Riparian Restoration Benefits for Rural Streams in the Mahurangi Catchment

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Stream monitoring reach, forested site, Mahurangi Catchment. Photograph by Danny McDougall.

Inside image credit, page iii
Mixed land cover hill country, Te Muri Regional Park. Photograph by Merrin Whatley.

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Executive summary

Te Waihē / Mahurangi Harbour and its surrounding catchment is characterised by steep hill country, stunning harbour views and numerous small freshwater streams and harbour inlets. Located on the north-eastern coast of the Auckland region, long-term monitoring reflects sedimentation, with ecological shifts in intertidal areas, including more mud-tolerant taxa and fewer sand-dependant species. Sediment source tracking identifies erosion from pastoral land as a key contributor. Reducing erosion from rural land is therefore essential to protecting the environmental health and cultural values of the Te Waihē / Mahurangi Harbour.

The Mahurangi Land Restoration Programme (MLRP) was a $6.15 million programme funded by the Ministry for the Environment (MFE) Manatū mō te Taiao that was initiated by Auckland Council in partnership with the Ngāti Manuhiri Settlement Trust in 2020. The goal of the MLRP was to increase the mauri of the Te Waihē / Mahurangi Harbour by achieving measurable reductions in human-induced sediment loss from rural land, primarily through supporting landowners to retire and plant riparian margins, wetlands and erosion-prone hill country. Most of the waterways that are retired as part of the programme were small, headwater, tributaries including springs and seeps. The MLRP builds on over 20 years of land restoration in the catchment, commencing with the Mahurangi Action Plan in 2004.

This report presents findings from a three-year, space-for-time stream monitoring programme (2023-2025). Monitoring was designed to assessing whether riparian restoration (fencing and native planting) measurably reduced fine sediment levels and improved ecological health in headwater streams. Nine permanently flowing headwater streams were repeatedly monitored. Streams represented a gradient of riparian habitat quality and stock exclusion. Each stream was assigned to one of three riparian treatment groups: 1) open pasture (unrestored), 2) fenced – planted native scrub (restored), and 3) fenced – mature native forest (reference). Indicators include deposited fine sediment, turbidity, macroinvertebrate indices, the taxonomically independent community index (TICI), and reach-scale habitat quality assessed using the Rapid Habitat Assessment (RHA) method. The role of catchment-scale characteristics, including catchment size, native forest cover and stock exclusion, were also investigated.

Results showed a clear gradient in reach-scale habitat quality across the sites, being highest in forested reference streams, intermediate in restored streams and lowest in open pasture streams. Reach-scale habitat quality and the degree of stock exclusion in the catchment were the strongest predictors of fine sediment levels in streams.

Restored streams contained approximately 32% less fine sediment than open pasture streams. Inorganic mud (< 63 μm) accounted for approximately 67% of the total sediment in open pasture streams, compared to 29% in restored streams and 37% in forested streams. High frequency turbidity monitoring showed similar patterns.

Ecological health indicators – including the Macroinvertebrate Community Index (MCI), Quantitative Macroinvertebrate Community Index (QMCI), pollution sensitive mayflies, stoneflies and caddisflies

(EPT) and TICI – were also positively correlated to reach-scale habitat quality (RHA) and key catchment characteristics (native bush cover, the degree of stock exclusion and catchment size). There were key differences in sensitive EPT taxa between riparian treatment groups. All three groups of EPT taxa were present in reference forested streams, mayflies and caddisflies were present in restored streams and only a few caddisfly taxa were present in open pasture streams.

Native freshwater species, including kākahi (freshwater mussel; *Echyridella menziesii*), giant kōkopu (*Galaxias argenteus*), and the regionally rare redfin bully (*Gobiomorphus huttoni*), were detection through environmental DNA analysis (eDNA). These species were predominantly found in forested and restored headwater streams, highlighting the importance of headwater streams as habitat for native fauna.

Overall, this study demonstrates that riparian restoration, through stock exclusion and native planting, is likely to be reducing fine sediment and improving ecological health in headwater streams. If left unprotected, headwater streams have the potential to deliver significant amounts of fine sediment cumulatively across the Mahurangi catchment. When protected, sediment levels reduce and the diversity of native species increases. The findings support prioritising headwater stream restoration as an effective strategy for improving catchment-wide outcomes.

Future restoration should prioritise headwater reaches as high-value intervention points, using eco-sourced native plant mixes, including 20% to 25% long-lived species, supported by sustained pest and weed control. Connecting restored areas to create continuous riparian corridors, linked to native forest remnants, will enhance recolonisation and ecological recovery.

A catchment-wide approach remains essential. Riparian management is most effective when combined with other mitigation approaches, such as spaced planting or afforestation of gullies and erosion-prone slopes, wetland enhancement, good soil management practices to reduce bare ground and compaction, and the use of vegetated filter strips and small detainment bunds to intercept runoff. Importantly, increasing catchment-wide forest cover and forest edge density, is expected to moderate peak flows, a primary driver of erosion and sediment mobilisation.

Ongoing monitoring should include periodic assessments of restored streams across the Mahurangi catchment using sediment, ecological and cultural indicators, alongside targeted monitoring of peak sediment-loss events. The degree of forest cover and any changes in forest edge extent, should also be documented.

Central to the long-term success of catchment restoration programmes is maintaining a strong partnership with mana whenua, landowners and the wider community. Recognising that ecological restoration is people powered and unfolds over decades.

He mauri tō te whenua, he mauri tō te wai, he mauri tō tangata.

There is a life force in the land, a life force in the water, and a life force in people.

Contents

1 Introduction

The Mahurangi catchment covers approximately 128 km^2 and lies about 40 km northeast of Tāmaki Makaurau / Auckland. It is centred around the Te Waihē / Mahurangi Harbour, which covers 25 km^2 and forms part of the Tīkapa Moana o Hauraki / Hauraki Gulf. The landscape comprises steep hill country, coastal inlets and numerous small streams which flow into the harbour. The area falls within the rohe of Ngāti Manuhiri and 10 other iwi groups. The Mahurangi is highly valued by both mana whenua and the wider community for its cultural importance and beautiful natural landscapes which offer lifestyle, fishing, farming, and recreational opportunities. Mahurangi is one of 10 watersheds identified in the Auckland region that centre around its unique harbours and coastlines as the main marine receiving environments. The harbour's location in the region is shown in Figure 1.

The Mahurangi catchment is characterised by its numerous small headwater streams, which flow through steep erodible hill-country. Headwater streams are recognised as being ecologically important waterbodies in the Auckland region (Parkyn, et al., 2006). Their small size makes them particularly vulnerable to the impacts of livestock access (Holmes, et al., 2016; Hughes, 2016) and significant volumes of sediment have been recorded moving through headwater streams, particularly during storm events (Hicks, et al., 2021; Tsyplenkov & Neverman, 2025). The cumulative transportation of sediment via headwater streams is therefore considered to contribute significant amounts of sediment to larger waterbodies in the catchment, the Mahurangi River and Te Waihē / Mahurangi Harbour.

The main environmental risk in the catchment is human-induced sedimentation from land-based activities including farming, urban development, and forestry. Sediment is impacting freshwater environments and degrading the ecological health in the harbour and its surrounding catchment (Drylie, 2025; Surrey & Storey, 2025; Tsyplenkov & Neverman, 2025).

Long-term monitoring of Te Waihē / Mahurangi Harbour's shallow, intertidal environment captured the rapid decrease in horse mussel (*Atrina zelandica)* beds across the estuary and concomitant declines in abundance of taxa sensitive to fine sediment between 1994 and 2000. Trends up until 2018 documented the continued increase in mud-tolerate taxa and decrease in some mud-sensitive taxa. Mud content was elevated at most sites in the harbour, impacting marine benthic (sand and mud dwelling fauna) health. Mud content was reported as increasing in Dyers Creek (Carter & Hailes, 2020). The most recent Auckland Council report on the state and trends (1995-2023) of Ecological Health in Tāmaki Makaurau / Auckland reported elevated mud content in sampled sediments at most locations across Te Waihē / Mahurangi Harbour. While trends in mud content could not be detected, numerous changes in species composition were indicative of sedimentation and the ecological health of most sites in the harbour were noted as fair to marginal due to impacts from sediment (Drylie, 2025).

Investigations into sediment sources and deposition levels across the harbour were investigated by Gibbs (2006) by applying compound specific isotope analysis of sediment cores collected across Te Waihē / Mahurangi Harbour in 2005. The study identified how key land uses (covering pasture, native forest and exotic pine forest) had contributed sediment to the harbour and how it was distributed across the waterbody. Soils from both pasture and exotic forests were found to have contributed a significant proportion of the sediment load to the harbour. Soil originating from pastoral land accounted for between 10% to 55% of the soil extracted from sediment cores and was found throughout the harbour, except near the entrance. Exotic pine forest was estimated to contribute approximately 14% of the total sediment load across the harbour, which was disproportionately high considering it only accounted for 8% of the catchments land area at the time. This study followed a period of forest harvest and was, therefore, likely to capture soil that had been mobilised following land clearance (Gibbs, 2006).

Stream walks and other ground-based assessments across the catchment have previously identified a lack of riparian vegetation cover and stock exclusion as contributing factors to stream bank erosion, damage to instream habitats and the mass movement (erosion events) on steeper slopes (Baddon, et al., 2010; Pohe & Jelley, 2005; Hicks & Hawcridge, 2004). More recently, expansion of urban development and wastewater overflows have introduced additional organic matter and inorganic fine sediment to the harbour in recent years. While direct attribution is not possible, harbour monitoring data show indicators consistent with nutrient enrichment and increasing mud content (Drylie, 2025).

Fine deposited inorganic sediment (< 2 mm) is widely recognized as being the most ecologically damaging, contributing to a decline in sensitive invertebrate and fish species (Clapcott, et al., 2011; Davis, et al., 2022; Wood & Armitage, 1997). These fine particles can be delineated further to muddy fines (< 63 μm) which represented the most mobile, abundant particles, and are therefore considered the most ecologically harmful. The accumulation of fine particles reduces habitat complexity in streams by smothering and homogenising benthic habitats. Their relatively high surface area, compared to larger particles, enhances their capacity to adsorb and transport other compounds, including nutrients, pesticides, heavy metals, and harmful microbes. As these fine particles accumulate, they act as vectors for these contaminants, compounding their impacts on water quality and ecological health (Clapcott, et al., 2011; Davies-Colley, et al., 1992; Gupta, et al., 2022; Ryan, 1991; Wood & Armitage, 1997).

1.1 Catchment actions and the Mahurangi Land Restoration Programme (MLRP)

In response to the recognised sedimentation issue within the catchment; Auckland Council, mana whenua and local communities have worked together for more than 20 years on various catchment restoration initiatives. Early coordinated action was formalised under the Mahurangi Action Plan in 2004, and successive, largely council led programmes have supported on farm mitigations, riparian protection, erosion control and community engagement (Auckland Regional Council, 2004).

The Mahurangi Land Restoration Programme (MLRP), launched in 2020, continued this long-term kaupapa (initiative) as a partnership programme between Auckland Council and the Ngāti Manuhiri Settlement Trust (NMST). MLRP was awarded $5 million of funding over six years through the

Ministry for the Environment's Jobs for Nature programme, with Auckland Council contributing an additional $1.15 million. The programme supports Auckland Council's strategic direction by linking catchment-scale restoration actions to measurable improvements in sediment reduction and ecological health, consistent with the Auckland Plan 2050 and Auckland Unitary Plan objectives. The goal of the programme was to increase the mauri (life force) of Te Waihē / Mahurangi Harbour and its waterways by achieving measurable reductions in human-induced sediment from rural land management activities and improving overall water quality.

A key action to help achieve this goal has been the provision of contestable grants for landowners in the catchment over a six-year period to provide co-funding for riparian restoration actions and afforestation of steep, erodible slopes. Alongside targeted landowner co-funding, a three-year stream monitoring programme has been run to assess the benefits of riparian protection measures, which have been successively implemented over the last 20 years throughout the catchment. Headwater streams (first and second order tributaries) were the focus of the monitoring programme because they represent the greatest proportion of waterbodies flowing into the harbour and are more often restored, through co-funded riparian remediation projects than larger streams and rivers. This report presents the findings of the stream monitoring programme.

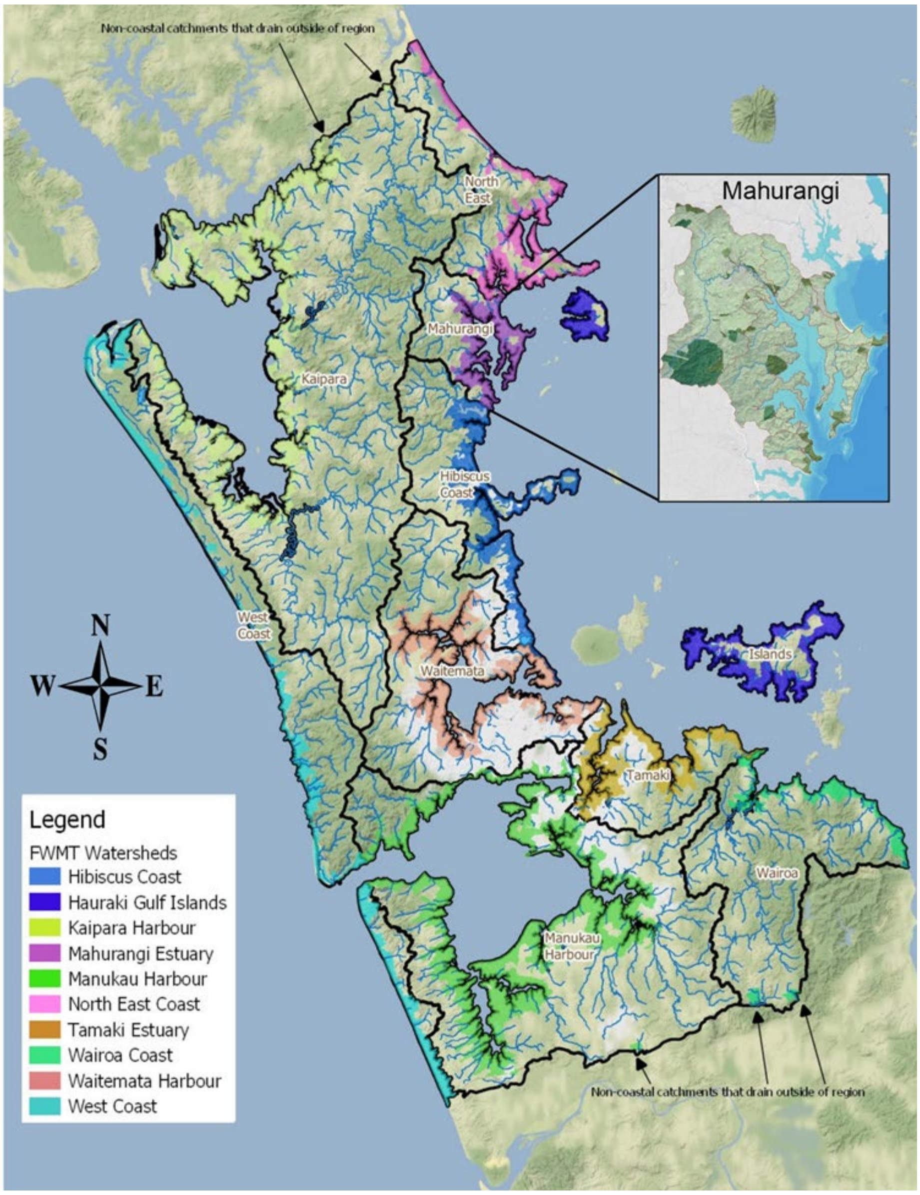


Figure 1. Mahurangi catchment in the Auckland region. Freshwater management tool (FWMT) watersheds, indicated by different colours along the coastline. Map adapted from Auckland Council (2021).

