0591x + 2.1595$; $R^2 = 0.0427$) (Figure 20). The mean TOC concentration at this well is 3.27 mg/L (Table 20), below both the site-specific threshold of 11.5 mg/L and the UPL(95%) value of 7.76 mg/L, supported by a strong negative ($t = -15.89$, $p<0.0001$).

A254 shows no trend ($y = -0.0036x + 0.1927$; $R^2 = 0.0994$,) (Figure 20). The mean A254 value at this well is 0.14, which is below both the site-specific threshold of 0.5 and the UPL(95%) value of 0.35, supported by a strong negative t-value ($t = -16.32$, $p<0.0001$).

Across the full suite of monitored parameters at the Ongaonga Closed Landfill, temporal patterns remain weak, with only minor trends observed that are largely influenced by short-term fluctuations rather than representing sustained changes throughout the monitoring period. An increase in recent nitrite suggests some minor changes in sub-surface nitrate reduction chemistry can occur over time; but this would not be considered unusual, and levels of nitrite remain very low. Concentrations for all parameters remain well below their respective thresholds and UPL (95%) values. Groundwater quality at the downgradient bore (Well 5356) remains consistently low across all indicators, with no evidence of enrichment indicative of landfill leachate.

3.2.7 Porangahau Closed Landfill

Figure 21: GIS map - Porangahau Closed Landfill

Groundwater at the Porangahau Closed Landfill is monitored at a single downgradient bore (Well 5661), which is located close to the Porangahau River. No upgradient monitoring well is currently included in the routine monitoring programme. Assessment of groundwater quality therefore relies on evaluation of absolute concentrations, temporal trend behaviour, and statistical comparison of monitoring results against site specific threshold values and UPL(95%) values.

Time-series monitoring results for the 9 targeted variables at the Porangahau Closed Landfill are presented in Figure 22. Means, standard deviations, t-tests results are provided in Table 22. For each variable, the t-tests were used to determine whether there was a statistical difference between the mean of monitoring results and the specified threshold value. Statistically significant ($p<0.05$) negative t-values denote cases where the mean of the monitoring results is distinguishably less than the threshold. The linear regression equations and their associated statistical measures are provided in Table 21.

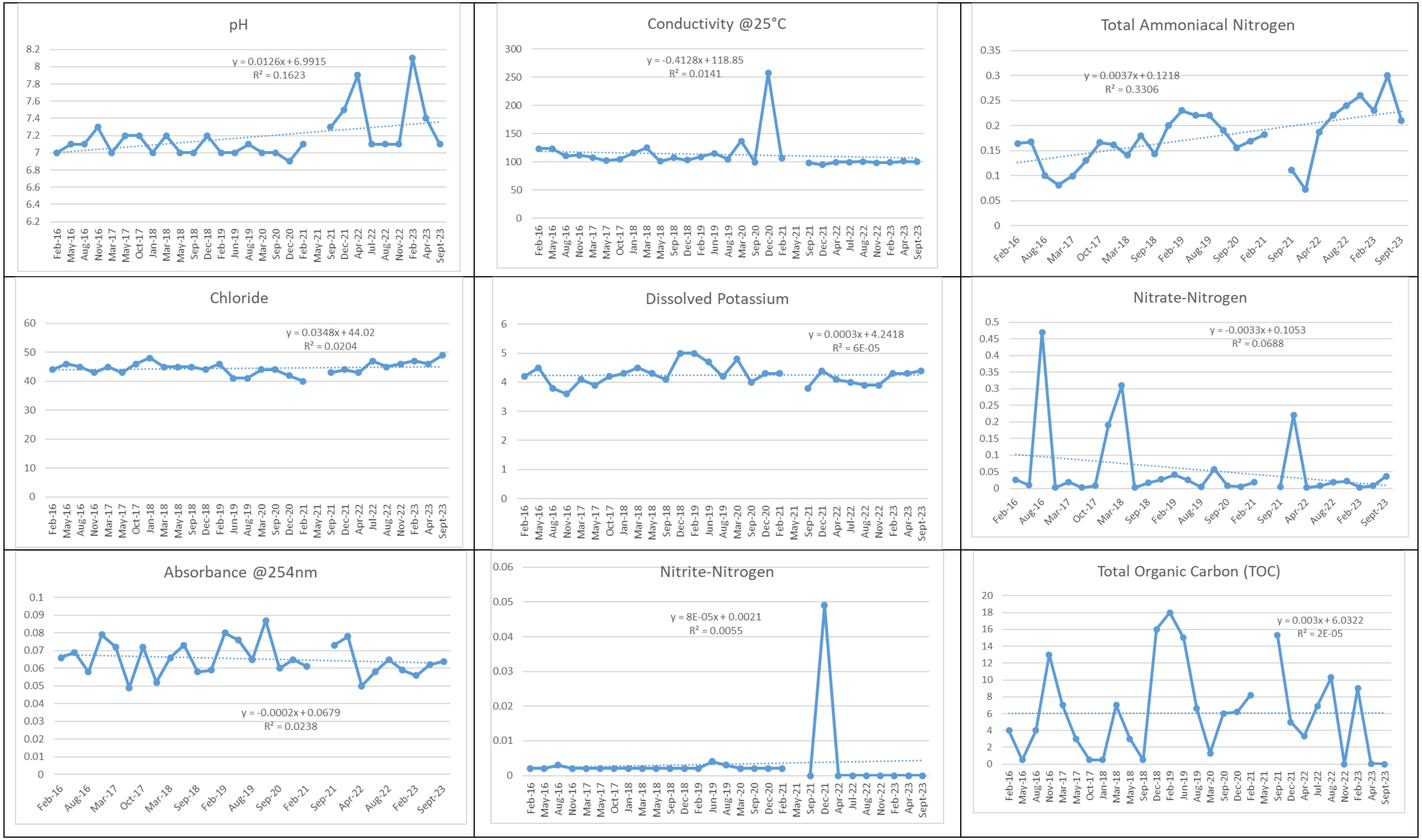
Table 21: trend equation and R^2 value for monitoring wells - Porangahau Closed landfill. Well 5661 is downgradient.

Well	5661	
	Equation	R^2 value
pH	y = 0.0126x + 6.9915	$R^2 = 0.1623$
Electrical Conductivity (EC)	y = -0.4128x + 118.85	$R^2 = 0.0141$
Dissolved Potassium	y = - 0.0003x + 4.2418	R^2 = 6E-05
Chloride	y = 0.0348x + 44.02	$R^2 = 0.0204$
Total Ammoniacal-N	y = 0.0037x + 0.1218	$R^2 = 0.3306$
Nitrite-N	y = 8E-05x + 0.0021	$R^2 = 0.0055$
Nitrate-N	y = -0.0033x + 0.1053	$R^2 = 0.0688$
Total Organic Carbon (TOC)	y = 0.003x + 6.0322	R^2 = 2E-05
Volatile Fatty Acids (VFA)	-	-
Absorbance at 254 nm	y = -0.0002x + 0.0679	$R^2 = 0.0238$

Table 22: Summary statistics and t-test results for Well No. 5661, the downgradient well. The t-tests were between the mean of the monitoring results, and the specified threshold value

			Mean	**Standard Deviation**	**Threshold**	**t-value**	**p-value**	**UPL(95%)**
pH		n	7.18	0.27	7.7	-11.99	<0.0001	7.65
Electrical Conductivity (EC)	μS/cm	28	112.73	29.97	174.1	-11.06	<0.0001	164.68
Dissolved Potassium	mg/L	28	4.25	0.34	5.3	-15.45	<0.0001	4.84
Chloride	mg/L	28	44.54	2.10	250	-517.95	<0.0001	48.17
Total Ammoniacal-N	mg/L	28	0.18	0.05	0.41	-22.53	<0.0001	0.27
Nitrite-N	mg/L	28	0.00	0.01	0.90	-452.32	<0.0001	0.02
Nitrate-N	mg/L	28	0.06	0.11	11.30	-542.33	<0.0001	0.25
Total Organic Carbon (TOC)	mg/L	28	6.54	5.30	19.2	-13.79	<0.0001	15.72
Volatile Fatty Acids (VFA)	mg/L	28	-	-	-	-	-	-
Absorbance at 254 nm		28	0.07	0.01	0.1	-30.41	<0.0001	0.08

Figure 22: Monitoring data Porangahau Closed Landfill – 5661 (downgradient):



For pH, the downgradient monitoring bore at the Porangahau Closed Landfill (Well 5661) shows a consistently stable, but with significant peaks in April 2022 and February 2023. The linear regression indicates an upward trajectory ($y = 0.013x + 6.99$; $R^2 = 0.162$) (Figure 22). However, the record shows that this increase is not uniform across the entire monitoring period and is influenced primarily by higher values observed in the more recent samples. Mean pH at this well is 7.18 (Table 22), which is within the aesthetic drinking water range (7.00–8.50). The mean pH is also significantly below both the specified threshold values of 7.7 and the UPL(95%) of 7.65, supported by a strongly negative t-value ($t = -11.99$, $p<0.0001$).

As noted in Section 1.4.1, Porangahau landfill is a coastal site situated about 50 metres from the tidally influenced Porangahau River. EC and chloride results at this site are therefore substantially higher than at the other locations, presumably reflecting the sea-salt influence in the coastal soils and/or the groundwater. Visual inspection of the EC record (Figure 22) suggests a very weak downward trend throughout the time, although this does not show up as being statistically significant due to the variance introduced by an apparent outlier in Dec 2020 ($y = -0.4128x + 118.85$; $R^2 = 0.0141$). The mean EC concentration at this well is 112.73 µS/cm (Table 22), which remains well below both the site-specific threshold value of 174.1 µS/cm and the UPL(95%) value of 164.68 µS/cm, supported by a strongly negative t-value ($t = -11.06$, $p<0.0001$).

The time-series record shows chloride values fluctuate within a narrow range around a stable long-term concentration. Visual inspection (Figure 22) suggests a slight upward trajectory, possibly due to the final observation in the record also being the highest; however, this is not statistically significant. The linear regression indicates a slight upward trajectory ($y = 0.035x + 44.02$; $R^2 = 0.020$, $p=0.47$). The mean chloride concentration for this well is 44.54 mg/L (Table 22), which remains below the aesthetic drinking water guideline of 250 mg/L and the UPL(95%) value of 48.17 mg/L, supported by a strongly negative t-value ($t = -517.95$, $p<0.0001$).

The linear regression for dissolved potassium concentration also shows no evidence for a trend over time ($y = 0.0003x + 4.242$; $R^2 = 6E\text{-}05$) (Figure 22). The mean dissolved potassium concentration for this well is 4.25 mg/L (Table 22), which remains below the site-specific threshold value of 5.3 mg/L and the UPL(95%) value of 4.84 mg/L, supported by a strongly negative t-value ($t = -15.45$, $p<0.0001$).

Of the three forms of nitrogen, Nitrate-N concentrations at this well are generally either very low (0.05 mg/L or below), or a bit higher but still low, with four cases where levels increased to between 0.15 and 0.5 mg/L (Figure 22). The mean nitrate-N concentration is 0.06 mg/L (Table 22), and the linear regression analysis shows no evidence for a trend ($y = -0.003x + 0.105$; $R^2 = 0.069$) (Figure 22). All values remain well below the drinking-water health guideline of 11.3 mg/L, supported by a strongly negative t-value ($t = -542.33$, $p<0.0001$).

As expected, nitrite-N concentrations remain extremely low throughout the monitoring record. Most results are just at or below the detectable level (Table 22), and regression analysis provides no evidence for a trend. There is one isolated peak observed in December 2021, which is also low in absolute terms (ca. 0.05 mg/L) but may be cause by analytical error.

Interestingly, mean TAN at this well (0.18 mg/L) is three times higher than that of nitrate (Table 22). Linear regression also shows an upward trend over time ($y = 0.004x + 0.122$; $R^2 = 0.331$, $p=0.0014$) (Figure 22). While the time series shows the increases occurring in the recent samples. Currently, TAN concentrations remain well below the aesthetic drinking water guideline of 0.41 mg/L ($t = -22.53$, $p<0.0001$, Table 22). In one of the most recent samples, TAN exceeded the UPL95% of 0.27 mg/L. The coexistence of elevated TAN with low nitrate suggests that the ammonium enrichment in this environment is related to natural geogenic causes. Specifically, (Xiong, 2024) report that in coastal aquifers, less-degraded dissolved organic matter (DOM) can cause higher production and levels of ammoniacal-N, from organic matter decomposition.

Measures of total organic carbon (TOC) and UV absorbance at 254 nm (UV254), which is an indicator of dissolved organic compounds, show variability but no evidence for an upward or downward trend over time. Least-squares lines for both TOC and A254 show slopes that are indistinguishable from zero (Table 21). The mean TOC concentration at this well is 6.54 mg/L (Table 22), which is well below both the site-specific threshold of 19.2 mg/L and the UPL(95%) value of 15.72 mg/L, supported by a strongly negative t-value ($t = -13.79$, $p<0.0001$). The mean A254 value of 0.07 (Table 22), below both the site-specific threshold of 0.1 and the UPL(95%) value of 0.08, supported by a strongly negative t-value ($t = -30.41$, $p<0.0001$).

Across the full suite of monitored parameters at the Porangahau Closed Landfill, the only temporal trend is for TAN. This appears to be increasing, the cause is most likely to be natural (because, as outlined above, natural processes can result in higher TAN in coastal aquifers, and nitrate-N remains low). Concentrations for all parameters remain well below their respective thresholds and UPL(95%) values. Groundwater quality at Well 5661 remains consistently low across all indicators, with no evidence of systematic enrichment that would be indicative of landfill leachate influence.

3.3 Framework application

In the previous section (Section 3.2) little or no evidence was found that any of the closed landfills have a significant impact on groundwater quality.

In this part of the thesis, the results from analysis of the groundwater monitoring data are incorporated into the wider source-pathway-receptor based framework (Section 2.2), to derive semi-quantitative evidence-based risk-rankings for each of the landfills.

This approach builds further than the preliminary (or 'informal') risk characterisation approach (Section 3.1, Table 5) in two main ways. The first is through formal inclusion (and consideration) of groundwater monitoring records, and the second is through systematic application of the source-pathway-receptor approach for each landfill, as a preferred overall risk methodology.

3.3.1 Waipawa Closed Landfill

Source and containment assessment

The Waipawa Closed Landfill formerly operated as a municipal waste disposal site serving the Waipawa community.

This site is an unlined landfill, with no documented soil permeability assessment or engineered design for the landfill basin. In addition, the depth and construction details of the landfill capping have not been confirmed. Despite this, downgradient groundwater shows no clear landfill influence: overall results of the monitoring programme assessment (Section 3.2.1) are summarised and interpreted in Table 23. This data shows that the Waipawa closed landfill is not (or no longer) acting as a significant source of contaminants to groundwater. The most likely reason for this lack of a source signature in groundwater is the age of this landfill, because containment is minimal.

Pathway and receptor assessment

Refer also to the preliminary risk matrix (Table 5) in Section 3.1. Due to unusual site topography, Well 5501 is regarded as being downgradient of the landfill, surface-water and groundwater both move away from the Waipawa River at this location. The closed drinking-water bore is located approximately 2.3 km from the landfill and not directly located on the groundwater flow path. The nearest surface water body (Waipawa River) is relatively close (<100 m). Based on available site information, the long groundwater travel distance, soil adsorption (attenuation) processes, organic contaminant degradation, and dilution within the underlying aquifer may have all had a historic impact on measurable leachate migration.

Monitoring data analysis

Table 23. Summary of monitoring evidence assessment for Waipawa closed landfill

Parameters	Upgradient Well 5686	Downgradient Well 5501	Temporal trend	Interpretation in relation to landfill influence
pH	Mean pH within aesthetic guideline value	Mean pH slightly below aesthetic guideline value	Well 5686 slight upward trend Well 5501 shows no trend	Higher upgradient pH and absence of downgradient response indicate control by background geochemistry or land-use factors (e.g. agricultural liming), with no evidence of leachate influence
EC	Mean EC concentrations remain below the site-specific threshold limit	Mean EC concentrations remain below the site-specific threshold limit	Well 5686 shows long-term stability Well 5501 shows recent increase not sustained across record	EC distribution is opposite to that expected under leachate impact; downgradient EC remains well below upgradient and background-dominated
Chloride	Mean chloride lower than aesthetic guideline value	Mean chloride substantially lower than aesthetic guideline value	Both wells show upward trend with recent increase not sustained across record	Higher upgradient chloride concentration reflects regional hydrochemistry or land-use inputs rather than a downgradient leachate plume
Dissolved Potassium	Mean potassium concentrations lower than threshold limit	Mean potassium concentration higher than upgradient and lower than threshold limit	Well 5686 shows weak declining trend (two isolated peaks) Well 5501 shows weak upward trend by recent increase	Concentrations and trends consistent with natural groundwater variability; not indicative of leachate migration
TAN	Exceed aesthetic guideline value	Exceed aesthetic guideline value	Both wells show very weak declining trend, long-term stability	Presence at both wells, indicates natural biogeochemical processes or non-landfill land-use influences
Nitrate-N	Mean concentration significantly lower than aesthetic guideline value	Mean concentration significantly lower than aesthetic guideline value	Well 5686 shows significant declining trend Well 5501 shows long-term stability	Minor to negligible environmental risk
Nitrite-N	Mean concentration well below the aesthetic guideline value	Mean concentration well below the aesthetic guideline value	Both wells show long-term stability	Presence at both wells, indicates natural biogeochemical processes or non-landfill land-use influences
TOC	Mean concentration well below site-specific threshold limit	Low and stable; statistically indistinguishable from upgradient	No trends; strong correspondence between wells	TOC controlled by regional groundwater conditions; no evidence of downgradient organic enrichment
A254	Mean concentration well below site-specific threshold limit	Mean concentration well below site-specific threshold limit	No temporal trends	Minor difference may indicate subtle variation in DOM composition; not supported by TOC or other leachate indicators

Based on the temporal variability observed across the monitoring record in Section 3.2, TAN is the only parameter which exceed the aesthetic drinking water guideline value at both the upgradient and downgradient wells. Importantly, TAN concentrations are comparable at both locations and show no statistically significant temporal trends. This indicates that the exceedance is unlikely to be attributable to landfill leachate. Instead, the consistent spatial distribution and temporal stability of TAN suggest control by natural biogeochemical processes and/or broader land-use influences within the catchment.

Monitoring results do not show increasing contaminant concentrations over time, suggesting that natural subsurface conditions provide functional containment and attenuation of any current residual loads, despite the absence of engineered structures

Considering the weight of evidence across all monitored parameters, the Waipawa Closed Landfill is interpreted to exert minimal to negligible influence on downgradient groundwater quality and therefore represents a low environmental risk.

Outcomes

The assessment has been based on the preliminary risk matrix (Section 3.1) and the groundwater monitoring data analysis (Section 3.2). This site contains historical municipal waste and lacks engineered containment systems that pose groundwater contamination. The groundwater monitoring parameters are detectable but stable. Monitoring records in the downgradient bore remain consistently below relevant thresholds, with the exception of TAN. TAN is also elevated in the upgradient bore, indicating a likely regional or natural influence rather than a landfill-derived source. The absence of more than minor downgradient enrichment confirms that significant contaminant transport is not occurring. In addition, the nearest downstream drinking water bore is located at a considerable distance from the site and is not directly with the inferred groundwater flow path.

Overall, the Waipawa Closed Landfill demonstrates limited measurable environmental impact and is appropriately classified as low risk. Ongoing routine monitoring remains appropriate. The evidence is sufficient to support the consent holder applying to the consent authority to reduce the monitoring frequency .

3.3.2 Waipukurau Closed landfill

Source and containment assessment

The Waipukurau Closed Landfill formerly operated as a municipal waste disposal site serving the Waipukurau community.

This site is an unlined landfill, with no documented soil permeability assessment or engineered design for the landfill basin. In addition, the depth and construction details of the landfill capping have not been confirmed. Despite this, downgradient groundwater shows no clear landfill influence: overall results of the monitoring programme assessment (Section 3.2.2) are summarised and interpreted in Table 24. This data shows that Waipukurau closed landfill is not (or no longer) acting as a significant source of contaminants to groundwater. The most likely reason for this lack of a source signature in groundwater is the age of this landfill, because containment is minimal.

Pathway and receptor assessment

Refer also to the preliminary risk matrix (Table 5) in Section 3.1. At Waipukurau, the nearest drinking-water bore is located less than 1 km from the landfill; however, it is situated upgradient, which significantly reduces the potential for contaminant migration. The nearest downgradient bore is approximately 3.3 km away, providing a substantial attenuation distance. Based on the available site information, the long groundwater travel distance, combined with soil adsorption (attenuation) processes, organic contaminant degradation, and dilution within the underlying aquifer, is likely to have historically limited measurable leachate migration.