1.2 Report purpose and scope

This report brings together monitoring data collected during the MLRP to evaluate the effects of riparian restoration on sediment deposition, sediment transportation and ecological health in headwater streams. It presents the results of a three-year monitoring programme carried out in nine streams spread across the catchment and highlights the following:

- Evidence that riparian restoration can measurably reducing sediment deposition and mobilisation in headwater streams.
- The influence of riparian restoration on ecological health in headwater streams.
- Recommendations for future restoration work and monitoring priorities.

2 Background

2.1 Geology, soils and the local environment

Geology and landform in the Mahurangi catchment increase its vulnerability to erosion. Waitemata turbidite sandstone underlies about 85% of the catchment; limestone, alluvium and mudstone make up the remainder. Dominant soils include Whangaripo clay and sandy clay loams. The climate is warm, wet and temperate (average monthly temperatures 11.5 - 19.7 °C) with a median annual rainfall of 1,454 mm/yr (1995-2020) and high month to month variability (61 - 573 mm). Steep terrain is widespread: 37% of the catchment has slopes >25° and is classed Land Use Capability (LUC) 6e or greater, indicating limited productive potential on some hill country and elevated erosion risk where vegetation cover is low, or stocking rates are moderate. An example of LUC6e land in Te Muri Regional Park that was previously grazed and is in the process of being planted up is shown in Figure 2.

Land use in the catchment is dominated by dry stock farming (~59%), but it still retains significant pockets of native vegetation, mature forest and regenerating scrub (~24%) and, to a lesser extent, exotic commercial forestry (~11%). There is rapidly increasing urban development (~6% at the time of writing). Horticulture occupies <1% of the catchment. These land uses, combined with topography and soils, facilitate soil disturbance and the potential mobilisation of fine sediment to be transported to the harbour.

Figure 2. This photo was taken in Te Muri Regional Park, showing recent native plantings in the foreground on highly erodible hill-country. The visible erosion scars in the background occurred during cyclone Hale in January 2023.

2.2 Sediment impacts in the Mahurangi catchment

Evidence for erosion-fed soil loss from land and the resulting sedimentation of the Te Waihē / Mahurangi Harbour is widely published (Gibbs, 2006; Hicks & Hawcridge, 2004; Temple & Parsonson, 2014). Soils from exotic pasture and exotic forestry have been found to contribute a significant portion of the land-derived sediment to harbour inlets and the Mahurangi River mouth. Depending on the location in the harbour, pasture accounted for between 10% to 55% of recently deposited sediment and exotic forestry accounting for between 45% to 80% (Gibbs, 2006).

Monitoring of marine benthic ecology (since 1994) and freshwater monitoring (since 1993) in the catchment also reflects decadal shifts in sediment transportation and accumulation rates. In the intertidal harbour environment the benthic communities changed significantly over the first six years of monitoring (1994-2000). Mud tolerant taxa, like the invasive Asian semele bivalve (*Theora lubrica*) and Cirratulid polychaetes (*Cirratulidae*) increased alongside notable decreases in sediment sensitive species like horse mussels (*Atrina zelandica*). These changes suggest an ecological shift in community composition arising from fine sediment accumulation in the harbour (Carter & Hailes, 2020; Cummings, et al., 2003). Long-term monitoring of marine benthic communities, between 1994 and 2018, showed consistent trends across intertidal zones in the harbour. Half of all sites had moderate to poor ecological health, and an overall increase in worms, associated with terrigenous (land-derived) muddy particulates. Mud sensitive taxa, including the wedge shellfish (*Macomona liliana*), the nut shellfish (*Linucula hartvigiana*) and the limpet-like marine gastropod (*Notoacmea scapha*), decreased at multiple sites over the monitoring period (Carter & Hailes, 2020). More recently Drylie (2025) highlighted that Te Waihē / Mahurangi Harbour was in 'Poor' ecological health and showing signs of further degradation with excess sediment underlying the current condition and trends.

In freshwater environments, turbidity measurements collected between 2010 to 2019 in the Mahurangi River indicated increasing rates of sediment transportation in the upper catchment (Ingley, 2021). Between 2020 and 2024, the Mahurangi River recorded the highest annual average sediment yields (221 t/km^2/yr) across eleven regional monitored sites and Te Muri-o-Tarariki stream also returned regionally 'high' yields (132 t/km^2/yr) (Tsyplenkov & Neverman, 2025).

Water quality index scores have declined between the 2014 to 2016 and 2019 to 2021 monitoring periods. This decline was gradual in the lower catchment at Warkworth (falling from 73.5 to 71.9), and more pronounced at the exotic forest site in the upper Mahurangi River catchment, where scores dropped from 80.8 to 65.1, over the same period (Ingley, et al., 2023). Both sites are classed as having 'fair' conditions at, indicating that water quality is occasionally impaired, with declines reflecting elevated turbidity in the upper catchment and intermittent increases in nitrogen and phosphorus at both sites. Notably, total suspended sediment and turbidity at Warkworth exceeded the 98th percentile over the past decade during a major storm event on the 23rd September 2021 (Ingley, et al., 2023).

Freshwater ecological health indicators show that both Macroinvertebrate Community Index (MCI) and Quantitative MCI scores (QMCI) are 'likely degrading' in the Mahurangi River (Right Branch) and MCI is 'very likely degrading' in the native forest site in Dyers Creek; opposite to the regional trend where 75% of low-intensity rural sites show improving trends over a ten-year period. Conversely, MCI and QMCI scores are ‘likely improving’ at the Dyers Creek Pasture site, and ‘very likely improving’ at Te Muri-o-Tarariki, a recently restored catchment. Stream Ecological Valuation scores (SEV) were likely or very-likely degrading at three out of four sites over the 15-years of monitoring (Surrey & Storey, 2025).

Together, these results indicate that the harbour and freshwater environments continue to be affected by sediment and impaired riparian habitat quality (SEV), while also showing early signs of recovery in Te Muri-o-Tarariki stream, where stock exclusion and native revegetation have been implemented.

2.3 Monitoring and evaluation framework

Prior to the current stream monitoring programme being developed, an integrated monitoring and evaluation framework was created for MLRP to guide the design and implementation of the freshwater monitoring programme. The framework provided guidance and references to monitoring techniques, frequency, and the criteria for selecting monitoring sites. The framework is complementary to the cultural monitoring framework developed by the Ngāti Manuhiri Settlement Trust; refer to Section 3.3.1.

The monitoring and evaluation framework draws from the programme goal and outlines seven stages of monitoring and data evaluation. The monitoring results presented in this report relate to steps 3 (monitoring sediment mitigation actions) and 5 (monitoring freshwater ecological health) of the framework; see Figure 3 (Whatley, 2022).

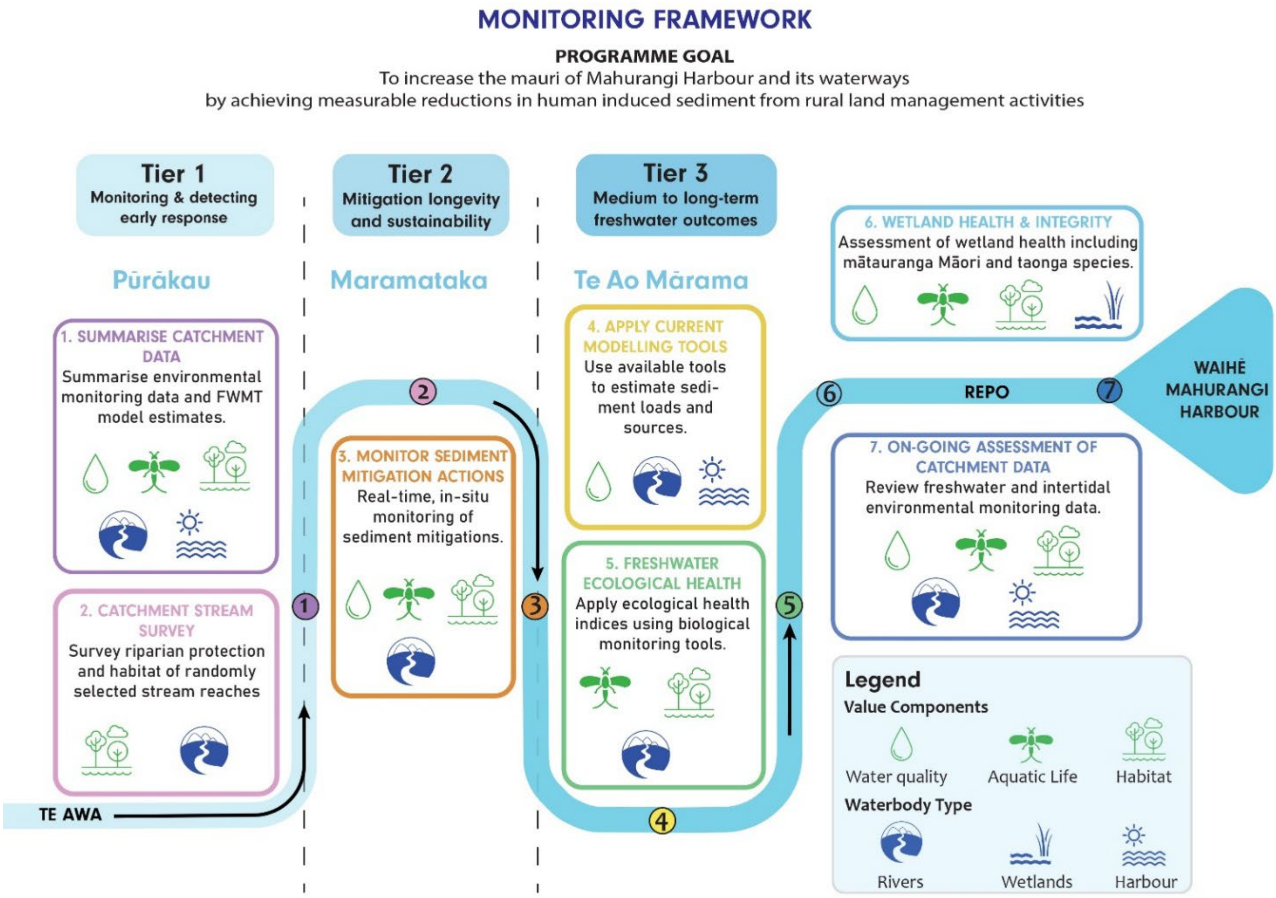


Figure 3. The MLRP Monitoring Framework guiding the field monitoring programme.

3 Stream monitoring programme

3.1 Research questions

Three research questions were defined to evaluate whether the MLRP was meeting its goals:

1. Is riparian restoration in the Mahurangi catchment measurably reducing fine sediment levels in streams?
2. Are these mitigations enhancing the ecological health and mauri of streams?
3. Has the occurrence and severity of bank erosion in streams decreased across the catchment?

This report addresses Questions one and two. Question three (stream bank erosion) was investigated separately, through a stream walk survey, with results presented in 'Mahurangi Riparian Surveys: 2022 Survey Results' (Drummond, et al., 2022). The stream walk builds on the earlier stream surveys carried out in 2005 and 2010 by Pohe & Jelley (2005) and Baddon, et al., (2010), which benchmark the level of riparian protection and riparian conditions at the start of the Mahurangi Action Plan (MAP) and MLRP, respectively.

3.2 Monitoring design

The MLRP stream monitoring programme was designed to compare conditions in headwater streams where riparian restoration (stock exclusion and native planting) has been implemented with those from unrestored open pasture streams. Forested streams were also included to provide a reference set representing headwater streams with minimally impacted riparian and catchment conditions.

Given the timescales of the study, a space-for-time design was used to evaluate the results between the three treatment groups, each representing a distinct level of riparian protection:

- Open pasture (Unrestored): no stock exclusion or native planting along the monitored reach. Less than 20% of the upstream riparian margin, above the monitored stream reach, is fenced and/or planted.

- Fenced – Planted native scrub (Restored): full stock exclusion and planted scrub (native trees and shrubs), at least 10 years old. At least 70% of the upstream riparian margin is fenced and planted with established natives.

- Fenced – Mature native forest (Reference): full stock exclusion of unmodified streams under mature native forest (over 50 years old). Over 50% of the upstream catchment is fenced and under mature native forest.

The below diagram (Figure 4) shows examples of stream reaches representative of each test group.

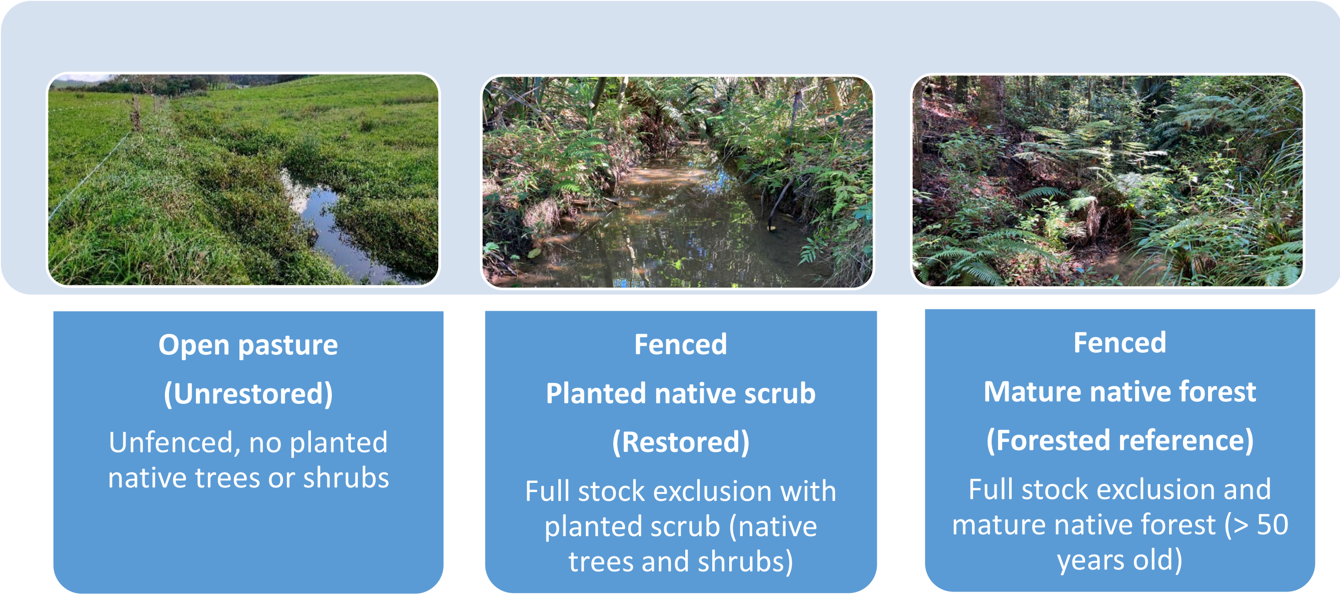


Figure 4. Examples of stream reaches in each of the three treatment groups: Open pasture (Unrestored), Fenced – Planted native scrub (Restored), and Fenced – Mature native forest (Forested reference).