Monitoring data analysis

Table 24. Summary of monitoring evidence assessment for Waipukurau closed landfill

Parameters	Downgradient Well 5503	Temporal trend (Well 5503)	Comparison with Thresholds / UPL(95%)	Interpretation in relation to landfill influence
pH	Mean pH slightly below aesthetic range; mildly acidic conditions	Apparently upward trend over time	Mean pH below site-specific threshold and UPL(95%)	Stable, mildly acidic groundwater; gradual increase likely reflects geochemistry or land-use factors (e.g. agricultural liming), with no evidence of leachate influence
EC	Mean EC concentrations remain below the site-specific threshold limit	Upward trend influenced by recent increase	Mean EC well below both site-specific threshold and UPL(95%)	EC levels remain far below nominated thresholds, no evidence of leachate influence; recent uptick and age of landfill suggest another nearby source
Chloride	Mean chloride lower than aesthetic guideline value	Upward trend with recent increase – tracks EC	Mean chloride well below aesthetic guideline and UPL(95%)	Chloride concentration remains far below nominated thresholds, no evidence of leachate influence. As above – likely to be another nearby source.
Dissolved Potassium	Mean potassium concentrations lower than threshold limit	No trend	Mean concentration well below site-specific threshold and UPL(95%)	Potassium concentrations and trends consistent over time and below nominated thresholds, no indicative of leachate migration
TAN	Very low concentrations	Upward trend driven by recent samples	Mean concentration well below aesthetic guideline and UPL(95%)	Concentrations remain very low, even with the recent increase, and are not indicative of leachate migration
Nitrate-N	Mean concentration significantly lower than aesthetic guideline value	No trend	Mean concentration well below health guideline and UPL(95%)	Nitrate concentrations and trends consistent over time and below nominated thresholds, no indicative of leachate migration
Nitrite-N	Not detected	No trend - consistently below detection limits	Below detection limits	Not detected
TOC	Mean concentration remain stable and well below site-specific threshold limit	No trend	Mean concentration well below site-specific threshold and UPL(95%)	no evidence of leachate migration
A254	Very low absorbance values	Possible weak upward trend	Mean concentration well below site-specific threshold limit and UPL(95%)	No indication of aromatic organic enrichment or leachate influence

Based on the temporal variability observed across the monitoring record in Section 3.2, no monitored parameter at the Waipukurau site exceeds its respective aesthetic drinking water guideline, site-specific threshold, or UPL(95%) value. Although several parameters (including pH, electrical conductivity, chloride, and total ammoniacal nitrogen) exhibit apparent upward trends, these are not uniform across the full monitoring period and are primarily influenced by higher values in more recent samples. Currently, these elevated values remain well below levels typically associated with landfill leachate. However, the recent increase shown in results for EC and chloride (in particular) suggests the influence of another newer nearby source. This is most likely to be related to activities on an adjacent site for two reasons: (a) the age of the landfill, making it very unlikely that EC and chloride in leachate would suddenly increase after such a long time, and (b) the fact of the existence of a nearby WWTP and refuse transfer operation.

Monitoring results do not indicate sustained increases in contaminant concentrations attributable to the landfill over time, suggesting that natural subsurface conditions are providing effective attenuation of any residual contaminant loads, despite the absence of engineered containment measures. However, the source of recent increases in EC, chloride and TAN are aspects that warrants further (separate) investigation.

Considering the weight of evidence across all monitored parameters, the Waipukurau Closed Landfill itself is interpreted to exert minimal to negligible influence on downgradient groundwater quality and therefore represents a low environmental risk under current conditions.

Outcomes:

The assessment is based on the preliminary risk matrix (Section 3.1) and the groundwater monitoring data analysis (Section 3.2). This site contains historical municipal waste and lacks engineered containment systems that pose groundwater contamination. However, groundwater monitoring parameters in the downgradient bore are detectable, but the inferred landfill leachate component remains low and stable over time. All monitored parameters are consistently below relevant aesthetic drinking-water guidelines, site-specific threshold values, and UPL(95%) baseline concentrations. Although apparent upward trends have been identified for some parameters, these trends are most likely linked to another source. In addition, the nearest downgradient drinking-water bore is located at a considerable distance from the site, thereby limiting the potential exposure of sensitive receptors.

Overall, the Waipukurau Closed Landfill demonstrates limited measurable environmental impact and is appropriately classified as low risk. Ongoing routine groundwater monitoring remains appropriate, as does investigation of groundwater contamination originating from the adjacent site. In the case where the consent holder applies to reduce monitoring activities, such an application would be considered, and the site may be eligible for a reduced monitoring frequency.

3.3.3 Takapau Closed Landfill

Source and containment assessment

The Takapau Closed Landfill formerly operated as a municipal waste disposal site serving the Takapau community.

The site is an unlined landfill, with no documented soil permeability testing. No details regarding the depth and composition of the landfill capping are recorded. Despite this, downgradient groundwater shows no clear landfill influence: overall results of the monitoring programme assessment (Section 3.2.3) are summarised and interpreted in Table 25. This data shows that Takapau closed landfill is not (or no longer) acting as a significant source of contaminants to groundwater. The most likely reason for this lack of a source signature in groundwater is the age of this landfill, because containment is minimal.

Pathway and receptor assessment

Refer also to the preliminary risk matrix (Table 5) in Section 3.1. No upgradient monitoring bore is available at this site. However, groundwater chemistry in the downgradient monitoring bore remains consistently low and stable, indicating no contaminant transport from the landfill. The nearest downgradient bore is approximately 3.4 km away, providing a substantial attenuation distance. Based on the available site information, the long groundwater travel distance, together with soil adsorption (attenuation) processes, organic contaminant degradation, and dilution within the underlying aquifer, the likelihood of contaminant migration resulting in adverse effects on downgradient receptors is considered low under current conditions.

Monitoring data analysis

Table 25. Summary of monitoring evidence assessment for Takapau closed landfill

Parameters	Downgradient Well 5660	Temporal trend	Comparison with Thresholds / UPL(95%)	Interpretation in relation to landfill influence
pH	Mildly acidic conditions	Stable over time, no trend, isolated short-term peak only	Mean pH below site-specific threshold and UPL(95%)	Stable pH, no evidence of leachate influence
EC	Very low ionic strength groundwater	Effectively constant over time	Mean EC well below both site-specific threshold and UPL(95%)	No evidence of leachate influence
Chloride	Low concentrations	Minor variability, weak declining trend over time	Mean chloride well below aesthetic guideline and UPL(95%)	No chloride enrichment, no evidence of leachate influence
Dissolved Potassium	Low and stable concentrations	No trend	Mean concentration well below site-specific threshold and UPL(95%)	No indicative of leachate migration
TAN	Very low concentrations	No trend	Mean concentration well below aesthetic guideline and UPL(95%)	Concentrations remain very low, no indicative of leachate migration
Nitrate-N	Dominant nitrogen species	No trend	Mean concentration well below health guideline and UPL(95%)	No indicative of leachate migration
Nitrite-N	Negligible concentrations	Consistently below detection limits	Below detection limits	Not detected
TOC	Low concentrations	No trend	Mean concentration well below site-specific threshold and UPL(95%)	No evidence of leachate migration
A254	Very low absorbance values	Weak declining trend	Mean concentration well below site-specific threshold limit and UPL(95%)	No indicative of leachate migration

Based on the temporal variability observed across the monitoring record in Section 3.2.3 and the summary presented in Table 25, no monitored parameter at the site exceeds its respective aesthetic drinking-water guideline, site-specific threshold value, or UPL(95%) baseline concentration. Groundwater quality in the downgradient monitoring bore is characterised by generally low and stable concentrations across all parameters.

Monitoring parameters in the downgradient monitoring bore either show no trend or a weak declining trend over time, suggesting that the source is no longer active and/or natural attenuation processes within the subsurface environment are effectively limiting contaminant migration. Considering the collective weight of evidence across all monitored parameters, the site is interpreted to have a minimal to negligible influence on downgradient groundwater quality and is therefore considered to represent a low environmental risk under current conditions.

Outcomes:

This assessment is based on the preliminary risk matrix (Section 3.1) and the groundwater monitoring data analysis (Section 3.2.3). The site contains historical municipal waste and lacks engineered containment systems that could pose a potential risk to groundwater quality. However, groundwater monitoring results from the downgradient bore indicate that contaminant concentrations are detectable but remain low and stable (or weakly declining) over time.

All monitored parameters are consistently below relevant aesthetic drinking-water guidelines, site-specific threshold values, and UPL(95%) baseline concentrations. The nearest downgradient drinking-water bore is located at a considerable distance from the site, thereby substantially reducing the potential exposure of sensitive receptors.

Overall, the Takapau closed landfill demonstrates limited measurable environmental impact and is appropriately classified as low risk under current conditions. Ongoing routine groundwater monitoring remains appropriate. In the case where the consent holder applies to reduce monitoring activities, such an application would be considered, and the site may be eligible for a reduced monitoring frequency.

3.3.4 Tamumu Closed Landfill

Source and containment assessment

The Tamumu Closed Landfill formerly operated as a municipal waste disposal site serving the local community. The site is an unlined landfill, with no documented soil permeability testing or engineered liner system. Details regarding the depth, material composition, and construction of the landfill capping are not available.

Despite this, downgradient groundwater monitoring does not show a consistent landfill influence. The overall results of the monitoring programme assessment (Section 3.2.4) are summarised and interpreted in Table 26. TAN exceeds the aesthetic drinking-water guideline but remains below the UPL(95%) baseline concentration, all monitored parameters are consistently below relevant guideline, threshold, and baseline values.

Pathway and receptor assessment

Refer to the preliminary risk matrix (Table 5) in Section 3.1. No upgradient monitoring bore is available at the Tamumu Closed Landfill to directly characterise background groundwater quality. However, groundwater chemistry measured in the downgradient monitoring bore (Well 5504) remains low and stable over time.

The nearest downgradient drinking-water bore is located approximately 500 m from the site, and the Tukituki River is located approximately 100 m from the site. Therefore, there is a short groundwater travel distance between the landfill and potential sensitive receptors. However, based on available site information and analytical results (Section 3.2.4), the likelihood of contaminant migration from the Tamumu Closed Landfill resulting in adverse effects on downgradient receptors is low under current conditions.

Monitoring data analysis

Table 26. Summary of monitoring evidence assessment for Tamumu closed landfill

Parameters	Downgradient Well 5504	Temporal trend (Well 5504)	Comparison with Thresholds / UPL(95%)	Interpretation in relation to landfill influence
pH	Mean pH of 6.81 slightly below aesthetic range	No trend	Mean pH below site-specific threshold and UPL(95%)	Mildly acidic but stable, pH value trend to increase towards to aesthetic range
EC	Mean EC concentrations remain low over time	Upward trend influenced by recent increase	Mean EC well below both site-specific threshold and UPL(95%)	Recent variability noted, but concentrations remain low and within acceptable limits
Chloride	Mean chloride concentration is low	Upward trend influenced by recent increase – tracks with EC	Mean chloride well below aesthetic guideline and UPL(95%)	Chloride concentration remains far below nominated thresholds, no evidence of leachate influence
Dissolved Potassium	Mean potassium concentration is low	No trend	Mean concentration well below site-specific threshold and UPL(95%)	Potassium concentrations and trends consistent over time and below nominated thresholds, no indicative of leachate migration
TAN	Mean concentration of 1.67 mg/L	Weak declining trend	Mean concentration exceeds aesthetic guideline (0.41 mg/L) but below UPL(95%)	TAN indicates historically influence from organic matter decomposition, TAN value decrease over time, closer toward to aesthetic guideline value
Nitrate-N	Mean concentration significantly lower than aesthetic guideline value	Weak decline trend over time	Mean concentration well below drinking-water guideline and UPL(95%)	Nitrate concentrations fluctuate over time, but remain below nominated thresholds
Nitrite-N	Not detected	Consistently below detection limits	Below detection limits	Not detected
TOC	Mean concentration is low	Upward trend	Mean concentration well below site-specific threshold and UPL(95%)	Increasing trend observed but concentrations remain low
A254	Very low absorbance values	Weak upward trend	Mean concentration well below site-specific threshold limit and UPL(95%)	Increasing trend observed but concentrations remain low

Based on the temporal variability observed across the monitoring record in Section 3.2.4 and the summary presented in Table 26, groundwater quality in the downgradient monitoring bore (Well 5504) is generally characterised by low concentrations across the majority of monitored parameters. All parameters remain below their respective site-specific threshold values and UPL(95%) baseline concentrations, except for TAN, which exceeds the aesthetic drinking-water guideline value. Notably, EC, chloride, TOC, and A254 show weak upward trends in recent years, and it remains to be seen (through future monitoring) whether these trends continue. Despite these trends, concentrations remain well within acceptable limits and below nominated thresholds. In contrast, TAN concentrations display a declining temporal trend and remain below the UPL (95%) baseline concentration. This pattern is consistent with a diminishing influence from historical organic matter decomposition rather than an active or increasing landfill-derived source. The landfill plume is low in oxygen, and microbial degradation of organic matter can convert organic nitrogen to ammonia. A decrease in TAN may reflect a transition to less reducing leachate conditions and/or depletion of readily degradable organic matter in the subsurface following decades of degradation.