3.2.1 Site selection

A total of nine streams, three for each treatment group, were included in the MLRP monitoring programme. Streams were selected using the criteria outlined above to represent a gradient of overall habitat quality, based on the presence and condition of native riparian vegetation and the extent of stock exclusion. Gradients in riparian habitat quality and catchment characteristics were also present within each treatment group, as described on page 13.

All nine streams had broadly similar geology and soil types (GNS, 2013; Manaaki Whenua / Landcare Research, 2024). Pakiri formation was the dominant underlying rock type, represented by alternating (thick-bedded) volcanic-rich, graded sandstone and siltstone. Several streams (4, 6 and 9) also contained small areas of Mahurangi limestone in the upper reaches of their catchments. Ultic, weathered clay was the dominant soil type in all catchments. Two streams (3a and 3b) also had small areas of alluvial soil in their lower reaches, while streams 2 and 5 contained discrete areas of gley soil, primarily in their lower catchments.

Although all streams were initially identified as permanently flowing, stream 3a ran dry during summer 2024. A nearby stream with similar geological characteristics was, therefore, selected as a replacement (3b), and monitoring continued at this location for the remainder of the programme. The two streams (3a and 3b) differed in upstream land cover, with stream 3b having a greater proportion of forest cover in the upstream catchment. These differences were documented and taken into consideration when interpreting the results.

Where possible, streams were selected that had upstream land cover like that of the assigned riparian treatment group, allowing the influence of upstream land cover and management practices on sediment and aquatic communities to be accounted for. This wasn't achieved for sites 1 and 3, however, which had significant amounts of native scrub and forest in the upstream catchment, 72% and 65% native vegetation cover, respectively. The location and the dominant land cover in the catchment for each monitored stream is shown in Figure 5.

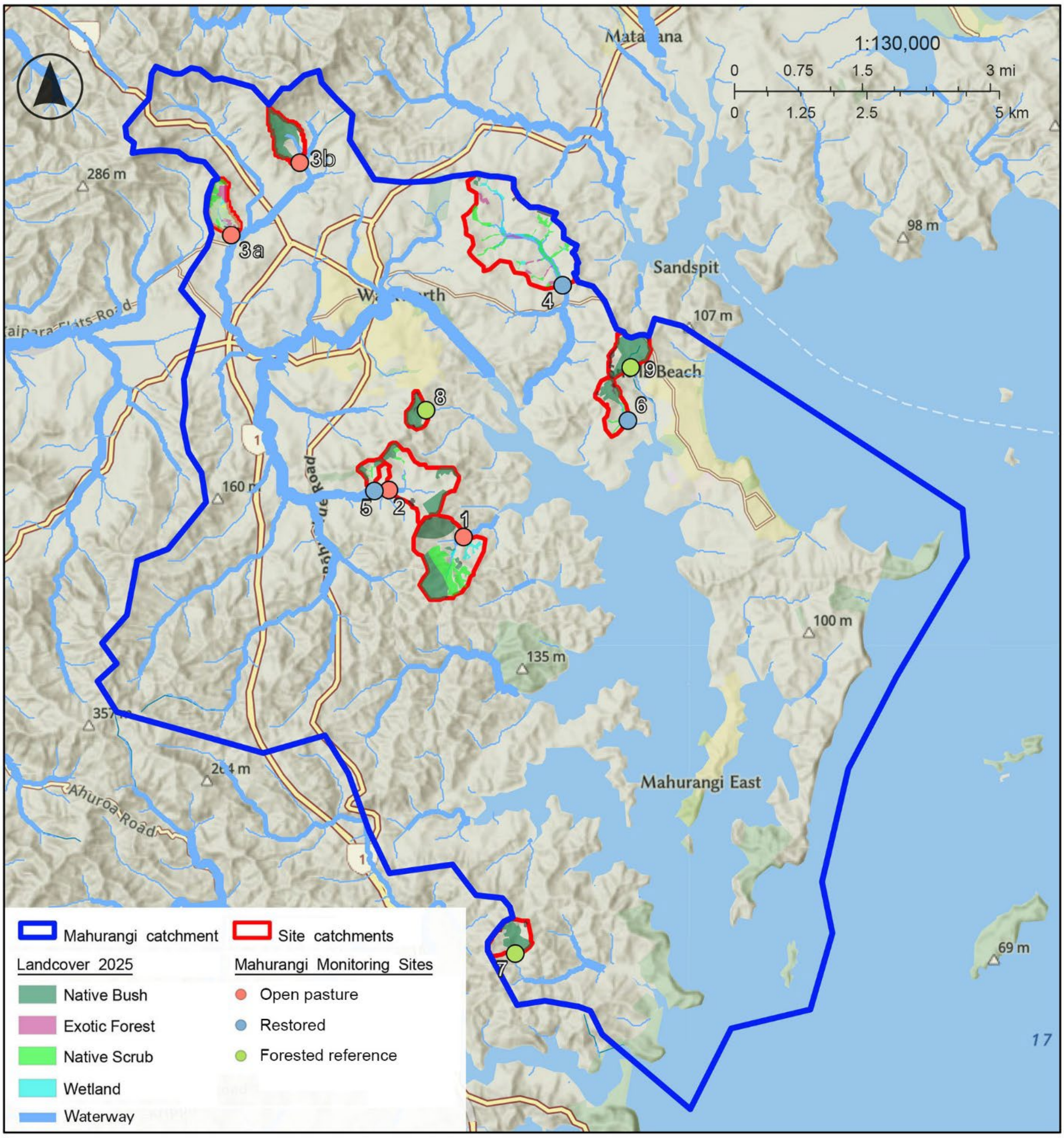


Figure 5. Monitoring site locations, colours denote the following treatment groups, Open pasture, Restored and Forested reference. Waterways are derived from the River Environment Classification of New Zealand (REC2).

To reduce the degree of variation between sites the selected tributaries were all headwater streams, being classed as first or second order streams (under the Strahler stream order classification system) (Strahler, 1957), with an average wetted channel width less than 2 m and upslope contributing area (catchment) no greater than 300 ha. The Strahler Stream order classification system classifies streams as follows: headwater streams at the top of a catchment – with no other streams feeding into them – are named first-order streams, streams with two or more first-order streams flowing into them are second-order streams and streams with two or more second-order streams flowing into them are third-order streams, and so on (Figure 6).

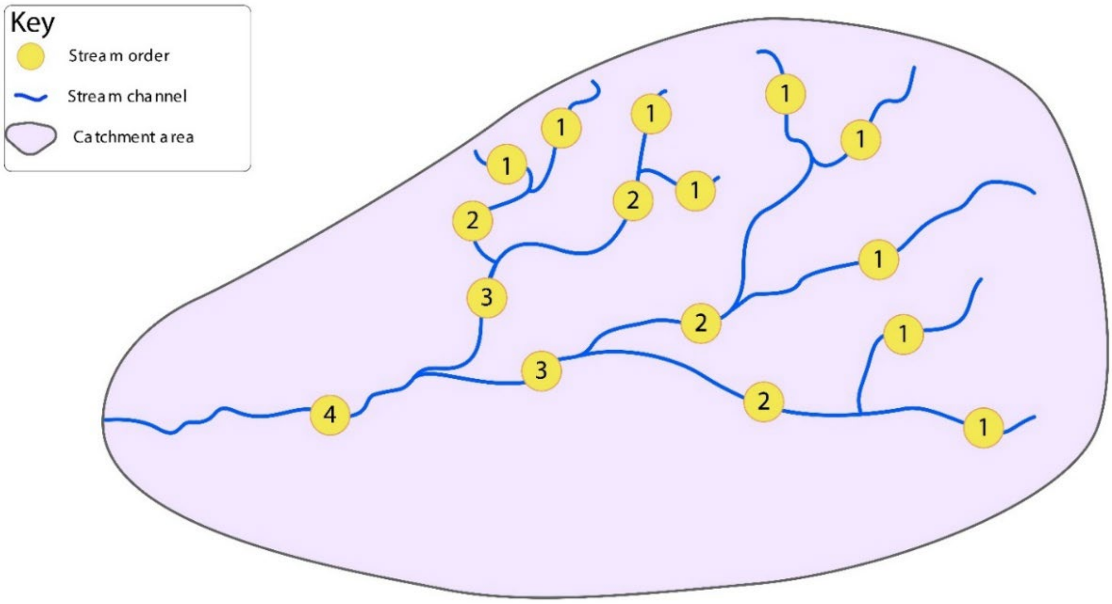


Figure 6. A diagram illustrating a stream catchment area and Strahler stream order classification system.

Key characteristics of the monitored streams, representing the three treatment groups, are outlined below:

- Open pasture sites (unrestored): All sites were rotationally grazed by cattle, although grazing intensity was much less at site 1, resulting in a dense sward of kikuyu grass along the stream banks. Sites 2 and 3 had an electric fence running along one side of the stream, but this did not prevent stock access.
- Fenced – Planted native scrub sites (Restored): All streams had been permanently retired from grazing at least six years before monitoring began. Sites 5 and 6 were fenced and planted at least 12 years prior, while site 4 had been retired for at least six years with riparian planting over 10 years old upstream of the monitored reach. A short section of temporary fencing excluded livestock downstream of site 4. Fence set-back distances varied, being widest at site 4 (14 m on average) and narrowest at site 5 (5 m on average). Canopy cover was greatest at site 6 and more open at sites 4 and 5, largely due to early flooding at site 4 and the narrower set-back at site 5. Both sites 4 and 5 had some invasive weed incursion.
- Fenced – Mature native forested sites (Forested reference): These streams had protected forested riparian margins and substantial forest cover across their catchments (54% - 77%). The established native forest ecosystems are characterised as kauri forest (sites 7 and 9) and kauri, podocarp, broadleaved forest (site 8) (Singers, et al., 2017). Sites 8 and 9 originating in protected areas (Parry Kauri Park and Lawries Scenic Reserve). Site 7 had only been fully retired from grazing two to three years before monitoring, resulting in a noticeably more open understorey. There were also signs of kauri dieback disease (*Phytophthora agathidicida*) at site 7.

Fencing set-back distances and the planting years for riparian vegetation for the monitored streams are summarised in Table 1, and sites photos are shown in **Figure 7**.

Table 1. Riparian set-back distance and vegetation age in the monitoring reach for the nine headwater streams monitored under the Mahurangi Land Restoration Programme.

Site ID	Treatment	Riparian set-back distance (m)			Year of planting
		Min	Average	Max	
1	Open pasture (Unrestored)	0	0	0	Unplanted
2		0	0	0	Unplanted
3a & b		0	0	0	Unplanted
4	Fenced – Planted native scrub (Restored)	4	14	21	2016-17
5		3	5	7	2010-11
6		2	10	22	2010-11
7	Fenced – Mature native forest (Forested reference)	30	184	298	Mature forest (> 50 years)
8		35	80	130	Mature forest (> 50 years)
9		30	63	101	Mature forest (> 50 years)

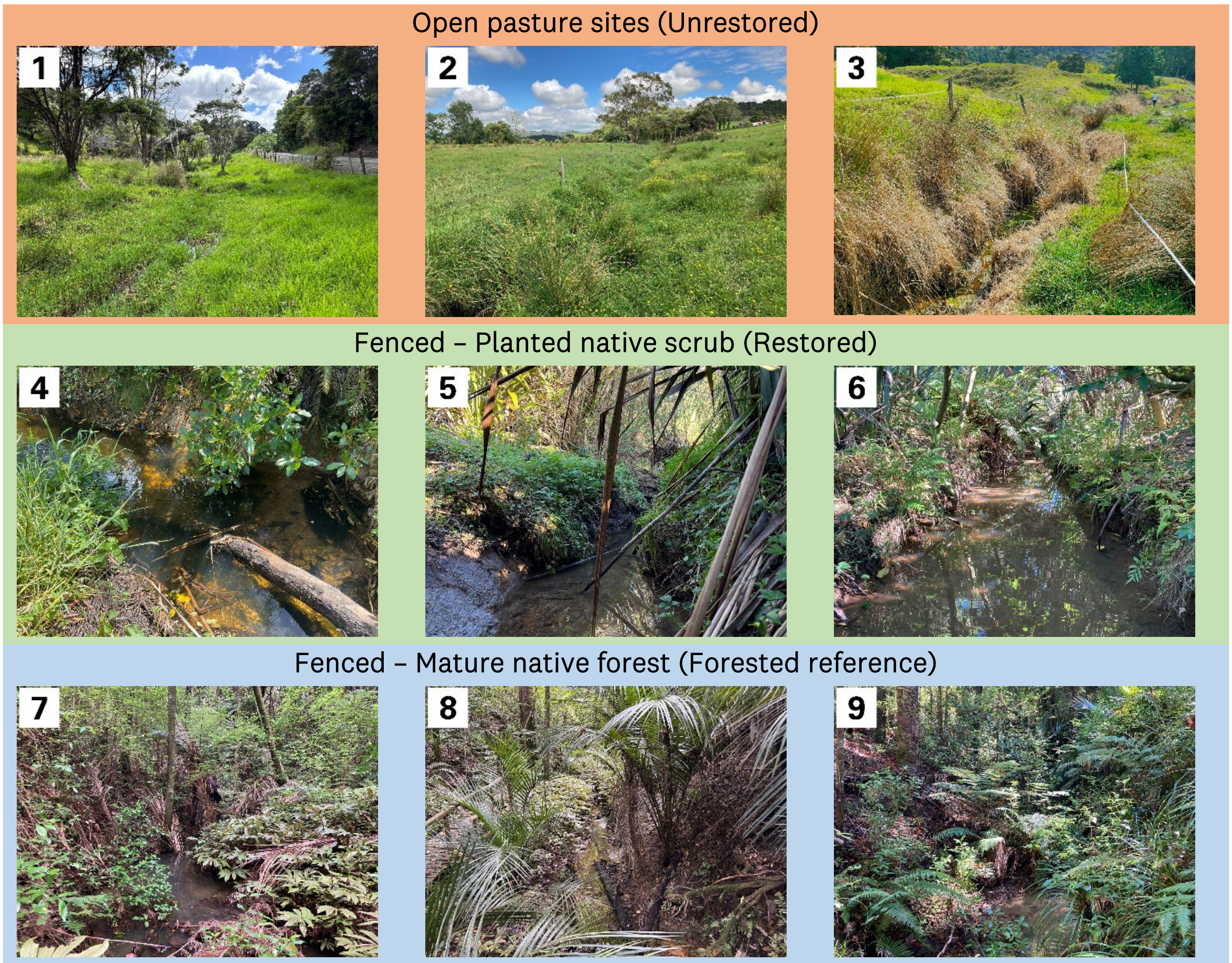


Figure 7. Photos showing the nine stream channels included in the monitoring programme.

Clear site-selection criteria were applied, but not all criteria could be met at every location. This resulted in an imbalanced monitoring design, with different stream characteristics unevenly represented across the nine sites. The final design captured a range of values, from low to high, for the fixed predictor variables, including catchment-scale and reach-scale characteristics.

The imbalance in the design was partially addressed by using cumulative link and linear mixed models and calculating Akaike weights (w_i) to help distinguish the relative influence of catchment-scale versus reach-scale variables. However, the small sample size limits the statistical power to detect relationships between catchment characteristics, riparian habitat quality, sediment indicators, and ecological health.

This formed a nested factorial design, where catchment-scale factors (e.g. catchment size, the percentage of different landcover types – native bush, native scrub and exotic pasture – and the degree of stock exclusion in the catchment) and reach scale-factors (e.g. riparian habitat quality) are nested within stream, as illustrated in Figure 8.

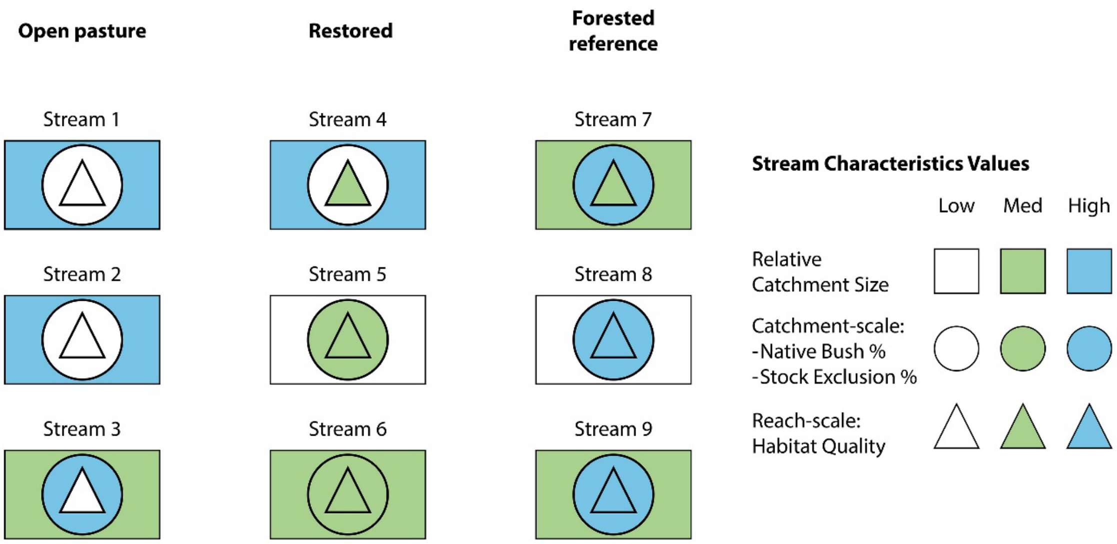


Figure 8. Schematic representing the final monitoring design showing upstream characteristics, including catchment-scale factors – percentage stock exclusion and native bush cover and reach-scale habitat quality.

3.3 Methods

Six monitoring techniques were used in the MLRP field monitoring programme to quantify fine sediment, habitat condition, and ecological health, including:

- **Sediment quantity, composition and transportation:** The Quorer method, shuffle index, and continuous suspended sediment monitoring using turbidity sensors
- **Instream and riparian habitat conditions, and catchment characteristics:** Rapid Habitat Assessment (RHA), mapping catchment land cover, proportion of stock exclusion and catchment area
- **Ecological health:** Macroinvertebrates, freshwater fish, kōura (freshwater crayfish), kākahi (freshwater mussels) and a taxonomic community index derived from environmental DNA (eDNA).

A description of each monitoring technique is provided below and summarised in Table 2.

Table 2. Overview of selected monitoring techniques including descriptions, what they measure, protocols and technical references.

Type of Assessment	Description	What it Measures	Protocols and Reference
Turbidity sensors (S40-SWW)	Continuous monitoring of suspended sediment	Suspended Sediment	Phathom Sensor Installation Guide & Point Orange IoT datasheet
Quorer method	A quantitative measure of the amount of suspendible sediment, including inorganic (SIS) and organic (SOS) sediment	Deposited Sediment	A modified version of the SAM 4 protocol published in Clapcott, et al. (2011)
Shuffle index	A qualitative measure of the suspendible fine sediment	Deposited Sediment	A modified version of SAM 5 protocol published in Clapcott, et al. (2011)
Rapid Habitat Assessment (RHA)	Provides a 'habitat quality score' for a stream reach based on 10 unique parameters. Includes photo points.	Habitat Integrity	Following the RHA protocol published on (Cawthron, 2021)
Catchment land cover and stock exclusion	Visual assessment, mapping and measurement of dominant vegetation types and stock excluded areas.	Catchment land cover types and livestock-exclusion extent	Visual assessment using Auckland Council satellite imagery (Auckland Council, 2024) at a spatial resolution of ± 0.08 m and watershed boundaries derived from the River Environment Classification 2 (version 5) (REC2) (NIWA, 2019).
Macroinvertebrates	Used to calculated ecological health indices: MCI, QMCI, EPT richness and %EPT abundance	Ecological Health	Using semi-quantitative protocols for soft-bottom streams (C2) and processing method (P2) (NEMS, 2022)
Environmental DNA (eDNA)	A non-invasive biodiversity monitoring technique. Used to detect freshwater fish, kākahi, kōura and calculate Taxonomic Independent Community Index (TICI)	Ecological Health	Modified from protocols published guidelines (Melchior & Baker, 2023)

In addition, Ngāti Manuhiri undertook cultural monitoring in parallel with field monitoring of sediment and ecological conditions. Although the cultural monitoring results are not presented in this report, collaboration between monitoring teams informed field practice and contributed to a more integrated interpretation of stream health.

3.3.1 Cultural monitoring

Cultural monitoring undertaken by the Manuhiri Kaitiaki Charitable Trust was included as part of an overall stream health assessment. Manuhiri Kaitiaki Charitable Trust is the operational arm of NMST and as kaitiaki (guardians) of the Mahurangi whenua (land), wai (water) and taiao (environment), they provide cultural oversight and leadership across the MLRP. Cultural monitoring ensures that restoration activities uphold the mana of Ngāti Manuhiri within their rohe and reflect the iwi's enduring relationship with their ancestral whenua, wai, wāhi tapu and taonga.

The cultural monitoring programme is embedded throughout the entire MLRP work programme, guiding restoration, planting and water-care activities so that tikanga (customary practices and principles) and mātauranga Māori (Māori knowledge) shape environmental decision-making. The programme supports the restoration of the Mahurangi catchment's mauri, strengthens iwi connections to the whenua and wai, and enables the intergenerational passing down of knowledge, pūrākau (narratives) and tikanga. It also supports strong community-iwi relationships and ensures that restoration actions respect culturally important landscapes while contributing to long-term environmental wellbeing.

Manuhiri Kaitiaki Charitable Trust kaitiaki led field-based monitoring using indicators that reflect both cultural and ecological health, drawing on observations of whenua, wai, hau (wind) and rerenga rauropi (biodiversity). All data is recorded through 'The Stream', a digital platform used nationally by iwi, which enables consistent monitoring, long-term knowledge retention and accessibility for future rangatahi (youth). This supports ongoing transmission of mātauranga Māori and strengthens adaptive management across the project. It is further guided by the Maramataka (Māori lunar calendar), ensuring seasonal patterns, energy flow and environmental rhythms are recognised when interpreting field conditions. This provides a culturally grounded lens for understanding changes in the taiao, with repeated observations showing clear correlations between Maramataka phases and environmental responses. These insights enhance decision-making and reinforce the holistic, values-based approach at the heart of the MLRP partnership.

Partnership processes between Manuhiri Kaitiaki Charitable Trust and project partners emphasise early engagement, open communication, and the incorporation of iwi perspectives at both governance and operational levels. This includes ensuring iwi decision-making is reflected in planning and reporting, and that activities are delivered in a manner consistent with the values of Te Tiriti o Waitangi, kaitiakitanga (guardianship) and manaakitanga (hospitality).

Through this cultural monitoring framework, the NMST ensures that all project activities contribute to the restoration of Mahurangi in a manner that uplifts mana of iwi authority, enhances cultural identity, and supports enduring environmental, social and cultural outcomes for iwi and the wider community.

3.3.2 Catchment characteristics

The catchment characteristics of each monitored stream were mapped visually using high-resolution satellite imagery displayed in ArcGIS. Briefly, Auckland Council satellite imagery (Auckland Council, 2025) at a spatial resolution of ± 0.08 m was overlaid with watershed boundaries derived from the

River Environment Classification 2 (version 5) (REC2) layer to delineate the catchment boundaries for each site (NIWA, 2019). Land-cover classes were delineated manually by interpreting visible features in the imagery, including vegetation type, canopy structure, land use, and the presence of built or modified surfaces. Each catchment was reviewed systematically to ensure full coverage, and mapped polygons were assessed against REC2 hydrological boundaries to confirm alignment with upstream contributing areas.

The following catchment characteristics were recorded for each site: catchment area, wetland extent, and the percentage of mature native bush, native scrub, exotic forest, exotic pasture, and stock exclusion (fenced areas around waterways). Land cover percentages are reported as a share of the total catchment area upstream of each monitoring location. The results were digitised and attributed within the GIS environment for subsequent analysis.

3.3.3 Reach-scale habitat characteristics

Riparian and instream habitat quality was measured once a year using the Rapid Habitat Assessment (RHA) (Clapcott, 2015). The RHA is a semi-quantitative method, comprising 10 parameters covering instream and riparian habitat conditions and floodplain connectivity, as follows:

1. deposited sediment
2. invertebrate habitat diversity
3. invertebrate habitat abundance
4. fish cover diversity
5. fish cover abundance
6. hydraulic heterogeneity
7. bank erosion
8. bank vegetation
9. riparian width
10. riparian shade.

RHA scores range from 0 to 100, with higher scores indicating better habitat quality and scores above 90 representing near-to intact habitat conditions. The quality grades from RHA scores are: Poor (< 25), Fair (25 – 50), Good (51 – 75), Excellent (> 75) (see Appendix A, Table A-1). The Rapid Habitat Assessment (RHA) was originally developed for wadable, hard bottomed streams (gravel–cobble dominated), but is now widely applied across both hard and soft-bottomed stream types in Aotearoa New Zealand including national scale reporting on freshwater physical habitat (Stats NZ/Tatauranga Aotearoa, 2020). Because RHA reference expectations are based on hard bottomed habitat structure, several metrics require cautious interpretation in soft sediment systems (e.g. invertebrate habitat diversity, invertebrate habitat abundance and hydraulic heterogeneity) which naturally score lower in soft-bottomed environments. However, as the monitored streams in this study were soft-bottomed (barring site 8 which had some small, discrete pockets of bedrock), RHA scores were comparable across sites.

Stream habitats in the Auckland region are typically assessed using the Stream Ecological Valuation (SEV). SEV is a detailed quantitative method that evaluates 14 parameters spanning hydraulic, biogeochemical, habitat, and biodiversity functions. It was developed for small, wadable streams in Auckland, including soft-bottomed streams. It is widely used by Auckland Council in state of environment reporting and for impact assessments. However, it was not applied in the Mahurangi Study due to time and resourcing constraints, and the RHA was identified as a suitable alternative.

To determine whether RHA scores adequately represented stream habitat conditions in the monitored soft-bottomed streams, RHA scores were compared to SEV scores (range 0 – 1) at three streams (2, 6 and 9), using the methods outlined in Neale, et al., (2015). Comparative analysis showed that reach-scale habitat scores derived from RHA and SEV were strongly correlated ($r^2 = 0.88$, $P = 0.004$). An evaluation of an earlier version of the RHA, completed for Northland Regional Council found the assessment reliably represented instream and riparian habitat quality, correlated with biological indices like MCI, and catchment factors like the degree of native forest cover (Clapcott, 2015). Together, these findings indicate that the level of detail provide by the RHA offers a reliable basis for evaluating reach-scale stream habitat characteristics in the Mahurangi catchment.

3.3.4 Sediment monitoring methods

Sediment monitoring focused on the measurement of fine sediment (< 2 mm) using both qualitative and quantitative methods. Assessments included fine deposited sediment, particle size analysis and suspended sediment, as outlined in this section.

3.3.4.1 Deposited fine sediment

Deposited fine sediment was assessed using the Quorer Sediment Assessment Method (SAM 4 protocol) and the shuffle index (SAM 5 protocol) following Clapcott, at al., (2011). In addition, bankside visual estimates of deposited-sediment cover were completed as part of the RHA method (similar to SAM1), as described in Clapcott, at al., (2011).

The Quorer method (SAM 4) is a quantitative assessment of resuspended (unconsolidated) sediment and includes volumetric estimates of resuspended inorganic and organic fractions. Although originally developed for hard-bottomed streams, several modifications were applied in this study to account for soft-bottomed stream environments. First, a grab sample of undisturbed water was collected to obtain background concentrations of suspended sediment, this was used as the control. Then, a 45 cm diameter cylinder was placed on the stream bed, to isolate a discrete column of water. To avoid disturbing the natural consolidated bed material, the stirring depth was set just above the stream bed so that only the unconsolidated sediment layer was agitated. This process was repeated six times along the stream reach, and the resulting samples were then combined to produce one composite sample per site.

Slurry samples were analysed at Hill Laboratories for Total Suspended Solids (TSS) and Volatile Suspended Solids (VSS), using standardised laboratory protocols. Suspended Inorganic Sediment (SIS) and Suspended Organic Sediment (SOS) were then calculated as follows:

- SIS (g/m^2) = ($TSS_{(sample - control)} - VSS_{(sample - control)}$) x average depth (m) in cylinder
- SOS (g/m^2) = $VSS_{(sample - control)}$ x average depth (m) in cylinder
- Average water depth (m) was used to calculate SIS or SOS in g/m^3

This method provides an estimate for deposited fine sediment per square metre of stream bed.

The shuffle index (SAM 5) is a rapid, qualitative measure of deposited fine sediment. A white porcelain tile (10 x 10 cm) is placed on the streambed within a run habitat, and the bed is agitated through foot movement approximately 1 m upstream. The time taken for the sediment plume to pass over the tile and clear again is recorded, with scores assigned on a scale of 1 – 5 (higher scores indicate larger sediment plumes). This process was repeated three times along each monitored reach, and scores were averaged for each site. Water depth and qualitative assessment of stream flow were recorded concurrently, as both factors can influence shuffle index results.

3.3.4.2 Particle size analysis

Particle size analysis was undertaken on the surface water samples collected at each site using the Quorer method (detailed above) to find out what fraction of the sediment collected was comprised of mud (< 63 μm). These very fine particles are of particular interest because they represent the most mobile and ecological damaging portion of all fine sediment. Briefly, particle size was measured by laser using either a EyeTech™ unit at Earth Sciences New Zealand's Water Quality Laboratory in Hamilton or Mastersizer Xplorere unit by Malvern Panalytical at the University of Waikato, following laboratory protocols.

3.3.4.3 Suspended sediment

Suspended sediment was measured using turbidity sensors. Due to resourcing constraints, sensors (Phathom S40-SWW) were installed in three streams only (sites 2, 4, and 9), with one stream representing each treatment group. Each sensor recorded data at 15-minute intervals, with readings uploaded daily via a remote telemetry unit, providing continuous suspended sediment monitoring in the three streams.