Outcomes:

This assessment is based on the preliminary risk matrix (Section 3.1) and the groundwater monitoring data analysis (Section 3.2.4). The Tamumu Closed Landfill contains historical municipal waste and lacks engineered containment systems that could pose a potential risk to groundwater quality. However, groundwater monitoring results from the downgradient bore indicate that contaminant concentrations are generally low.

Most monitored parameters remain below site-specific threshold values and UPL(95%) baseline concentrations, except for total ammoniacal nitrogen (TAN), which exceeds the aesthetic drinking-water guideline value. Importantly, TAN concentrations show a declining temporal trend and remain below the UPL(95%) baseline, indicating attenuation over time and suggesting that the exceedance is not indicative of an active or increasing landfill-derived contaminant source.

Overall, the Tamumu Closed Landfill demonstrates limited measurable environmental impact and is appropriately classified as medium risk under current conditions. Ongoing routine groundwater monitoring remains appropriate, primarily because of higher readings for some parameters in more recent samples.

3.3.5 Tikokino Closed landfill

Source and containment assessment

The Tikokino Closed Landfill formerly operated as a municipal waste disposal site serving the Tikokino community.

The site is an unlined landfill, with no documented soil permeability testing for the landfill basin and the depth and construction details of the landfill capping are not available. Despite this, downgradient groundwater monitoring results indicate no clear evidence of landfill-related contaminant release. The overall results of the groundwater monitoring programme assessment (Section 3.2.5) are summarised and interpreted in Table 27. Monitoring data show that groundwater quality in the downgradient wells remains comparable to upgradient conditions, with all parameters consistently below nominated threshold values and drinking-water guidelines.

Pathway and receptor assessment

Refer to the preliminary risk matrix (Table 5) in Section 3.1. Groundwater at the Tikokino Closed Landfill is monitored using one upgradient bore (Well 4228) and two downgradient bores (Wells 5662 and 4227). The nearest downgradient drinking-water bores are located approximately 2.2 km away, a considerable distance from the site, providing substantial attenuation between the landfill and sensitive receptors. The nearest downgradient surface-water body (Mangaoho Stream) is approximately 100 m away from the site.

Based on available site information, the long groundwater travel distances, together with soil adsorption (attenuation), organic contaminant degradation, and dilution within the underlying aquifer, are expected to significantly reduce the likelihood of contaminant migration causing adverse effects on downgradient receptors . Observed groundwater quality in both upgradient and downgradient wells remains generally low and stable, supporting the conclusion that measurable contaminant transport from the landfill is negligible under current conditions.

Monitoring data analysis

Table 27. Summary of monitoring evidence assessment for Tikokino closed landfill

Parameters	Upgradient Well 4228	Downgradient Well 5662 & 4227	Temporal trend	Comparison with Thresholds / Guidelines	Interpretation in relation to landfill influence
pH	Mean pH 6.85, slightly below aesthetic range	Mean pH for Well 5662 is 6.74 Mean pH for Well 4227 is 6.75	No or weak trends with slopes close to zero	All means well below site-specific thresholds	Comparable values across all wells, mildly acidic observed but stable
EC	Mean 24.22 µS/cm, stable	Mean 35.93 µS/cm (Well 5662); 12.95 µS/cm (Well 4227	No meaningful trends. Isolated event was detected in Well 4227 in Sept 2018, but not repeated.	All values well below site-specific thresholds	Low EC across network, differences reflect local hydrogeology rather than landfill impact. Well 5662 shows the higher value than upgradient well (more likely influence by surrounding environment)
Chloride	Mean chloride concentration is 11.29 mg/L	Mean chloride concentration is low for both downgradient well	All monitoring wells stable over time. Isolated peaks including Sept 2018 event.	All values far below aesthetic guideline (250 mg/L)	No downgradient enrichment; concentrations at Well 5662 likely influenced by regional land use
Dissolved Potassium	Mean potassium 1.93 mg/L	Mean 2.34 mg/L (5662); 1.50 mg/L (4227)	No trends (only isolated spikes)	All means well below threshold values	Potassium concentrations and trends consistent over time and below nominated thresholds, no indicative of leachate migration
TAN	Mean concentration is low or lower than the detectable level	Mean concentrations are low or lower than the detectable level	Stable over time (isolated spikes only)	All values well below threshold (0.41 mg/L)	No indicative of leachate migration
Nitrate-N	Mean concentration is low	Mean concentration 3.30 mg/L (5662); 1.59 mg/L (4227)	Well 4228 shows decline trend over time Both downgradient wells shows steady over time with only isolated spick	All values well below aesthetic guideline (11.3 mg/L)	Higher downgradient values at Well 5662 likely reflect catchment influences rather than landfill Mean concentration at Well 4227 is lower than detectable level (one isolated spick or analytical error in September 2018)
Nitrite-N	Very low; occasional isolated detections	Very low or below detectable level	No meaningful trends for downgradient wells; decrease in upgradient well may be linked with land-use practices.	All values below nominated thresholds	Not detected
TOC	Mean 2.94 mg/L	Mean 2.95 mg/L (5662); 2.19 mg/L (4227)	Well 4228 shows weak upward trend over time Both downgradient wells show no trend over time (isolated spikes only)	All values well below site-specific thresholds	Low and stable organic carbon,
A254	Mean 0.04	Mean 0.06 (both downgradient wells)	Well 4228 and 5662 shows weak decline trend over time Well 4227 show steadies over time (one isolated spick)	All values well below threshold values	Slightly higher downgradient values, but not indicative of sustained leachate influence

Based on the temporal variability observed across the monitoring record in Section 3.2 and the summary presented in Table 27, all monitored parameters at the Tikokino Closed Landfill remain well below their respective site-specific thresholds or aesthetic drinking-water guideline values. Groundwater quality across the network, including the upgradient bore (Well 4228) and the downgradient bores (Wells 5662 and 4227), is characterised by generally low and stable concentrations across all indicators.

Observed temporal trends are weak, with only minor isolated fluctuations at all monitoring locations, and low R^2 values indicate that these changes do not reflect sustained directional trends. pH is mildly acidic across the network, with levels moving towards the aesthetic drinking-water guideline range. EC, chloride, and dissolved potassium remain low and consistent, with minor differences between bores likely reflecting local hydrogeological or catchment influences rather than landfill-derived impacts. TAN, nitrate-N, and nitrite-N are low, stable, and show no evidence of increasing concentrations over time. TOC and A254 values remain low across all wells, with slightly higher values at downgradient bores, but without sustained upward trends, indicating no persistent leachate influence.

Outcomes

This assessment is based on the preliminary risk matrix (Section 3.1) and the groundwater monitoring data analysis (Section 3.2.5). The Tikokino Closed Landfill contains historical municipal waste and lacks engineered containment systems that could pose a potential risk to groundwater quality. However, groundwater monitoring results from both downgradient bores (Wells 5662 and 4227) indicate that contaminant concentrations are generally low and stable over time. In December 2018, a modest increase in 6 parameters was detected in Well 4227, but as an isolated event which has not repeated.

All monitored parameters remain below site-specific threshold values, aesthetic drinking-water guideline values, and UPL(95%) baseline concentrations. Temporal trends are weak, with only isolated fluctuations observed, and no evidence of sustained directional changes or plume development is apparent. Differences between upgradient and downgradient wells are minor and appear to reflect local hydrogeological or broader catchment influences rather than landfill-derived leachate.

Overall, the Tikokino Closed Landfill demonstrates limited measurable environmental impact and is appropriately classified as low risk under current conditions. Routine groundwater monitoring remains appropriate. In the case where the consent holder applies to reduce monitoring activities, such an application would be considered, and the site may be eligible for a reduced monitoring frequency.

3.3.6 Ongaonga Closed Landfill

Source and containment assessment

The Ongaonga Closed Landfill formerly operated as a municipal waste disposal site serving the Ongaonga community. This site is an unlined landfill, with no documented soil permeability testing or engineered design for the landfill basin. Details regarding the depth and composition of the landfill capping are not recorded. Despite these limitations, downgradient groundwater shows no clear evidence of landfill influence: overall results of the monitoring programme assessment (Section 3.2.6) are summarised and interpreted in Table 28. This data shows that Ongaonga closed landfill is not (or no longer) acting as a significant source of contaminants to groundwater. The most likely reason for this lack of a source signature in groundwater is the age of this landfill, because containment is minimal.

Pathway and receptor assessment

Refer also to the preliminary risk matrix (Table 5) in Section 3.1. No upgradient monitoring bore is available at the Ongaonga site. Groundwater quality in the single downgradient monitoring bore is consistently good and stable. All monitored parameters remain at low concentrations, with no evidence of contaminant transport from the landfill. The nearest drinking water bore is located perpendicular to the groundwater flow direction, a considerable distance (3 km) away from the landfill, providing substantial attenuation between the source and potential sensitive receptors.