All sensors were calibrated to estimate Total Suspended Solids (TSS) from turbidity readings using standard silicate solutions ranging in concentration from 0 to 3,000 mg/L. Because particle size, colour, and the resulting light absorbance can vary between sites, sensor readings were also calibrated to field conditions. Briefly, field calibration was completed by collecting surface water grab samples adjacent to each sensor whilst recording the sensor's TSS readings concurrently. These samples were analysed at Hill Laboratories to generate a site-specific TSS gradient. Sensor readings were then adjusted using a regression equation derived from paired sensor and laboratory TSS measurements for each site.

TSS measurements from turbidity sensors were compared to local rainfall and discrete measurements of stream flow to assess the influence of these factors on suspended sediment concentrations.

3.3.4.4 Rainfall

Rainfall data was compared to TSS data collected by three sediment sensors over the same period. Rainfall records from two Auckland Council rain gauges, which record data at 15-minute intervals. For each sensor site, data from the nearest rain gauge were used.

Rainfall data from the Satellite Dish gauge (located approximately 0.25 km from site 2) were compared to TSS measurements recorded by the sensor at site 2. Rainfall data from the Warkworth sewage treatment plant gauge were paired with TSS measurements from sensors installed at sites 4 and 9, located approximately 2.4 km and 3.8 km away from rain gauge, respectively.

3.3.4.5 Stream flow

Stream flow was measured between September and December 2025 using a Global Water FP111 flow probe. Discrete flow measurements were collected at several points along each monitoring reach adjacent to the three sediment sensors. Resource limitations meant that only a small number of flow measurements could be obtained. Monthly measurements were collected in the three streams with sediment sensors, while a single measurement was collected in the remaining six streams (without turbidity sensors).

3.3.5 Indicators of ecological health

The ecological health of the nine streams was assessed by collecting aquatic macroinvertebrates and environmental DNA (eDNA) from surface water using the active syringe sampling method. Identifying species of note and calculating associated ecological indices (see Table 2).

3.3.5.1 Aquatic macroinvertebrates

Aquatic macroinvertebrates are small animals that live on or in the stream bed and submerged substrates, including stones, woody debris, plants and mud. They include a wide variety of insect larvae (*Insecta*), worms and leeches (*Lophotrochozoa*), molluscs (e.g. snails and kākahi / freshwater mussels) (*Mollusca*), crustaceans (kōura / crayfish, shrimp) (*Crustacea*), and spiders and mites (*Arachnida*). Macroinvertebrates are well established indicators of stream health because they respond rapidly to environmental change, and many species having complex life cycles that depend on key habitat features and resources such as food availability and quality, refugia, suitable water quality and habitat complexity (Clapcott, et al., 2017; Wood & Armitage, 1997).

All streams were sampled annually during the monitoring period between March and May (inclusive). This timing was selected to align with stable flow periods, when community assemblages are most representative of site conditions, and to maintain comparability with the first monitoring round, which commenced in May 2023 – the earliest feasible start date. Although this sampling window falls toward the later end of the recommended timeframes, samples collection between November and April remain eligible for the highest quality rating under the National Environmental Monitoring Standards (NEMS, 2022).

The monitored streams were predominantly soft-bottomed, although one stream (site 8) contained bedrock sections in the upstream reach. For consistency, all streams were sampled using a D-net following the C2 protocol for soft-bottomed streams, modified from Stark, et al., (2001). Samples were preserved in ethanol, and taxa were identified and quantified in a laboratory using the P2 protocol (NEMS, 2022).

Several macroinvertebrate Indices were calculated for this study, including the Macroinvertebrate Community Index (MCI), the Quantitative MCI (QMCI) (Maxted & Scarsbrook, 2003; Stark & Maxted, 2004; Stark, et al., 2001), and Ephemeroptera, Plecoptera and Trichoptera (EPT) richness and abundance. EPT taxa includes mayflies, stoneflies and caddisflies (excluding Hydroptilidae), which are sensitive to sedimentation, pollution and habitat degradation. The calculated indices were used for assessing stream ecological health and were selected because they are well established and widely used in ecological monitoring across Tāmaki Makaurau / Auckland and Aotearoa New Zealand, allowing for comparison of these results with other studies (Clapcott, et al., 2017) (see Appendix A, Tables A-2 and A-3 for NPS-FM (2020) attribute bands and ecological interpretation for macroinvertebrate indices).

3.3.5.2 Environmental DNA (eDNA)

eDNA was collected at all locations twice during the monitoring period (autumn 2024 and spring 2025). Samples were collected using the active syringe-filtration method. Due to resourcing constraints, the standard six replicates were reduced down to three replicate samples at each site. Species were reported as detected when a positive DNA signal was obtained and the species is either known to occur to in the local environment or is likely to be present based on their known range and habitat preferences.

Prevalence indexes were calculated for each species to indicate the strength of detection at a given site, following the methods outlined by Melchior and Baker (2023). Because only three replicates were collected (instead of the recommended six) the maximum possible prevalence index was 'Medium' rather than the usual maximum category of 'Very high' (Melchior & Baker, 2023).

Replicates were pooled to assess overall biodiversity and provide an indication of overall ecological health using the taxonomically independent community index (TICI) (Wilkinson, et al., 2024). The eDNA results were also utilised to assess freshwater fish diversity, including calculating the fish Index of Biotic Integrity (F-IBI) as detailed below. The presence of kākahi (freshwater mussels, *Echyridella spp.*) and kōura (freshwater crayfish, *Paranephrops planifrons*) were also detected from eDNA, both species are considered taonga to Ngāti Manuhiri and are sensitive to poor water quality, making them useful indicators of ecological health.

The F-IBI combines six metrics of fish community composition, which accounts for a sample locations distance to the sea and altitude. It was calculated based on the methods of Joy and Death, (2004) and using the Manatū Mō Te Taiao / Ministry for the Environment's online tool (Joy & Death, 2004; Ministry for the Environment, 2023a; Ministry for the Environment, 2023b).

Grading systems and ecological interpretation of the TICI and F-IBI are provided in Appendix A, see Tables A-3 and A-4. All monitoring methods were based on established protocols as referenced in Table 2.

3.3.6 Monitoring timeline

Field monitoring commenced in May 2023 and concluded in December 2025. Over this period, the nine sites were visited six times in total, once a year in autumn (March - May) and early summer (November - December) as detailed below.

Deposited sediment measurements using the shuffle index and the Quorer method were taken twice a year. Sediment particle size analysis, macroinvertebrate sampling and habitat assessments (RHA) were conducted annually, in autumn. A final RHA assessment and eDNA sample were collected at the end of the monitoring period in November – December 2026, prior to the conclusion of the field programme. eDNA samples were collected twice over March – April 2024 and November – December 2025. Suspended sediment data from the three turbidity sensors were collected between May – December 2025, following field calibration of the sensors.

The split-season monitoring approach is generally aligned with established ecological monitoring protocols (Clapcott, et al., 2011; Clapcott, 2015; Cawthron, 2021; NEMS, 2022; Wilkinson, et al., 2024) and coincided with periods of typically lower flows and more stable weather conditions, although autumn 2023 was notably wet. The timeline for each monitoring method is presented in Figure 9 below.

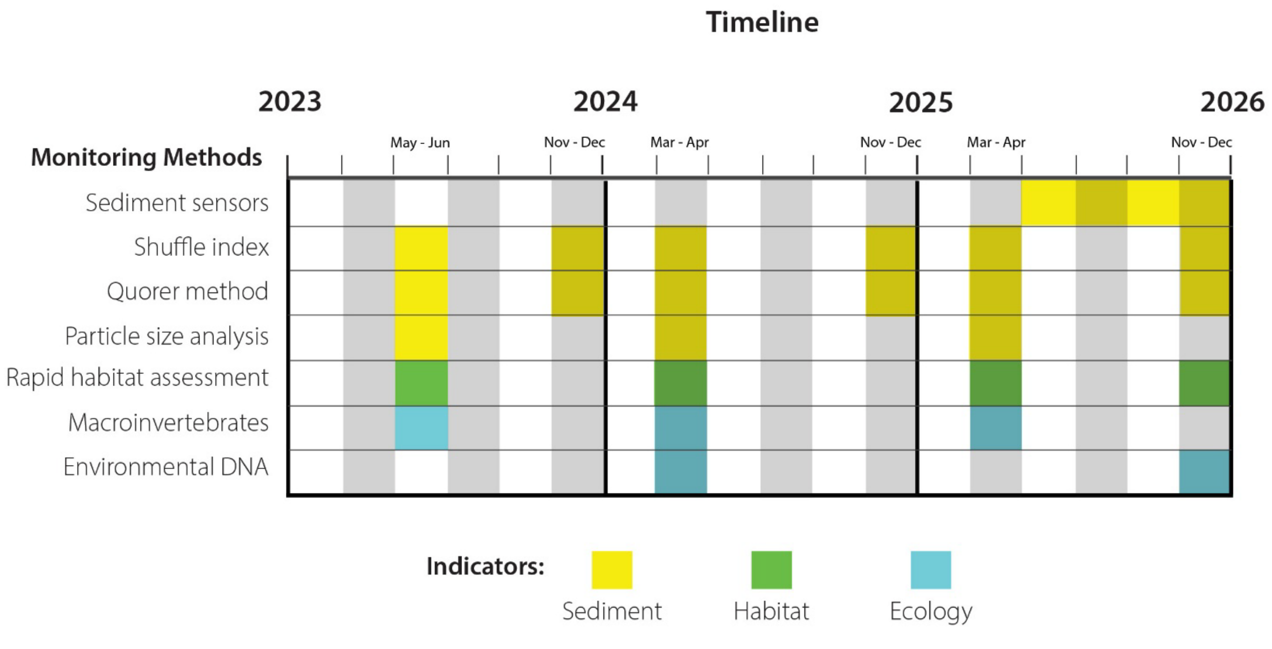


Figure 9. The monitoring methods and the timeline during the MLRP study. Each method represents a different indicator for restoration effectiveness, sediment, habitat and ecology.

3.3.7 Data preparation and statistical analysis

Data was initially collated in Microsoft Excel and inspected with scatterplots and pivot tables to calculate summary statistics and check for anomalies. The turbidity sensor records were cleaned to remove unreliable or unexplained values, but readings were retained unless there was clear evidence that they lay outside natural levels of variation.

Before statistical analyses were run, the data was checked visually and statistically to ensure it complied with model assumptions. Where model assumptions of consistent variance (homogeneity) and normal distribution of residuals were not met, numeric variables were transformed using the natural logarithm (e.g. macroinvertebrate counts used ln(x+1)). Land cover percentages, RHA and shuffle index scores were not transformed and kept on their original form, as they met model assumptions and transformation did not improve model performance. Repeated measurements were treated as nested within each stream/site to account for inherent site level variation.

Scatterplot matrices with Pearson correlation coefficients and histograms were used to inspect distributions and identify strong correlations among predictors (land cover type and RHA scores) and response variables (macroinvertebrate indices, TICI scores and sediment variables); highly correlated predictors were not entered together in the same model to avoid over fitted models and unstable estimates.

For ordered categorical data (shuffle index) ordinal regression and reported McFadden's pseudo-r^2 were used. For all other response variables fitted nested linear mixed models were applied with fixed predictors (reach-scale RHA scores, catchment-scale land cover percentages and treatment group) and stream (9 levels) entered as a random effect to account for repeated measures over time.

Competing models were compared with Akaike Information Criterion, corrected for small sample size (AICc) and Akaike weights to show relative support, model residuals were inspected for stability, and conservative p-values with significance levels at ($P < 0.05$) reported for Restricted Maximum Likelihood (REML) estimates. Pairwise differences among treatment groups were tested with Tukey post hoc comparisons using Kenward-Roger degrees of freedom to improve small sample inference. Marginal (r^2) values were calculated to show the variance explained by fixed effects alone.

Model performance was assessed by reporting the fixed-effect coefficient (β), its standard error (SE), the associated degrees of freedom (df), and the test statistic, which represents the ratio of the coefficient to its standard error (t). All analyses were run in R (version 2025.05.1, build 513).

4 Results

4.1 Rainfall

Monitoring commenced in April 2023 and concluded in December 2025. Rainfall varied substantially during the monitoring period. The annual rainfall records collected from the Auckland Council rain gauge at the Warkworth sewage treatment plant, were 2,420mm in 2023, 1,230 mm in 2024 and 1,583 mm in 2025. July 2025 was the wettest month, during which 316 mm was recorded, and March 2025 was the dryest month with only 26 mm. Results from the Warkworth sewage treatment plant are reported here to provide a general indication for rainfall in the catchment as it is a located centrally in the catchment.

4.2 Catchment characteristics and stream habitat quality

Key findings

- Catchment-scale variables were broad similarity across restored and open pasture streams.
- Forested reference streams had the highest levels of native forest cover and catchment-scale stock exclusion, and these two factors were strongly correlated with each another.
- There were significant differences in stream habitat quality (RHA) between riparian treatments, with the highest scores in forested reference streams, intermediate scores in restored streams, and the lowest scores in open pasture streams. Most habitat parameters (8 of 10) followed this same pattern.
- Catchment-scale and reach-scale characteristics were uncorrelated, enabling both to be included in linear mixed models to separate local riparian effects from broader catchment influences on sediment and ecological condition.

The nine streams monitored in this programme were all small first or second-order streams. The average (± 1 standard deviation) water depth across all streams was 19 cm ± 8.5 cm and the average wetted width was 1.2 m ± 0.6 m. Channel width was narrowest in open pasture streams (0.62 m ± 0.26 m), intermediate in planted streams (0.76 m ± 0.59 m) and greatest in forested streams (1.05 m ± 0.71 m). The average channel area (wetted depth by wetted width) across the nine streams ranged from 0.01 - 0.32 m^2. The size of the catchments above the monitoring sites ranged from 19 - 283 hectares with an average catchment size of 78 ha.

Measured flow rates were low in all streams. All sites recorded rates below 0.1 m^3/s with values ranging from < 0.001 to 0.068 m^3/s. Because these are very small streams with highly localised and discrete flow events, the measurements are unlikely to represent anything greater than the base-flow rates.

The main landcover types represented across the monitored streams were exotic pasture (accounting for between 23% – 81% of the catchment area), mature native forest (< 1% – 77%) and native scrub (<1% – 43%). Other land cover types included wetland, exotic forest and weedy scrub, each contributing less than 2% of the catchment land cover for any one site, with the exception of extoic forest, which accouted for 12.3% in Site 3a and approximatly 2% at Sites 3b and 4. Stream dimensions, reach-scale habitat scores and upstream catchment characteristics for the nine streams are presented in Table 3.

Table 3. Descriptions of the nine streams in the MLRP study, including reach-scale characteristics and catchment-scale characteristics. Average water flow, depth and width of the channel are all measured at water-level, i.e. they represent the water depth and wetted width.