Monitoring data analysis

Table 28. Summary of monitoring evidence assessment for Ongaonga closed landfill

Parameters	Downgradient Well 5356	Temporal trend (Well 5356)	Comparison with Thresholds / UPL(95%)	Interpretation in relation to landfill influence
pH	Mean pH of 6.81 slightly below aesthetic range	No trend (two isolated spikes)	Mean pH below site-specific threshold and UPL(95%)	Mildly acidic but stable, the isolated spicks likely analytical or short-term environmental effects
EC	Mean EC concentrations remain low over time	Very weak upward trend over time	Mean EC well below both site-specific threshold and UPL(95%)	Stable low EC concentrations
Chloride	Mean chloride concentration is low	Very weak upward trend over time	Mean chloride well below aesthetic guideline and UPL(95%)	Chloride concentration remains far below nominated thresholds, no evidence of leachate influence
Dissolved Potassium	Mean potassium concentration is low	Very weak decline trend over time	Mean concentration well below site-specific threshold and UPL(95%)	Potassium concentrations and trends consistent over time and below nominated thresholds, no indicative of leachate migration
TAN	Mean concentration is 1.67 mg/L	No trend	Mean concentration well below site-specific threshold and UPL(95%)	TAN value slightly increases over time, but below the nominated thresholds
Nitrate-N	Mean concentration is 3.07 mg/L	Possible downward trend, not significant	Mean concentration well below drinking-water guideline and UPL(95%)	Nitrate concentrations fluctuate over time, but remain below nominated thresholds
Nitrite-N	Mean concentration is 0.01 mg/L	Weak upward trend with recent raise	Well below nominated threshold	Very low concentrations
TOC	Mean concentration is low	Possible upward trend, not significant	Mean concentration well below site-specific threshold and UPL(95%)	Stable low organic carbon; no indicative of leachate migration
A254	Very low absorbance values	No trend	Mean concentration well below site-specific threshold limit and UPL(95%)	Slightly decreasing concentrations, remain low

Based on the temporal variability observed across the monitoring record in Section 3.2 and the summary presented in Table 28, all monitored parameters at the Ongaonga Closed Landfill remain well below their respective site-specific threshold values, aesthetic drinking-water guideline values, and UPL(95%) baseline concentrations. Groundwater quality in the downgradient monitoring bore (Well 5356) is characterised by generally low and stable concentrations across all parameters.

Temporal trends are either absent or weak, with only minor isolated fluctuations throughout the monitoring period. pH values are mildly acidic and remain slightly below the aesthetic drinking-water guideline range; however, they are stable overall, with two isolated peaks more likely attributable to poor in-field sampling techniques, cross-contamination or short-term environmental effects rather than sustained changes in groundwater chemistry.

Outcomes

This assessment is based on the preliminary risk matrix (Section 3.1) and the groundwater monitoring data analysis (Section 3.2.6). The Ongaonga Closed Landfill contains historical municipal waste and lacks engineered containment systems that could pose a potential risk to groundwater quality. However, groundwater monitoring results from the downgradient bore (Well 5356) indicate that contaminant concentrations are consistently low and stable over time.

All monitored parameters remain below site-specific threshold values, aesthetic drinking-water guideline values, and UPL(95%) baseline concentrations. Temporal trends are weak, with only isolated fluctuations observed, and no evidence of sustained directional change, contaminant plume development, or progressive loading is apparent.

Overall, the Ongaonga Closed Landfill demonstrates limited measurable environmental impact and is appropriately classified as low risk under current conditions. Routine groundwater monitoring remains appropriate. In the case where the consent holder applies to reduce monitoring activities, such an application would be considered, and the site may be eligible for a reduced monitoring frequency.

3.3.7 Porangahau Closed Landfill

Source and containment assessment

The Porangahau Closed Landfill formerly operated as a municipal waste disposal site serving the Porangahau community. This site is an unlined landfill located at a coastal site, with no documented soil permeability assessment for the landfill basin, and no depth, composition and construction details of the landfill capping are available.

Despite this, downgradient groundwater monitoring does not indicate a clear landfill-related contaminant signature. The overall results of the monitoring programme assessment are summarised and interpreted in the relevant monitoring evidence tables (Section 3.2.7) are summarised and interpreted in Table 29. These findings indicate that the Porangahau Closed Landfill is not, or is no longer, acting as a significant source of contaminants to groundwater.

Pathway and receptor assessment

Refer also to the preliminary risk matrix (Table 5) in Section 3.1. The nearest downgradient bore is approximately 1 km away, and the Porangahau River is located less than 100 m from the site.

Based on the available site information, any potential leachate migration from the landfill would be subject to attenuation through natural processes, including soil adsorption, dilution within the aquifer, and degradation of organic contaminants during subsurface transport. In combination with the long groundwater travel distances and the absence of identified sensitive receptors along the flow path, these factors are likely to have limited the extent of any historic contaminant migration from the landfill.

Monitoring data analysis

Table 29. Summary of monitoring evidence assessment for Porangahau closed landfill

Parameters	Downgradient Well 5661	Temporal trend (Well 5661)	Comparison with Thresholds / UPL(95%)	Interpretation in relation to landfill influence
pH	Mean pH of 6.81 slightly below aesthetic range	Upward trend with recent raise	Mean pH below site-specific threshold and UPL(95%)	Mildly acidic but stable, slight increase towards to aesthetic range
EC	Mean EC is higher than usual, due to the coastal location	Apparent weak declining trend (significance influenced by one isolated spike)	Mean EC well below both site-specific threshold and UPL(95%)	Stable low EC concentrations
Chloride	As for EC, mean chloride concentration higher than usual, due to the coastal location	Weak upward trend, not significant	Mean chloride well below aesthetic guideline and UPL(95%)	Chloride concentration remains far below nominated thresholds, no evidence of leachate influence
Dissolved Potassium	Mean potassium concentration is low	No trend	Mean concentration well below site-specific threshold and UPL(95%)	Potassium concentrations and trends consistent over time and below nominated thresholds, no indicative of leachate migration
TAN	Mean concentration is 0.18 mg/L, three times higher than nitrate-N	Weak upward trend over time	Mean concentration well below site-specific threshold	TAN value being higher than nitrate is consistent with chemistry of natural organic matter degradation in coastal areas. Slight increase over time, but below the nominated threshold.
Nitrate-N	Mean concentration is 0.06 mg/L	No trend	Mean concentration well below drinking-water guideline and UPL(95%)	Nitrate concentrations generally steady over time, but remain below nominated thresholds
Nitrite-N	Near or lower than detectable level	No trend	Well below nominated threshold	Extremely low concentrations; no evidence of contamination
TOC	Mean concentration is low	No trend	Mean concentration fluctuates over time, well below site-specific threshold and UPL(95%)	Stable low organic carbon; no indicative of leachate migration
A254	Very low absorbance values	No trend	Mean concentration well below site-specific threshold limit and UPL(95%)	Low UV absorbance; no evidence of aromatic organic enrichment

Based on the temporal variability observed across the monitoring record in Section 3.2 and the summary presented in Table 29, all monitored parameters at the Porangahau Closed Landfill remain well below their respective site-specific threshold values, aesthetic drinking-water guideline values, and UPL(95%) baseline concentrations. Groundwater quality in the downgradient monitoring bore (Well 5661) is characterised by generally low and stable concentrations across all parameters. This well consistently shows higher EC and chloride than the other landfills, and higher ammoniacal-N than nitrate; these are all linked with its coastal location.

Observed temporal trends are weak, with only minor isolated fluctuations evident in the monitoring record, and do not represent sustained or systematic directional changes over time. There is no evidence of solute enrichment attributable to landfill leachate for this site.

Outcome

This assessment is based on the preliminary risk matrix (Section 3.1) and the groundwater monitoring data analysis (Section 3.2.7). The Porangahau Closed Landfill contains historical municipal waste and lacks engineered containment systems that could pose a potential risk to groundwater quality. However, groundwater monitoring results from the downgradient bore (Well 5661) indicate that contaminant concentrations are consistently low and stable over time.

All monitored parameters remain below site-specific threshold values, aesthetic drinking-water guideline values, and UPL(95%) baseline concentrations. Temporal trends are weak and largely influenced by isolated fluctuations, with no evidence of sustained directional change, contaminant plume development, or progressive contaminant loading.

Overall, the Porangahau Closed Landfill demonstrates limited measurable environmental impact and is appropriately classified as low risk under current conditions. Routine groundwater monitoring remains appropriate. In the case where the consent holder applies to reduce monitoring activities, such an application would be considered, and the site may be eligible for a reduced monitoring frequency.

3.4 Synthesis across all landfills

The seven closed landfills share common historical characteristics typical of small municipal landfills developed prior to modern regulatory standards, including the absence of engineered liners, leachate collection systems, and low-permeability caps in the Central Hawke's Bay District, New Zealand. Their key similarities and differences are summarised as follows:

Source and containment

All seven landfills contain residual municipal waste and therefore retain the theoretical potential to generate leachate. However, given the age of the sites (generally closed for more than 30 years), waste decomposition is expected to be in late methanogenic or stabilisation phases. This is supported by the generally low concentrations of organic carbon, nitrogen species attributable to leachate, and conservative ions observed across monitoring records. In the absence of engineered containment, risk mitigation relies primarily on natural attenuation processes and site-specific geological and hydrological conditions.

Pathway and receptor

At most landfills, potential groundwater transport pathways are weak due to a combination of subsurface adsorption and degradation chemistry, long contaminant travel distances, unfavourable hydraulic gradients, limited hydraulic connectivity to sensitive receptors, and dilution effects associated with nearby rivers or drainage networks. Sites including Waipukurau, Takapau, Tikokino, Ongaonga, and Porangahau are characterised by either long downgradient flow paths (>1 km), the upgradient location of drinking-water bores, or groundwater flow directions that divert potential leachate migration away from receptors. Collectively, these factors substantially reduce the likelihood of meaningful contaminant transport, even in the absence of engineered containment systems. Tamumu represents the shortest and most direct potential pathway scenario, with both downgradient groundwater and surface-water receptors located within approximately 500 m of the landfill footprint. While the Tukituki River provides some degree of hydraulic buffering and dilution, the reduced travel distance increases the theoretical exposure potential and distinguishes Tamumu from the other landfills assessed.

Receptor assessment further supports the overall low-risk classification across the study area. With the exception of Tamumu, no drinking-water supplies are located downgradient within 1 km of any landfill. Although surface-water bodies are commonly situated within 100 m of the sites, long-term monitoring data show no direct evidence of sustained impacts attributable to landfill-derived contamination. Overall, receptor sensitivity across the study area is assessed as low to very low.

Cross-site implications

Taken together with the monitoring data analysis in Section 3.2 and framework application in Section 3.3, the synthesis confirms that legacy unlined landfills do not necessarily pose significant long-term groundwater risks when favourable hydrogeological conditions and receptor separation are present. The consistent Low-Risk outcomes across most sites demonstrate the effectiveness of the applied framework in integrating multiple lines of evidence into transparent and defensible risk classifications. Importantly, the framework also differentiates sites where continued monitoring remains necessary, ensuring that management effort is proportionate to actual environmental risk rather than uniformly applied.

This cross-site assessment provides an evidence basis for rationalising groundwater monitoring programmes across the district, while maintaining protection of sensitive receptors and compliance with regulatory expectations. It also illustrates the value of structured, evidence-based decision tools for managing large portfolios of legacy landfills under constrained regulatory and financial resources. The outcomes of the cross-site implications are summarised in Table 30.