		Reach-scale factors					Catchment-scale factors				
Site ID	Treatment	Average flow (m^3/s)	Average depth (cm)	Average width (m)	Average RHA score (out of 100)	Habitat Quality Rating	Catchment Area (ha)	% Stock Exclusion	% Exotic pasture	% Mature native forest	% Native scrub
1	Open pasture (Unrestored)	0.009	25	0.41	36	Fair	143	44%	32%	22%	43%
2		0.029	27	0.91	35	Fair	110	19%	81%	18%	1%
3a		Not measured	30	Not measured	44	Fair	39	4%	63%	2%	19%
3b		0.009	23	0.53	28	Fair	46	70%	28%	72%	<1%
4	Fenced – Planted native scrub (Restored)	0.068	24	1.35	65	Good	283	12%	87%	<1%	8%
5		0.002	8	0.17	68	Good	16	37%	63%	29%	8%
6		0.004	14	0.75	53	Good	39	32%	68%	26%	6%
7	Fenced – Mature native forest (Reference)	0.001	22	0.77	70	Good	40	48%	46%	54%	<1%
8		0.026	12	0.52	81	Excellent	19	76%	24%	76%	<1%
9		< 0.001	16	1.85	87	Excellent	44	77%	23%	77%	<1%

4.2.1 Catchment-scale characteristics

Land cover and catchment area did not differ significantly between the three riparian treatments ($P > 0.05$) because of variation among individual sites within each treatment group. The median catchment area was largest for open pasture streams (78 ha) and similar for the two fenced treatment groups (39 ha and 40 ha). Median exotic pasture cover was highest for restored streams (68%), intermediate for open pasture streams (47%), and lowest for forested reference streams (24%); ref to Figure 10. Stock exclusion and mature native bush cover were both highest in forested reference streams (76%) and similar, yet relatively low, in the other two treatments (native bush cover = 20% and 26%; stock exclusion = 31% and 32% in open pasture and restored streams, respectively). Native scrub cover in upstream catchments was low across all three treatments with values ranging from 0% in forested reference streams to 10% in open pasture streams, see Figure 10.

Most catchment land-cover variables were strongly correlated with each other (r = 0.55 - 0.97). Stock exclusion and native forest cover were very strongly correlated (r = 0.97) as bush blocks were usually protected from livestock. Native scrub showed weaker correlations with other land cover variables (r = 0.30-0.50), reflecting a more dispersed distribution of scrub across catchments and much less livestock exclusion to protect native scrub from grazing.

Because catchment-scale and reach-scale characteristics were not correlated, both types of variables could be run in linear mixed models. This allowed for an estimation of how much variation in fine sediment and ecological health was explained by adjacent riparian conditions (local, reach-scale) versus upstream catchment conditions (broader, catchment-scale). Refer to section 4.3.3. for more detail.

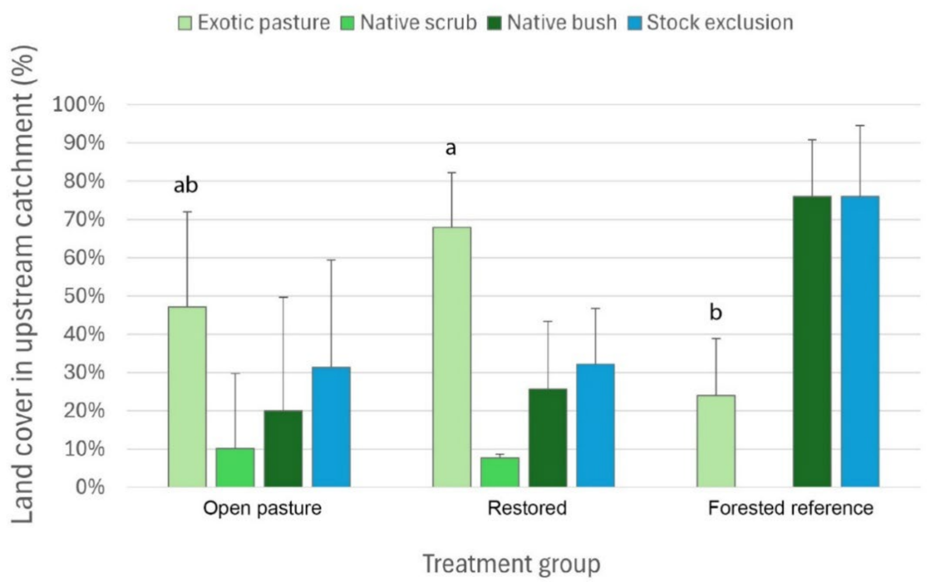


Figure 10. Median percentage land cover and the degree of stock exclusion in the catchments of monitored streams, grouped by treatment.

4.2.2 Reach-scale habitat quality using the Rapid Habitat Assessment (RHA)

Differences in reach-scale habitat were evaluated between the three riparian treatment groups: open pasture, restored streams and forested reference streams. All habitat characteristics, including the total RHA score and all 10 individual parameters which make up the RHA, were compared across the three treatment groups. As expected, the combined median total habitat score (RHA out of 100) was highest for the three forested reference streams (score = 81.5), moderate for the three restored streams (score = 63), and lowest for the three open pasture streams (score = 38); refer to Figure 11a. Median RHA scores for each group indicated 'Excellent' habitat conditions for forested reference streams, 'Good' habitat conditions for restored streams and 'Poor' conditions for open pasture streams. The difference in habitat scores were statistically significant between open pasture streams and forested reference sites (β = -46.3, SE = 5.9, df = 6, t = -7.9, $P < 0.001$) open pasture streams and restored streams (β = 29.3, SE = 5.9, df = 6, t = 5.0, $P = 0.006$). Conversely, overall habitat scores did not differ significantly between the two fenced treatment groups ($P = 0.06$), Figure 11a.

Individual habitat parameters scores (out of 10) for bank vegetation type (with higher scores for native vegetation), stream channel shading and riparian width, were all significantly higher for restored streams and forested reference streams, compared with open pasture streams ($P < 0.05$). Stream flow complexity scores were significantly higher for forested reference streams (score = 8) compared to open pasture streams (score = 5) (β = -3.78, SE = 1.16, df = 6, t = -3.3, $P = 0.04$), Figure 11b.

Invertebrate habitat diversity and fish habitat diversity scores were both significantly higher in restored and forested reference streams, compared to open pasture streams ($P < 0.05$). The abundance of invertebrate habitat was significantly greater in forested reference streams compared to the other two treatment groups ($P < 0.01$). There was no significant difference in the abundance of fish habitat between the three treatment groups, Figure 11c.

There was no significant difference in the deposited sediment or active bank erosion scores between the three treatment groups (Figure 11d). Eight out of 10 reach-scale habitat parameters were strongly correlated to the total RHA score, and to one another across all sites (r = 0.64 - 0.93).

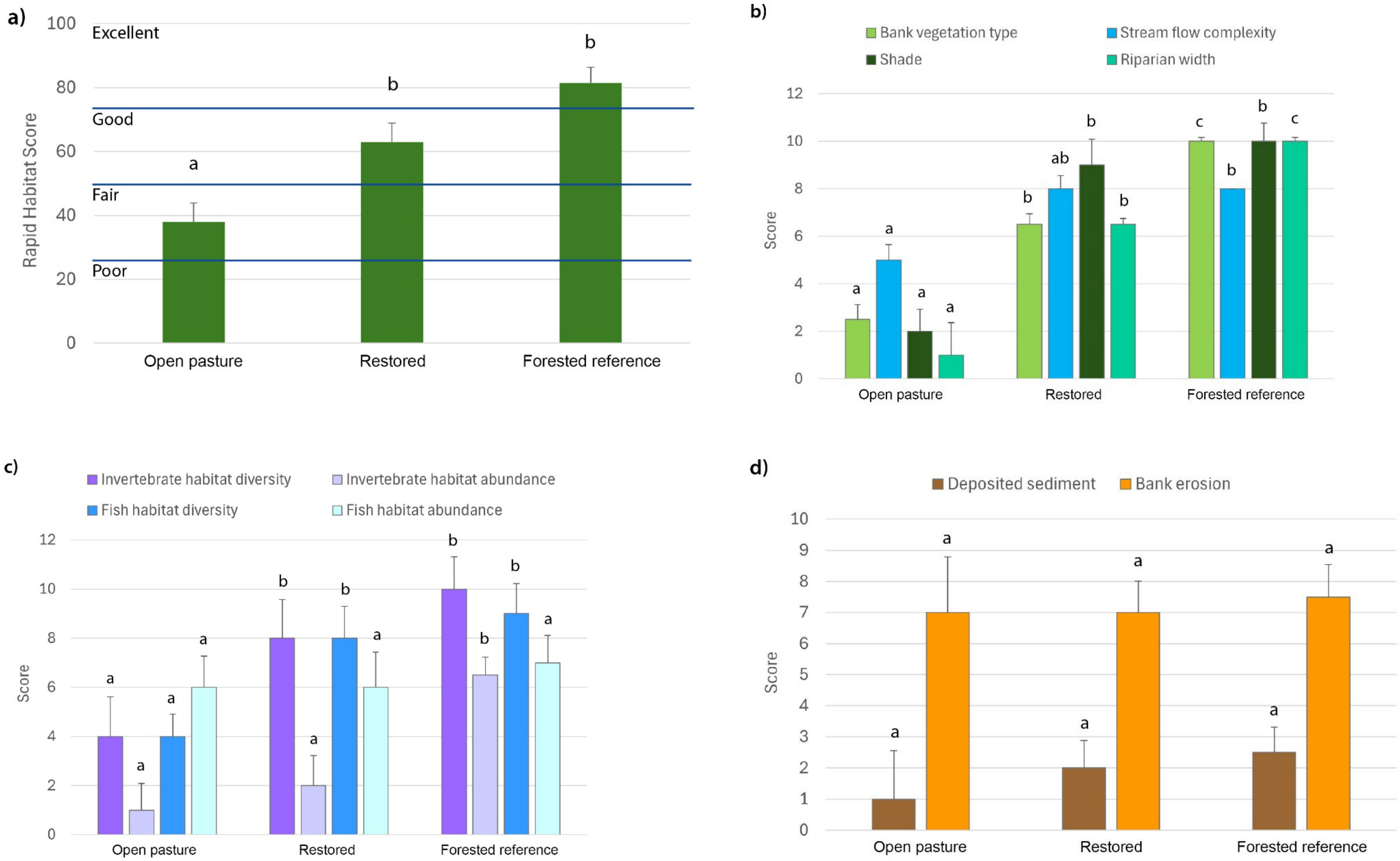


Figure 11. Median rapid habitat scores for the three treatment groups: **a)** total Rapid Habitat Assessment (RHA) score, **b)** bank vegetation, shade, flow complexity and riparian width, **c)** invertebrate and fish habitat variables, and **d)** deposited sediment and bank erosion. Different letters above the bars indicate statistically significant differences between groups; bars sharing the same letter are not significantly different from one another. Error bars show 95% confidence intervals.

4.3 Fine sediment

Key findings

- Fine sediment levels were greatest in open pasture streams and lowest in forested reference streams.
- Results indicate that restored streams had less fine sediment than open pasture streams.
- Fine sediment declined along a gradient of increasing reach-scale habitat quality, forming a clear gradient across the three riparian treatment groups.
- The volume of very fine inorganic mud (< 63 µm) was greatest in open pasture streams (accounting for approximately 67% of inorganic sediment) and lowest in restored streams (29%).

4.3.1 Correlations between sediment variables

All sediment indicators were correlated to each other. The shuffle index showed weak, but significant positive correlations with concentrations of organic sediment (McFadden's $r^2 = 0.04$, $P = 0.04$) and inorganic sediment (McFadden's $r^2 = 0.03$, $P = 0.05$). While concentrations of organic and inorganic particles were very strongly correlated to each other ($r^2 = 0.9$, $P < 0.001$) (see Appendix B, Figure B-1, Figure B-2 and Figure B-3).

4.3.2 Fine sediment under different riparian treatment groups and habitat quality

Shuffle index scores reflected clear differences between treatment groups. Significantly smaller sediment plumes occurred in forested reference streams (median score = 2.5 out of 5) and consistently larger plumes were recorded in open pasture streams (median score = 4.3 out of 5), ($\beta = 4.31$, SE = 1.33, z = 3.24, $P = 0.0035$; Tukey-adjusted). Restored streams had intermediate scores (median score = 3.7 out of 5) that did not differ significantly from the other two groups (**Figure 12**).

Total resuspended sediment concentrations (Quorer method) were highly variable across sites and within treatment groups over the monitoring period. Despite this variability, median concentrations of total fine sediment were highest in open pasture streams (538 g/m³), approximately 32% lower in restored streams (367 g/m³), and approximately 33% lower in forested reference streams (358 g/m³). The median concentration of total inorganic sediment followed a similar pattern, with the highest values in open pasture streams (425 g/m³) and the lowest in forested reference streams (283 g/m³). Based on median values there was substantially more fine inorganic mud (< 63 µm) in open pasture streams (346 g/m³; 67% of inorganic fines) compared to restored streams (106 g/m³; 29%) and forested reference streams (129 g/m³; 37%). These differences were not statistically significant due to high variability within each group (**Figure 13**).

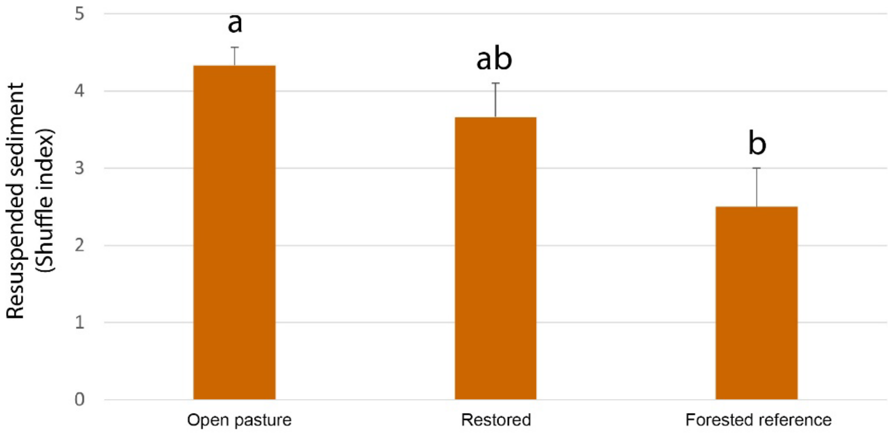


Figure 12. Qualitative estimates of resuspended sediment using the shuffle index, median values plotted against riparian treatment group. Different letters indicate statistically significant different results between groups, error bars represent 95% confidence intervals.

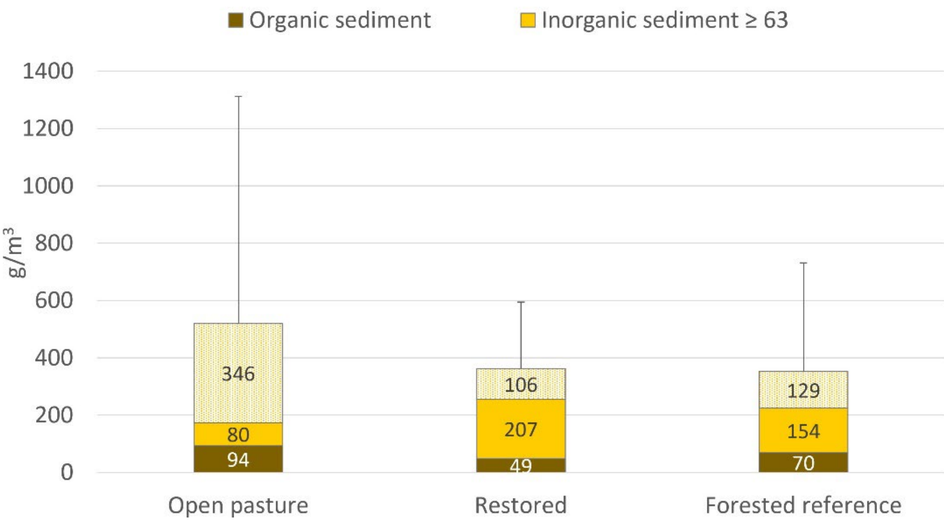


Figure 13. Total resuspended sediment from the Quorer method plotted against treatment group. The different colours represent the median concentration of total organic sediment, total inorganic sediment (≥ 63 µm), and inorganic mud (< 63µm).

Fine sediment was significantly less in streams with higher reach-scale habitat scores. All indicators for resuspended fine sediment – the shuffle index, the concentration of total suspended sediment (organic and inorganic) and inorganic mud (< 63 µm) – declined significantly as habitat scores increased ($P < 0.05$, $r^2 \geq 0.1$; **Figure 14** a-c). Treatment groups were clearly represented along this habitat gradient: open pasture streams (red dots) had the lowest habitat scores and highest sediment values, forested reference streams (blue dots) had the highest habitat scores and lowest sediment values, and restored streams (green dots) were generally intermediate. The strongest relationships were observed for the shuffle index (McFadden's $r^2 = 0.1$, $P < 0.001$) and the total concentration of inorganic sediment ($r^2 = 0.2$, $P = 0.008$).

Total resuspended sediment remained negatively correlated with habitat quality even after reference streams (mature native forest) were removed. This shows that restored streams with higher reach-scale habitat quality still contained significantly less fine sediment than open pasture streams (r^2 = 0.2, P = 0.009; see Figure 14 d)

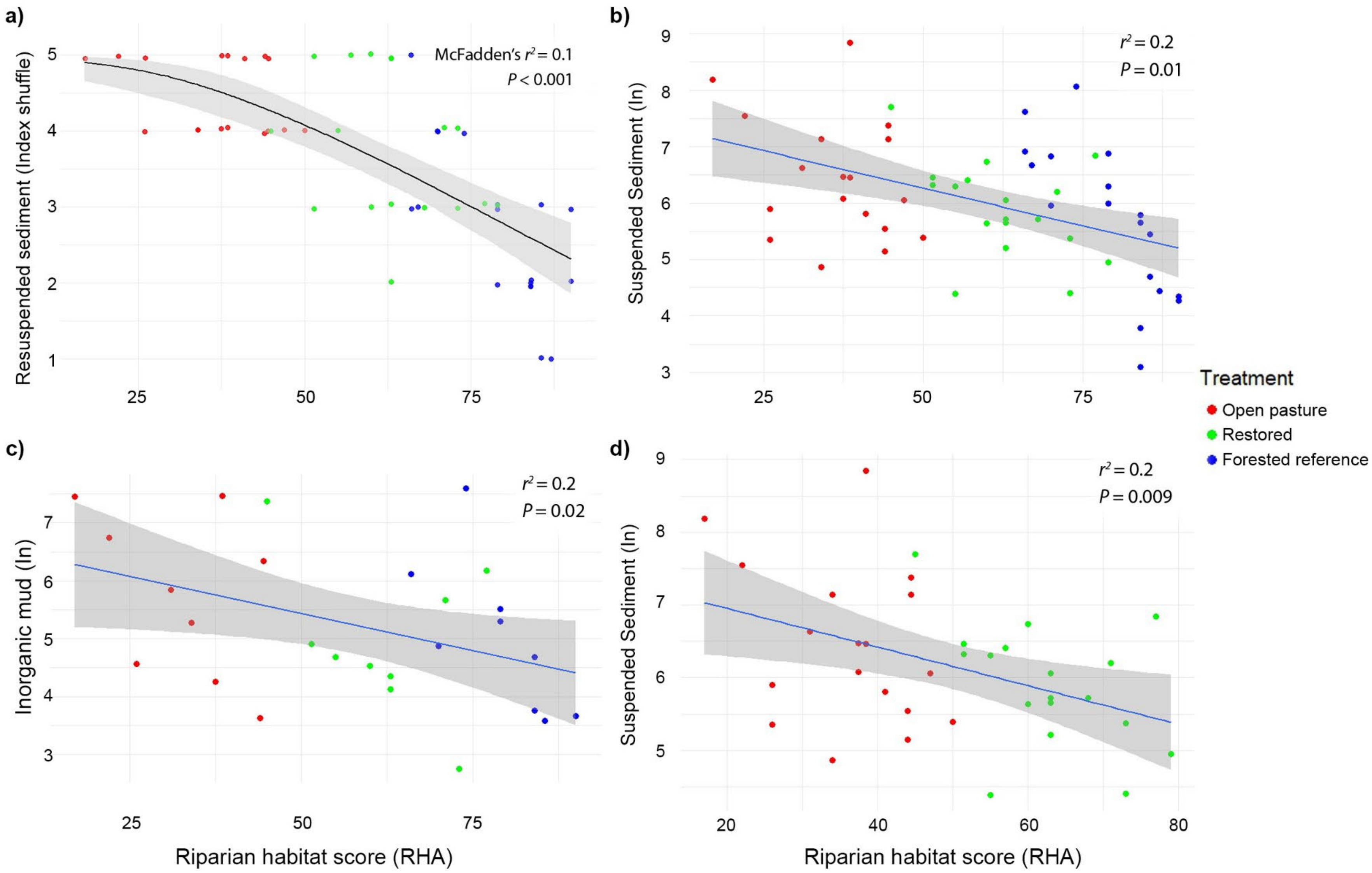


Figure 14. Sediment attributes plotted against reach scale habitat scores (RHA), **a)** shuffle index, **b)** total resuspended (organic + inorganic) sediment, **c)** inorganic mud < 63µm, and **d)** total resuspended sediment excluding forested reference streams.

4.3.3 The influence of reach-scale habitat quality (RHA) versus catchment-scale factors

Comparison of statistical models showed that sediment plume size (shuffle index) was most strongly related to reach-scale habitat quality (RHA) in combination with the degree of upstream stock exclusion in the catchment, i.e. the full model (McFadden's n = 53, w_i = 0.52, r^2 = 0.12, P < 0.001). The next best model was reach-scale habitat (RHA) (McFadden's w_i = 0.46, r^2 = 0.10, P < 0.001), followed by treatment group (McFadden's w_i = 0.018, r^2 = 0.06, P = 0.001) (see Appendix C, Table C-1).

Total resuspended inorganic (SIS; g/m^3) and organic sediment (SOS; g/m^3) were most strongly associated with RHA scores (SIS: n = 54, w_i = 0.39, r^2 = 0.23, P = 0.008; SOS: n = 54, w_i = 0.57, r^2 to 0.20, P = 0.004). The next most supported models included the degree of upstream stock exclusion in the catchment (SIS: n = 54, w_i = 0.09, r^2 = 0.14, P = 0.03; SOS: n = 54, w_i = 0.06, r^2 to 0.11, P = 0.04). RHA was also the only significant predictor of inorganic mud (n = 27, w_i = 0.60, r^2 = 0.18, P = 0.02). Full model outputs are provided in Appendix C, Table C-2.

4.3.4 Differences in continuously monitored suspended sediment between treatment groups

Continuous monitoring (every 15-minutes) of total suspended solids (TSS) in the three sampled streams (between 14 May and 31 December 2025) indicated that average estimated sediment concentrations were highest in the open pasture stream (44 g/m^3), intermediate in restored streams (39 g/m^3), and lowest in forested reference streams (37 g/m^3). Comparative results of median resuspended TSS derived using the Quorer method (SAM4) for the same three streams exhibited a corresponding pattern to the sensor readings, albeit with much higher concentrations. Median resuspended TSS was greatest in the open pasture stream (956 g/m^3), intermediate in the restored stream (294 g/m^3) and lowest in the forested reference stream (93 g/m^3) (Figure 15).

The sensor measurements for TSS exhibited wide ranges of values, both across and within sites. The largest range occurred in the restored stream (0.4 - 5,468 g/m^3), followed by the open pasture stream (0.1 - 2,244 g/m^3) and the forested reference stream (0.1 - 1,478 g/m^3). Suspended sediment concentrations generally increased following heavy rainfall; this response was more pronounced in the two fenced streams, whereas the open-pasture stream expressed higher concentrations during dry periods with observations noted during monitoring this may have coincided with livestock grazing upstream of the turbidity sensor. Continuous suspended sediment readings plotted against rainfall are provided in Appendix D, Figures D-1 to D-3.

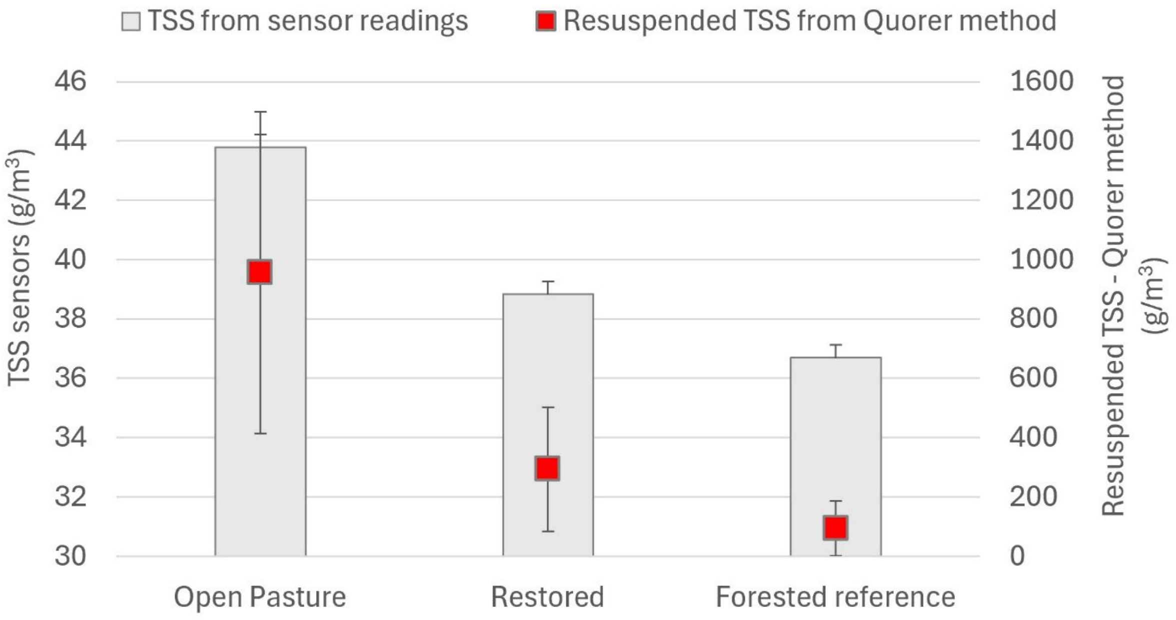


Figure 15. Average estimated total suspended solids measured in three streams using turbidity sensors and resuspended median TSS collected in the same sites using the Quorer method. Error bars represent 95 per cent confidence intervals.

4.4 Ecological health

Key findings

- Macroinvertebrate-derived ecological health indices (MCI, QMCI, EPT richness and EPT abundance) and eDNA derived TICI scores were highest in forested reference streams and lowest in open pasture streams.
- Restored streams had slightly higher scores for QMCI and EPT abundance compared to open pasture streams, although differences were not statistically significant.
- Environmental drivers differ for each ecological health index; treatment group explained the highest proportion of variation in QMCI scores and EPT abundance, reach-scale habitat quality (RHA) explained the highest degree of variation in EPT richness and TICI scores, and the combined influence of catchment-scale native bush cover and RHA explained the greatest variation in MCI scores.
- Fish IBI scores indicated high to moderate integrity of the fish communities across treatment groups, corresponding to NSP-FM A band (four sites) and B band (five sites).
- Nationally and regionally rare native fish species were detected across multiple sites, predominantly in forested reference streams and restored streams. No introduced fish species were detected across sites.
- Kākahi (freshwater mussel) and kōura (freshwater crayfish) were also detected at several locations, predominantly in forested reference streams and restored streams.

4.4.1 Aquatic macroinvertebrates

Ecological health indices derived from aquatic macroinvertebrates showed broadly consistent response patterns across the three riparian treatment groups. MCI, QMCI, EPT abundance and EPT richness were all highest in forested reference streams and lowest in open pasture streams. However, index scores varied considerably among the three restored streams, and QMCI together with the two EPT metrics showed clearer differentiation between the three treatment groups than MCI. Macroinvertebrate results for each site are provied in Table E-1 in Appendix E.

Median MCI and QMCI scores were highest in forested reference streams (MCI = 100; QMCI = 5.5) and lower in restored streams (MCI = 77; QMCI = 3.9) and open pasture streams (MCI = 76; QMCI = 3.1). In forested reference streams, MCI scores were classed as 'Good', based on the quality classes in Stark and Maxted (2007), corresponding to NPS-FM[1] attribute band C. In contrast, MCI scores in both planted and open pasture streams were classed as 'Poor', corresponding to band D (Stark & Maxted, 2007; NPS-FM, 2020). QMCI scores showed a similar pattern, with forested reference streams classed as 'Good', corresponding to band B, and both restored and open pasture streams were classed as 'Poor', corresponding to band D (Stark & Maxted, 2007; NPS-FM, 2020).

The difference in MCI scores was significant between the forested reference streams and open pasture streams (β = -0.33, SE = 0.10, df = 6, P = 0.03; Tukey-adjusted), but were not significant between forested and planted streams or between planted and open pasture streams; Figure 16. QMCI scores differed significantly between mature native forested streams and both open pasture streams (β = -0.47, SE = 0.08, df = 6, P = 0.035; Tukey-adjusted) and planted with native scrub streams (β = -0.33, SE = 0.08, df = 6, P = 0.02; Tukey-adjusted). MCI and QMCI did not differ significantly between planted streams and open pasture streams (Figure 16).

[1] National Policy for Freshwater Management (2020)

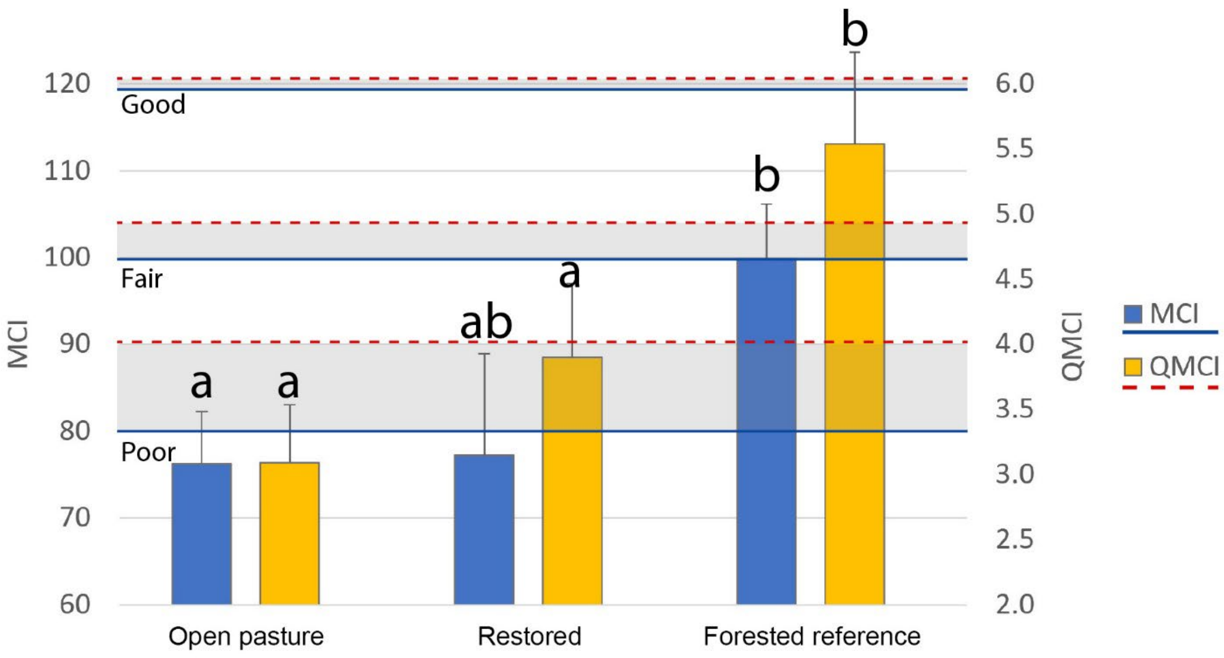


Figure 16. Median MCI and QMCI plotted against treatment group. Different letters reflect statistical differences between groups, error bars are 95 per cent confidence intervals. **Quality class boundaries,** as defined by Stark & Maxted (2007), are indicated by the blue solid lines for MCI and the red dashed lines for QMCI. The grey shading shows the intermediary space between the MCI and QMCI class boundaries.