Table 30. Cross-site implications using the risk matrix

Landfill	Source Activity	Pathway Strength & Receptor Sensitivity	Monitoring Evidence	Overall Risk
Waipawa	Aged municipal waste; stabilised	Very low, long groundwater pathway (2.3 km)	Trends stable or weakly increasing; concentrations remain low, except TAN exceedance the threshold	**Low**
Waipukurau	Aged municipal waste; stabilised	Very low, long groundwater pathway (3.3 km)	No significant trends or threshold exceedances	**Low**
Takapau	Aged municipal waste; stabilised	Very low, long downgradient pathway (3.4km)	Stable groundwater quality; no downgradient enrichment	**Low**
Tamumu	Aged municipal waste; stabilised	Moderate, short pathway distance (<500 m)	Trends stable or weakly increasing; concentrations remain low, except TAN exceedance the threshold	**Medium**
Tikokino	Aged municipal waste; stabilised	Low, long downgradient pathway (2.2 km)	No significant trends or threshold exceedances	**Low**
Ongaonga	Aged municipal waste; stabilised	Low, relatively long downgradient pathway (>500 m)	Stable groundwater chemistry; no evidence of leachate impact	**Low**
Porangahau	Aged municipal waste; stabilised	Low, long downgradient pathway (1 km)	No significant trends or threshold exceedances	**Low**

3.5 Key uncertainties and limitations

The application of the risk-based decision-making framework provides a structured and evidence-informed assessment of environmental risk across the seven closed landfill sites. Key uncertainties and limitations should be acknowledged. These relate primarily to data availability, the reliability and consistency of analytical chemistry test data, hydrogeological complexity, monitoring network design, and the inherent constraints of retrospective assessment of legacy sites.

The isolated concentration "spikes" may represent either genuine short-term environmental variability or analytical artefacts; conversely, some transient elevated values may have gone undetected. In the absence of sufficient replicate data or the ability to request repeat analyses for historical samples, individual anomalous results cannot be conclusively accepted or rejected. Accordingly, all reported results have been retained within the dataset and interpreted in the context of long-term trends, spatial coherence, and consistency across parameters, rather than on the basis of single data points. Importantly, these uncertainties do not undermine the overall conclusions of low to medium risk but define the confidence bounds within which the results should be interpreted and reinforce the value of a weight-of-evidence approach that integrates trend analysis, hydrogeological context, and receptor sensitivity.

Monitoring network limitations

A key limitation identified across several sites is the constrained groundwater monitoring network, particularly the reliance on a single downgradient monitoring bore at some locations. Although bore placement is generally consistent with inferred groundwater flow directions, the absence of multiple downgradient or cross-gradient wells limits the ability to fully characterise plume geometry, lateral dispersion, and spatial variability. This limitation is most evident at sites where subtle chemical signals are observed, such as Waipawa and Tamumu, where additional monitoring points (both upgradient and downgradient) would improve confidence in delineating plume extent and attenuation behaviour.

In addition, where no dedicated upgradient well is available, the UPL (95%) is derived from the historical data of the downgradient well itself. This approach assumes that earlier monitoring results reflect background conditions, an assumption that may not hold if the well has experienced low-level impacts since monitoring commenced.

Furthermore, historical changes to monitoring infrastructure, including the loss or discontinuation of the monitoring of wells, have reduced spatial continuity within the long-term dataset. As a result, the assessment relies on the assumption that the remaining monitoring bores are representative of downgradient conditions. While this assumption is reasonable in the context of the available hydrogeological information, it cannot be independently verified in all cases.

Hydrogeological interpretation uncertainty

Groundwater flow directions and hydraulic connectivity were inferred from Central Hawke's Bay District Council records or limited site data rather than detailed tracer studies or a dense network of piezometers. At sites like Waipawa, where the downgradient well shows lower contaminant levels than the upgradient well, the assumed flow path could be incorrect, or complex local hydrogeology (e.g., preferential flow, groundwater mounding) may be influencing results in ways not captured.

3.6 Potential for further improvements

Although the current application of the risk-based decision-making framework provides an assessment of environmental risk across the seven closed landfills, several opportunities exist to further refine monitoring and management approaches. These improvements aim to reduce residual uncertainties, enhance the precision of risk evaluation, and support adaptive, evidence-based decision-making.

Optimisation of monitoring networks

Expanding the groundwater monitoring network at select sites would increase confidence in plume delineation and trend assessment. Installing additional downgradient and cross-gradient wells would better capture potential lateral variability and provide earlier detection of emerging contaminant signals. Network optimisation should prioritise areas with shorter flow paths, higher receptor sensitivity, or observed low-level chemical signals, while maintaining cost-effectiveness and proportionality at sites with demonstrably low risk (Gladish , Pagendam , Janardhanan , & Gonzalez, 2023) (Hudak, 2001).

Data integration and predictive modelling

Integrating historical monitoring data into predictive models could improve long-term risk forecasts, optimise monitoring frequency, and inform prioritisation of resource allocation. By coupling empirical data with site-specific hydrogeological parameters, predictive models could simulate contaminant attenuation, estimate potential exposure scenarios, and evaluate the effects of extreme events, further enhancing regulatory confidence (Thakur, 2017).

Benchmarking and validation

Applying the framework to a wider set of landfills across New Zealand, including sites with known, active impacts (high-risk), would validate its ability to correctly discriminate across the full risk spectrum. This would build a national evidence base and increase regulatory confidence in its outcomes.

Development of standardised protocols and guidance

Translating the academic framework into a formal Council Guideline or Technical Standard would ensure consistent implementation by consultants and council staff, complete with standardized reporting templates and decision logs.

4. Conclusion

This study has applied a structured, risk-based framework to assess the environmental risk posed by seven closed, unlined municipal landfills in the Central Hawke's Bay District and has developed and applied a standardised, risk-based framework to evaluate the justification for their current groundwater monitoring regimes. By integrating hydrogeological setting, long-term monitoring data, and the S-P-R model, this leads to the following key conclusions in direct response to the research questions set in Section 1.5.

Current level of environmental risk

The comprehensive assessment indicates that all seven closed landfills (Waipawa, Waipukurau, Takapau, Tikokino, Ongaonga, and Porangahau) present a generally low environmental risk to groundwater resources. Only Tamumu closed landfill appeared medium risk to downgradient groundwater quality. Monitoring results show that contaminant concentrations are predominantly stable or declining, consistent with landfills that have progressed into late methanogenic or maturation phases. Key leachate indicators, including chloride, potassium, ammoniacal nitrogen, and organic parameters (TOC, VFA, UV254), are typically at or near background concentrations, with no evidence of widespread or increasing contaminant plumes. Isolated parameter exceedances or weak temporal trends, such as minor increases in EC or chloride at some sites, do not indicate a systemic or significant risk. These patterns are more plausibly explained by local hydrogeological variability, catchment-scale influences, or analytical variability rather than ongoing landfill-derived impacts. In addition, the generally favourable hydrogeological settings of many sites, including river-adjacent locations that promote dilution and groundwater flow paths directed away from sensitive receptors, further mitigate potential risks. No site showed evidence of impacts on nearby drinking water supplies that could be conclusively attributed to landfill leachate.

Efficacy of the standardised risk assessment framework

A standardised and defensible decision-making framework was successfully developed and applied. The framework integrates the S–P–R model with a semi-quantitative, multi-evidence assessment of site conditions, monitoring data (including statistical exceedance analysis and trend evaluation), and receptor sensitivity. Its application demonstrated effectiveness in standardising assessments across heterogeneous legacy sites, synthesising datasets into transparent and repeatable risk classifications (Low, Medium, High), and supporting proportionate resource allocation by distinguishing sites requiring continued vigilance from those where monitoring rationalisation may be scientifically and legally justified.

The framework therefore, provides a practical and defensible tool for environmental managers and consent authorities making long-term decisions regarding the monitoring of closed landfills.

Recommendations for monitoring rationalisation

Based on the application of the framework, Tamumu is classified as a Medium Risk site, while the remaining six sites are classified as Low Risk. This classification reflects the convergence of evidence across source strength, pathway characteristics, receptor sensitivity, and groundwater quality trends.

Continued monitoring is recommended for all sites; however, the frequency and scope of monitoring can be reduced for most locations. For Waipukurau, Takapau, Tikokino, Ongaonga, and Porangahau, the consistently stable groundwater conditions support a transition from quarterly to biannual or annual monitoring of core parameters (e.g. pH, nitrate, chloride, TAN). This approach would maintain protective oversight while optimising the use of monitoring resources. The same conditions could apply to Waipawa, with the proviso that a benefit of continuing the same frequency of monitoring well would be to further characterise development of an apparent new adjacent source, which is (probably) unrelated to the landfill or its resource consent, but could involve the same consent-holder.

Enhanced vigilance is also recommended for sites where weak upward trends or greater receptor sensitivity have been identified. At Waipukurau, recent increases in EC, TAN, and chloride, while remaining below assessment thresholds, justify continued quarterly monitoring in the short term to determine whether these trends persist or attenuate. At Tamumu, the exceedance of the TAN aesthetic guideline (although remaining below the statistical UPL), combined with the presence of a relatively close downgradient receptor, supports retention of quarterly monitoring to confirm the ongoing decline in TAN and the stability of other parameters.

Any decision to formally reduce monitoring requirements should be documented through an application process that references this risk assessment. A periodic review cycle (e.g. every five years) is recommended to reassess site conditions using the framework and to ensure continued protection against unforeseen changes.

This research confirms that the assessed legacy landfills in Central Hawke's Bay are in a state of advanced stabilisation and present low, well-managed environmental risks. There are some differences between landfills – for example, groundwater chemistry at Porangahau Closed Landfill has higher EC and chloride, reflecting its coastal location. However, the developed risk-based framework can accommodate these and translates complex environmental data into clear and defensible decision-making pathways, addressing a critical need for consistent long-term management of historic landfill sites. By enabling risk-proportionate monitoring, it supports ongoing protection of groundwater and downgradient receptors while allowing for more efficient allocation of regulatory and operational resources.

5. Appendices

Appendix 5.1 Property information

	Site Address	Site Area	Legal Description
Ongaonga Closed Landfill	Blackburn Road, Ongaonga	0.7 ha	Sec 4 Blk XI Ruataniwha SD
Tamumu Closed Landfill	River Road, Tamumu	2.0 ha	Lt 1 DP 12148
Waipukurau Closed Landfill	Mt Herbert Rd, Waipukurau	1.0 ha	Lt 1 DP 9735
Waipawa Closed Landfill	Tikokino Road, Waipawa	1.4 ha	Lt 1 DP 16272 and Pt DP 2404
Porangahau Closed Landfill	Keppel Street, Porangahau	0.6 ha	Sec 7 Blk XII Porangahau SD
Tikokino Closed Landfill	Holden Road, Tikokino	0.3 ha	Sec 1 SO 2365
Takapau Closed Landfill	Paulsen Road, Takapau	0.7 ha	Otawhao A3 Sec 49A No.2 Blk

Appendix 5.2 Location of the nearest registered drinking water supplies

Landfill	Nearest drinking water supply
Waipawa	CHBDC Johnson Street Bore (~ 2.3 km)
Waipukurau	Farm Road Bore (< 1,000 m)
Takapau	CHBDC Meta Street Bore (~ 1 km)
Tamumu	Hautope Bore (< 500 m)
Tikokino	Tikokino School Bore (~ 2.3 km)
Ongaonga	Ongaonga School Bore (~ 3 km)
Porangahau	CHBDC Porangahau Bore (~ 4 km)
Kairakau	No downstream drinking water supplies

Appendix 5.3 Closed landfill site descriptions

Site name	Site address	Legal description	District Plan Zoning	Current sit layout and topography	Adjacent site receptors	Geology and hydrogeological sitting
Waipawa Closed Landfill	Tikokino Road, Waipawa	Lt 1 DP 16272 and Pt DP 2404	Residential	The site is a mounded area located on a river terrace. The site is bounded by Tikokino Road to the north, and by the Waipawa River to the south. Runoff from the site will flow towards the roadside drain, and towards the river. The site is largely not utilised with the exception of the eastern end of the site which is used as a refuse transfer station.	The site is located on the edge of Waipawa township, with recreational areas to the north and east. The Waipawa River is located on the southern boundary of the site.	Recent alluvial deposits including gravel, sand, silt and mud forming alluvial terraces.
Waipukurau Closed Landfill	Mt Herbert Rd, Waipukurau	Lt 1 DP 9735	Rural	This site is a mounded site located on a river terrace. The site is bounded by a drain on the southern site, and by a public walkway to the north. Runoff from the site will discharge towards the drain, or flow overland towards the Tukituki River. The site is currently being used to store bags of sludge from the wastewater treatment plant but is otherwise not utilised.	The site is located on the edge of Waipukurau township. The wastewater treatment plant and transfer station is located to the west of the site, and rural residential development is occurring across Mt Herbert Road to the south and east of the site. The Tukituki River is located approximately 85 m to the north of the site.	Recent alluvial deposits including gravel, sand, silt and mud forming alluvial terraces.
Onga Onga Closed Landfill	Blackburn Road, Onga Onga	Sec 4 Blk XI Ruataniwha SD	Rural	Flat site with incised channel on southern and western sides. A drain is present along the western edge of the site. Mounds of cleanfill and concrete rubble present at the site have been spread across the site to improve the cover thickness. Site currently not utilised.	Located in a rural setting. Nearest surface water body is a small ephemeral stream located 600 m away.	Recent alluvial deposits including gravel, sand, silt and mud forming alluvial terraces.
Porangahau Closed Landfill	Keppel Street, Porangahau	Sec 7 Blk XII Porangahau SD	Rural	Flat site located on a river terrace. Additional cleanfill material was added to the site to improve the cover thickness in 2015. Site currently not utilised.	Located in a rural setting adjacent to Porangahau township. Poranagahau River is located 50m from the site. The river is tidal in the vicinity of the site, and there is a wastewater treatment plant outfall located 1.7 km downstream of the landfill.	Recent alluvial deposits including gravel, sand, silt and mud forming alluvial terraces.
Tikokino Closed Landfill	Holden Road, Tikokino	Sec 1 SO 2365	Rural	Very small site comprising a flat area level with the adjacent road, and a step bank leading down to a low level river terrace. The site is currently not utilised.	The Mangaoho River is located approximately 30 m southwest of the site. The river is ephemeral in this reach, and has been highly modified by agricultural activities within the catchment.	Recent alluvial deposits including gravel, sand, silt and mud forming alluvial terraces.

Tamamu Closed Landfill	River Road, Tamamu	Lt 1 DP 12148	Rural	The site is a mounded site located on a flat river terrace. The site is bounded by a small ephemeral stream along the northern edge, and the Tukituki River on the eastern boundary. Surface runoff will main drain towards these two water bodies. The site is currently used for grazing.	The site is located in a rural setting. The Waipawa River is located on the eastern boundary of the site.	Recent alluvial deposits including gravel, sand, silt and mud forming alluvial terraces.
Takapau Closed Landfill	Paulsen Road, Takapau	Otawhao A3 Sec 49A No.2 Blk	Rural	The site comprises a flat area in the south, which slopes down towards a stream on the northern edge of the site. The site is currently utilised as a refuse transfer station.	The site is located within a rural area and is surrounded by pastoral land use. A small stream is located on the northern boundary of the site. The stream is heavily modified by the surrounding agricultural use. It flows into the Porangahau Stream approximately 1 km downstream of the site.	Recent alluvial deposits including gravel, sand, silt and mud forming alluvial terraces.
Kairakau Closed Landfil	Kairakau Road, Kairakau	Kairakau 1A Blk	Rural	This site is small mounded site located at the base of a valley. The site is located near the Ronui Stream which drains into the Mangakuri River 80 downstream of site. The site is currently used for grazing purposes.	The site is located in a rural area. The Ponui Stream, a tributary of the Mangakuri River, is located approximately 30 m to the south of the site. The stream and river are tidal.	Kairakau Limestone, yellow grey, barnacle rich, cross-bedded sandy limestone.

References

Abiriga, D. V. (2020). Groundwater contamination from a municipal landfill: Effect of age, landfill closure, and season on groundwater chemistry. *Science of The Total Environment, 737*. doi:https://doi.org/10.1016/j.scitotenv.2020.140307

Aleksandra, W. A. (2020). How to Choose Pollution Indicators for Monitoring Landfill Leachates. *Proceedings, 51*. doi:https://doi.org/10.3390/proceedings2020051023

Anne, J. L., & Lee, G. F. (1993). *GROUNDWATER POLLUTION BY MUNICIPAL LANDFILLS: LEACHATE COMPOSITION, DETECTION AND WATER QUALITY SIGNIFICANCE*. Retrieved 4 3, 2025, from G.Fred Lee & Associates EnviroQual: https://www.gfredlee.com/lf-conta.htm

Bilgili, M. S. (2012). Effect of leachate recirculation and aeration on volatile fatty acid concentrations in aerobic and anaerobic landfill leachate. *Waste Management & Research, 30*(2), 161-170. doi:DOI: 10.1177/0734242X11417983

Bove, D. M. (2015). A Critical Review of Biological Processes and Technologies for Landfill Leachate Treatment. *Chemical Engineering & Technology, 38*(12), 2115-2126. doi:https://doi-org.ezproxy.massey.ac.nz/10.1002/ceat.201500257

Brand, J. H., Spencer, K. L., O'shea, F. T., & Lindsay, J. E. (2017). Potential pollution risks of historic landfills on low-lying coasts and estuaries. *5*(1). doi:https://doi.org/10.1002/wat2.1264

Christensen, H. T.-J. (2001). Biogeochemistry of landfill leachate plumes. *Applied Geochemistry, 16*(7-8), 659-718. doi:https://doi.org/10.1016/S0883-2927(00)00082-2

El-Fadel, M. A. (1997). Environmental Impacts of Solid Waste Landfilling. *Journal of Environmental Management, 50*(1), 1-25. doi:https://doi.org/10.1006/jema.1995.0131

Environment Agency. (2004). *Guidance on monitoring of landfill leachate, groundwater and surface water (LFTGN02).* Retrieved from Environment Agency: https://assets.publishing.service.gov.uk/media/5a7ece0840f0b62305b8349c/LFTGN02.pdf

Environment Agency. (2025). *Land contamination risk management (LCRM).* Retrieved 7 23, 2025, from GOV.UK: https://www.gov.uk/government/publications/land-contamination-risk-management-lcrm/lcrm-stage-1-risk-assessment

EPA. (2013). *Statistical Software for Environmental Applications for Data Sets with and without Nondetect Observations.* Retrieved 3 12, 2025, from EPA:

https://www.epa.gov/sites/default/files/2015-03/documents/proucl_v5.0_tech.pdf

Farquhar, G. (1989). Leachate: production and characterization. *Canadian Journal of Civil Engineering, 16*(3), 317-325. doi:https://doi.org/10.1139/l89-057

Farquhar, G. (1989). Leachate: production and characterization. *Canadian Journal of Civil Engineering, 16*(3), 317-325. doi:https://doi.org/10.1139/l89-057

Farquhar, G. J. (1989). Leachate: Production and characterization. *Canadian Journal of Civil Engineering, 16*(3), 317-325. doi:https://doi.org/10.1139/l89-057

Gladish , D., Pagendam , D., Janardhanan , S., & Gonzalez, D. (2023). Geostatistical based optimization of groundwater monitoring well network design. *Frontiers in Earth Science, 11*. doi:https://doi.org/10.3389/feart.2023.1188316

Gove, A., Oates, J., & Archibald, R. (2015, 10 29). *Control charts as an environmental monitoring tool for decision making*. Retrieved from Environmental Institute of Australia and New Zealand.

Helsel, D. R. (2020, 5 22). *Statistical Methods in Water Resources.* doi:https://doi.org/10.3133/tm4A3

Huang, Z. L. (2024). Assessing the impacts and contamination potentials of landfill leachate on adjacent groundwater systems. *Science of The Total Environment, 930*. doi:https://doi.org/10.1016/j.scitotenv.2024.172664

Huang, Z., Liu, G., Zhang, Y., Yuan, Y., Xi, B., & Tan, W. (2024). Assessing the impacts and contamination potentials of landfill leachate on adjacent groundwater systems. *Science of The Total Environment, 930*. doi:https://doi.org/10.1016/j.scitotenv.2024.172664

Hudak, F. (2001). Effective contaminant detection networks in uncertain groundwater flow fields. *Waste Management, 21*(4), 309-312. doi:https://doi.org/10.1016/S0956-053X(00)00079-9

Jamrah, A. A.-Z.-Q. (2024). An Extensive Analysis of Combined Processes for Landfill. *Water, 16*(12). doi:https://doi.org/10.3390/w16121640

Juliao, C., Díaz, J., Bermúdez, Y., & Aldana, M. (2023). Integration of geoelectric and geochemical data using Self-Organizing Maps to characterize a landfill. doi:https://doi.org/10.48550/arXiv.2309.09164

Kang, K. H. (2002). Characterization of humic substances present in landfill leachates with different landfill ages and its implications. *Water Research, 36*(16), 4023-4032. doi:https://doi.org/10.1016/S0043-1354(02)00114-8

Kjeldsen, P. B. (2002). Present and Long-Term Composition of MSW Landfill Leachate: A Review. *Critical Reviews in Environmental Science and Technology*, 297-336. doi:https://doi.org/10.1080/10643380290813462

Koda, E. O. (2015). Areal Distribution of Ammonium Contamination of Soil-Water Environment in the Vicinity of Old Municipal Landfill Site with Vertical Barrier. *Water, 7*(6), 2656-2672. doi:https://doi.org/10.3390/w7062656

Kulikowska, D., & Klimiuk, E. (2008). The effect of landfill age on municipal leachate composition. *Bioresource Technology, 99*(13), 5981-5985. doi:https://doi.org/10.1016/j.biortech.2007.10.015

Lorah, M., Cozzarelli, I., & Böhlke, J. (2009). Biogeochemistry at a wetland sediment-alluvial aquifer interface in a landfill leachate plume. *Journal of Contaminant Hydrology, 105*(3-4), 99-117. doi:10.1016/j.jconhyd.2008.11.008

Magda, M. A.-S.-Z. (2015). Impact of landfill leachate on the groundwater quality: A case study in Egypt,. *Journal of Advanced Research,, 6*(4), 579-586. doi:https://doi.org/10.1016/j.jare.2014.02.003

MfE. (2001). *A guide to the management of closing and closed landfills in New Zealand*. Retrieved from Ministry for the Environment: https://environment.govt.nz/assets/Publications/Files/closed-landfills-guide-may01_0.pdf

MfE. (2020). *Small landfill closure criteria: Risk assessment for small closed landfills* . Retrieved from Ministry for the Environment: https://environment.govt.nz/publications/small-landfill-closure-criteria-risk-assessment-for-small-closed-landfills/

MfE. (2025, 12 9). *Government unveils major overhaul of New Zealand's planning system*. Retrieved from Ministry of Environment: https://environment.govt.nz/news/government-unveils-major-overhaul-of-new-zealands-planning-system/

MfE. (2025, 12 9). *New Planning System: Transitioning into the new planning system*. Retrieved from Ministry of Environment: https://environment.govt.nz/publications/transitioning-into-the-new-planning-system/

MfE. (2023). *Technical Guidelines for Disposal to Land: Revision 3.1*. Retrieved from Ministry for the Environment: https://environment.govt.nz/assets/publications/Waste/Version-3.1-TG-for-Disposal-to-Land-September-2023.pdf

Milosevic, N., Thomsen, N. I., Juhler, R. K., Albrechtsen, H. J., & Bjerg, P. L. (2012). Identification of discharge zones and quantification of contaminant mass discharges into a local stream from a landfill in a heterogeneous geologic setting. *Journal of Hydrology, 446-447*, 13-23. doi:https://doi.org/10.1016/j.jhydrol.2012.04.012

Mor, S. R. (2006). Leachate characterization and assessment of groundwater pollution near municipal solid waste landfill site. *Environmental Monitoring and Assessment, 118*(1-3), 435-456. doi:DOI: 10.1007/s10661-006-1505-7

Mukherjee, S. M. (2015). Contemporary Environmental Issues of Landfill Leachate: Assessment and Remedies. *Critical Reviews in Environmental Science and Technology, 45*(5), 472-590. doi:https://doi.org/10.1080/10643389.2013.876524

Nanda, S. B. (2021). Municipal solid waste management and landfilling technologies: a review. *Environ Chem Lett, 19*, 1433-1456. doi:https://doi.org/10.1007/s10311-020-01100-y

Negi, P. M. (2020). Impact of landfill leachate on the groundwater quality in three cities of North India and health risk assessment. *Environment, Development and Sustainability , 22*, 1455–1474. doi:https://doi.org/10.1007/s10668-018-0257-1

Neill, P., & Megson, D. (2024). Landfill leachate treatment process is transforming and releasing banned per- and polyfluoroalkyl substances to UK water. *Frontiers in Water*. doi:https://doi.org/10.3389/frwa.2024.1480241

NES-CS. (2011). *Resource Management (National Environmental Standard for Assessing and Managing Contaminants in Soil to Protect Human Health) Regulations 2011*. Retrieved from New Zealand Legislation: https://www.legislation.govt.nz/regulation/public/2011/0361/latest/dlm4052228.html

NES-FW. (2020). *Resource Management (National Environmental Standards for Freshwater) Regulations 2020*. Retrieved from New Zealand Legislation: https://www.legislation.govt.nz/regulation/public/2020/0174/latest/LMS364099.html

NPS-FM. (2020). *The National Policy Statement for Freshwater Management*. Retrieved from MfE: https://environment.govt.nz/acts-and-regulations/national-policy-statements/national-policy-statement-freshwater-management/

Oyebamiji, A. H. (2024). Regional Assessment of Groundwater Contamination Risk from Crude Oil Spillages in the Niger Delta: A Novel Application of the Source-Pathway-Receptor Model. *Earth Systems and Environment, 9*, 117-133. doi:https://doi.org/10.1007/s41748-024-00416-x

Przydatek, G. (2021). Using advanced statistical tools to assess the impact of a small municipal landfill on the aquatic environment. *Environmental Monitoring and Assessment, 193*(2). doi:https://doi.org/10.1007/s10661-021-08850-4

Przydatek, G., & Kanownik, W. (2019). Impact of small municipal solid waste landfill on groundwater quality. *Environmental Monitoring and Assessment, 191*. doi:https://doi.org/10.1007/s10661-019-7279-5

Przydatek, G., & Kanownik, W. (2021, 06 24). Physicochemical indicators of the influence of a lined municipal landfill on groundwater quality: a case study from Poland. *Environmental Earth Sciences , 80*. doi:https://doi.org/10.1007/s12665-021-09743-y

Przydatek, G., & Kanownik, W. (2021). Physicochemical indicators of the influence of a lined municipal landfill on groundwater quality: A case study from Poland. *Environmental Earth Sciences*. doi:https://doi.org/10.1007/s12665-021-09743-y

Raco, B. D. (2010). GEOCHEMICAL INVESTIGATION OF AQUIFER POLLUTION FROMWASTE MANAGEMENT.THE CASE OF KOMOTINI LANDFILL (GREECE). *Bulletin of the Geological Society of Greece, 43*(4), 1840-1849. doi:https://doi.org/10.12681/bgsg.11375

Renou, S. G. (2008). Landfill leachate treatment: Review and opportunity. *Journal of Hazardous Materials, 150*(3), 468-493. doi:https://doi-org.ezproxy.massey.ac.nz/10.1016/j.jhazmat.2007.09.077

RMA. (1991). *Resource Management Act 1991*. Retrieved from New Zeland Legislation: https://www.legislation.govt.nz/act/public/1991/0069/latest/DLM230265.html

RRMP. (2006). *Regional Resource Management Plan*. Retrieved from Hawkes Bay Regional Council: https://www.hbrc.govt.nz/our-documents/rrmp/

Siddiqua et al. (2022). An overview of the environmental pollution and health effects associated with waste landfilling and open dumping. *Environmental Science and Pollution Research, 29*, 58514–58536. doi:https://doi.org/10.1007/s11356-022-21578-z

Siddiqua, A. H.-A. (2022). An overview of the environmental pollution and health effects associated with waste landfilling and open dumping. *Environmental Science and Pollution Research, 29*, 58514-58536. doi:https://doi.org/10.1007/s11356-022-21578-z

Slack, R. J. (2005). Household hazardous waste in municipal landfills: contaminants in leachate. *Science of The Total Environment, 337*(1-3), 119-137. doi:https://doi.org/10.1016/j.scitotenv.2004.07.002

Slack, R. J. (2005). Household hazardous waste in municipal landfills: contaminants in leachate. *Science of The Total Environment, 337*(1-3), 119-137. doi:https://doi.org/10.1016/j.scitotenv.2004.07.002

Slack, R. j., Gronow, J., & Voulvoulis, N. (2005). Household hazardous waste in municipal landfills: contaminants in leachate. *Science of The Total Environment, 337*(1-3), 119-137. doi:https://doi.org/10.1016/j.scitotenv.2004.07.002

Slack, R., Gronow, J., & Voulvoulis, N. (2005). Household hazardous waste in municipal landfills: contaminants in leachate. *Science of The Total Environment, 337*(1-3), 119-137. doi:https://doi.org/10.1016/j.scitotenv.2004.07.002

Sundaram, B., Wallace, L., Feitz, A. J., & Caritat, P. d. (2009). *Groundwater Sampling and Analysis: A Field Guide - 2nd Edition*. doi:DOI:10.26186/149678

SWANA. (1991). *Guidelines: Environmental Monitoring at Municipal Solid Waste Landfills.* Retrieved from SCRIBD: https://www.scribd.com/document/513947884/guidelines-environmental-monitoring-municipal-solid-waste-landfills

Thakur, J. (2017). Hydrogeological modeling for improving groundwater monitoring network and strategies. *Applied Water Science , 7*, 3223-3240. doi:https://doi.org/10.1007/s13201-016-0469-1

Uisung Lee, J. H. (2017). Evaluation of landfill gas emissions from municipal solid waste landfills for the life-cycle analysis of waste-to-energy pathways. *Journal of Cleaner Production, 166*, 335-342. doi:https://doi.org/10.1016/j.jclepro.2017.08.016

US EPA. (2009). *National Service Center for Environmental Publications (NSCEP)*. Retrieved from US EPA: https://nepis.epa.gov/Exe/ZyNET.exe/P10055GQ.txt?ZyActionD=ZyDocument&Client=EPA&Index=2006%20Thru%202010&Docs=&Query=&Time=&EndTime=&SearchMethod=1&TocRestrict=n&Toc=&TocEntry=&QField=&QFieldYear=&QFieldMonth=&QFieldDay=&UseQField=&IntQFieldOp=0&ExtQFiel

Vaverková, M. (2019). Landfill Impacts on the Environment - Review. *Geosciences, 9*(10). doi:https://doi.org/10.3390/geosciences9100431

Vaverková, M. (2019). Landfill Impacts on the Environment-Review. *Geosciences, 9*(10). doi:https://doi.org/10.3390/geosciences9100431

WasteMINZ. (2023, 9). *Technical Guidelines for Disposal to Land (Revision 3.1).* Retrieved from Waste Management Institute New Zealand: https://environment.govt.nz/assets/publications/Waste/Version-3.1-TG-for-Disposal-to-Land-September-2023.pdf?utm_source=chatgpt.com

Water Services (Drinking Water Standards for New Zealand) Regulations. (2022). *Water Services (Drinking Water Standards for New Zealand) Regulations 2022*. Retrieved from New Zealand Legislation: https://www.legislation.govt.nz/regulation/public/2022/0168/latest/whole.html

Water Services Act. (2021). *Water Services Act 2021*. Retrieved from New Zealnad Legislation: https://www.legislation.govt.nz/act/public/2021/0036/latest/whole.html

Williams, P. T. (2005). *Waste Treatment and Disposal*. England: John Wiley & Sons, Ltd. Retrieved from https://books.google.co.nz/books?hl=en&lr=&id=ZH9kkhd56IcC&oi=fnd&pg=PP11&dq=waste+treatment+and+disposal+Williams+(2005)&ots=gvRwjxB6wo&sig=BEhY7YVLeNLo2W_9YrjMjjG1xgA#v=onepage&q=waste%20treatment%20and%20disposal%20Williams%20(2005)&f=false

Williams, P. T. (2005). Waste Treatment and Disposal (2nd ed.). doi:DOI:10.1002/0470012668

Xiong, Y. D. (2024, 02 15). Characteristics of dissolved organic matter contribute to Geogenic ammonium enrichment in coastal versus alluvial-lacustrine aquifers. *Water Research, 250*(121025). doi:https://doi.org/10.1016/j.watres.2023.121025

Yang, S. L. (2021). Methanogen Community Dynamics and Methanogenic Function Response to Solid Waste Decomposition. *12*. doi:https://doi.org/10.3389/fmicb.2021.743827