There were clear differences in the occurrence and abundance of mayflies, stoneflies and caddisflies across the three treatment groups. Forested streams were the only sites where stoneflies were recorded (a single species, *Acroperla*). Mayflies were collected only in forested and planted streams, with *Zephlebia* being the most abundant taxon. Planted streams supported a greater number and abundance of caddisflies than open pasture streams. These patterns were reflected in both EPT richness and EPT abundance scores. Forested streams had higher median EPT richness (27%) and abundance (35%) than planted streams (EPT richness = 18%; abundance = 3%) and open pasture streams (EPT richness = 10%; abundance = 1%). Although EPT richness did not differ significantly among riparian treatment groups, EPT abundance was significantly greater in forested streams than planted streams (β = -1.62, SE = 0.48, df = 6, P = 0.035; Tukey adjusted) and open pasture streams (β = -2.66, SE = 0.48, df = 6, P = 0.004; Tukey adjusted) (**Figure 17**).

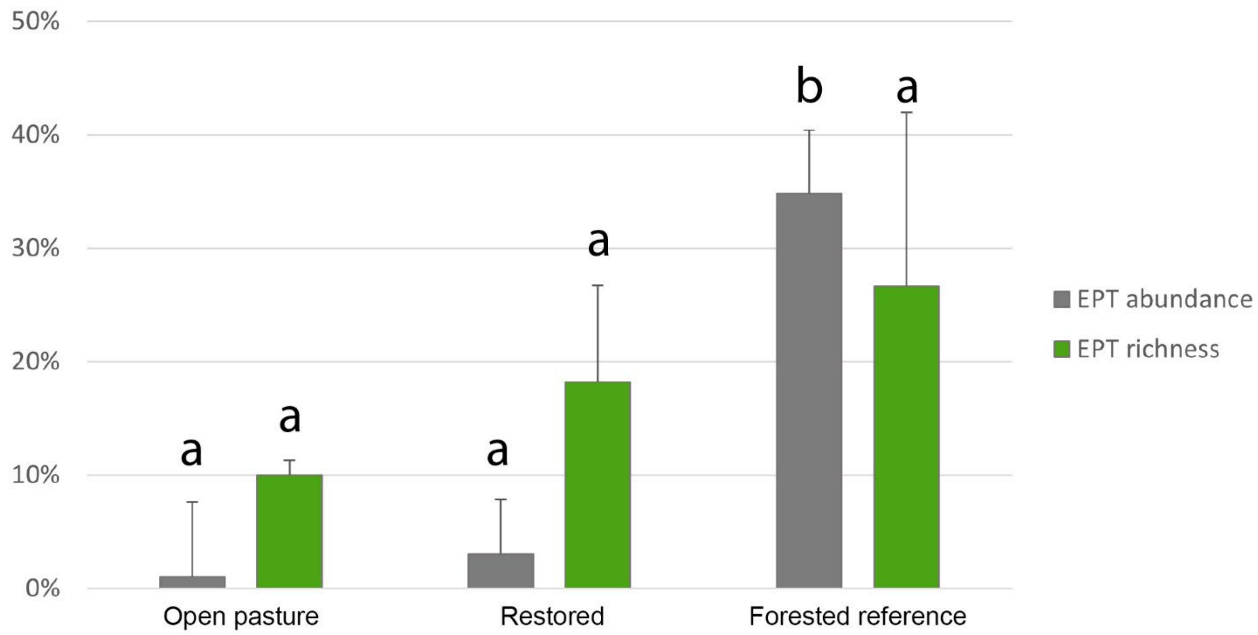


Figure 17. Median EPT abundance and EPT richness plotted against treatment group. Different letters reflect statistical differences between treatment groups; error bars represent 95 per cent confidence intervals.

All four invertebrate indices (MCI, QMCI, EPT richness, and EPT abundance) were positively correlated to reach-scale habitat scores (RHA). There was a clear gradient from forested streams to open pasture streams, with open pasture streams being clustered towards the lower end of the scale, restored streams having intermediate scores and forested reference streams have the highest scores. The abundance of pollution sensitive EPT species had the strongest corrleation to RHA (r^2 = 0.43, P = 0.003) and MCI had the weakest correlation (r^2 = 0.25, P = 0.03) (Figure 18 a-d).

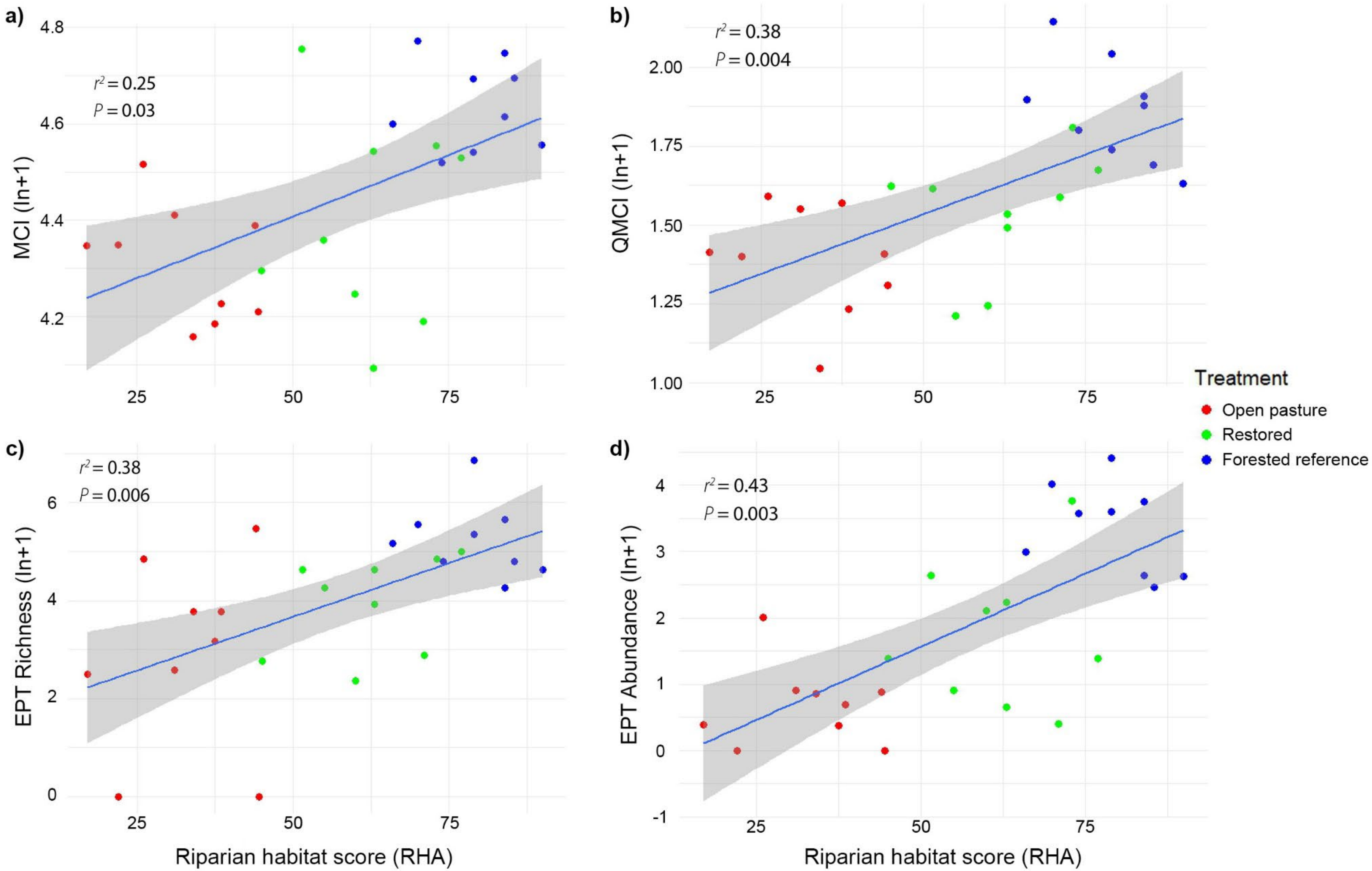


Figure 18. Macroinvertebrate indices plotted against reach-scale habitat scores (RHA), a) MCI, b) QMCI, c) EPT species richness and d) EPT abundance. Grey bands represent 95 per confidence intervals.

A comparison of linear mixed models assessing the influence of reach-scale and catchment-scale factors on macroinvertebrate indices showed that MCI was significantly related to all catchment-scale factors (% native bush cover, % stock exclusion and catchment size) and reach-scale habitat quality (RHA). The best performing model combined upstream native bush cover and reach-scale habitat quality (RHA): the full model (n = 36, w_i = 0.48, r^2 = 0.60, $P \leq 0.05$), followed by upstream native bush cover alone (n = 36, w_i = 0.28, r^2 = 0.50, $P < 0.001$), catchment size (n = 36, w_i = 0.12, r^2 = 0.49, P = 0.002), upstream stock exclusion (n = 36, w_i = 0.10, r^2 = 0.43, P = 0.001), treatment group (n = 36, w_i = 0.03, r^2 = 0.50, P = 0.002) and lastly RHA (n = 36, w_i = 0.01, r^2 = 0.25, P = 0.03).

QMCI scores were also significantly related to multiple catchment-scale variables and reach-scale RHA. The best performing model was treatment group (n = 36, w_i = 0.74, r^2 = 0.58, $P < 0.001$), followed by the full model combining upstream native bush cover and RHA (n = 36, w_i = 0.20, r^2 = 0.54, $P \leq 0.01$), then RHA (n = 36, w_i = 0.03, r^2 = 0.38, P = 0.004), upstream native bush cover (n = 36, w_i = 0.03, r^2 = 0.35, P = 0.004) and then upstream stock exclusion (n = 36, w_i = 0.01, r^2 = 0.25, P = 0.02).

The variation in EPT richness was most strongly related to reach-scale RHA (n = 36, w_i = 0.42, r^2 = 0.38, P = 0.006), followed by treatment group (n = 36, w_i = 0.04, r^2 = 0.38, P = 0.01), then upstream stock exclusion (n = 36, w_i = 0.04, r^2 = 0.22, P = 0.03), and lastly catchment size (n = 36, w_i = 0.05, r^2 = 0.27, $P < 0.05$). While EPT abundance was best explained by treatment group (n = 36, w_i = 0.67, r^2 = 0.68, $P < 0.001$), followed by the full model combining upstream native bush cover and RHA (n = 36, w_i = 0.26, r^2 = 0.63, $P \leq 0.01$), then RHA (n = 36, w_i = 0.04, r^2 = 0.43, P = 0.003), then upstream native bush cover (n = 36, w_i = 0.01, r^2 = 0.28, P = 0.01), catchment size (n = 36, w_i = 0.01, r^2 = 0.32, P = 0.04) and lastly upstream stock exclusion (n = 36, w_i = 0.01, r^2 = 0.2, P = 0.01) (full model outputs are provied in Appendix D, Table C-3).

In summary, of all environmental predictors tested in the set of models, treatment was the strongest predictor for QMCI and EPT abundance. Reach-scale RHA was the strongest predictor for EPT richness, and the full model combining RHA and upstream native bush cover peformed the best for MCI (Appendix D, Table C-3).

4.4.2 Biodiversity from eDNA (environmental DNA)

The nine streams were sampled for eDNA twice during the monitoring period, between March and May 2024 and between November and December 2025. The following sections highlight biodiversity integrity at the nine sites across the three treatment groups, based on results from TICI calculations and the diversity of fish and target invertebrate species detected.

4.4.2.1 Taxonomically independent community index score (TICI)

The median TICI scores were highest in forested reference streams (97.9), intermediate in restored streams (93.1) and lowest in open pasture streams (89.7). These scores indicated ‘Poor’ ecological health in open pasture streams and 'Average' health in planted streams (see **Figure 19** and Appendix A, Table A-4).

Although TICI scores did not differ significantly between treatment groups following pairwise comparisons with Tukey adjustment, this likely reflects the small sample size assessed (two repeated samples across nine streams). The largest contrast occurred between forested reference streams and open pasture streams.

TICI scores were significantly positively correlated with reach-scale RHA (β = 0.16, d.f. = 6.7, r^2 = 0.27, P = 0.03) (**Figure 20**).

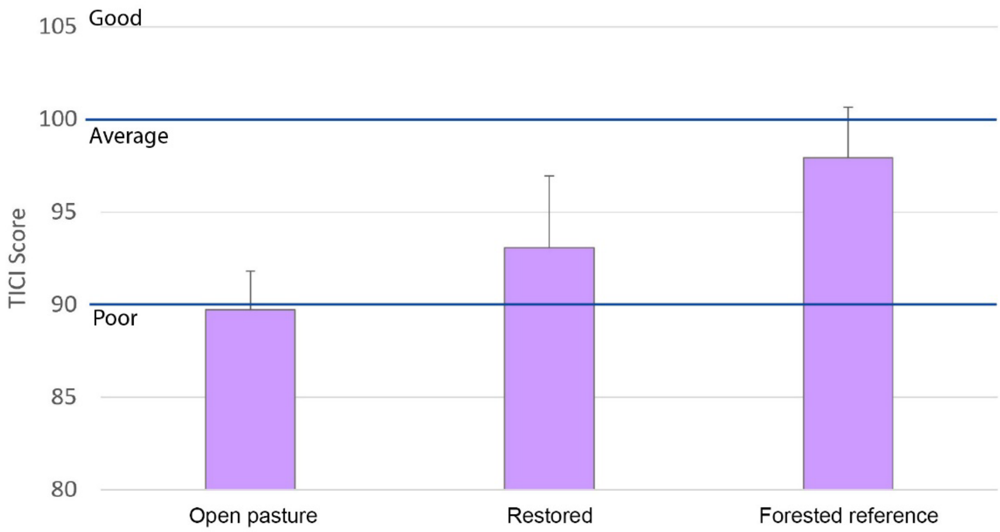


Figure 19. Median TICI scores plotted against treatment group. Error bars represent 95 per cent confidence intervals.

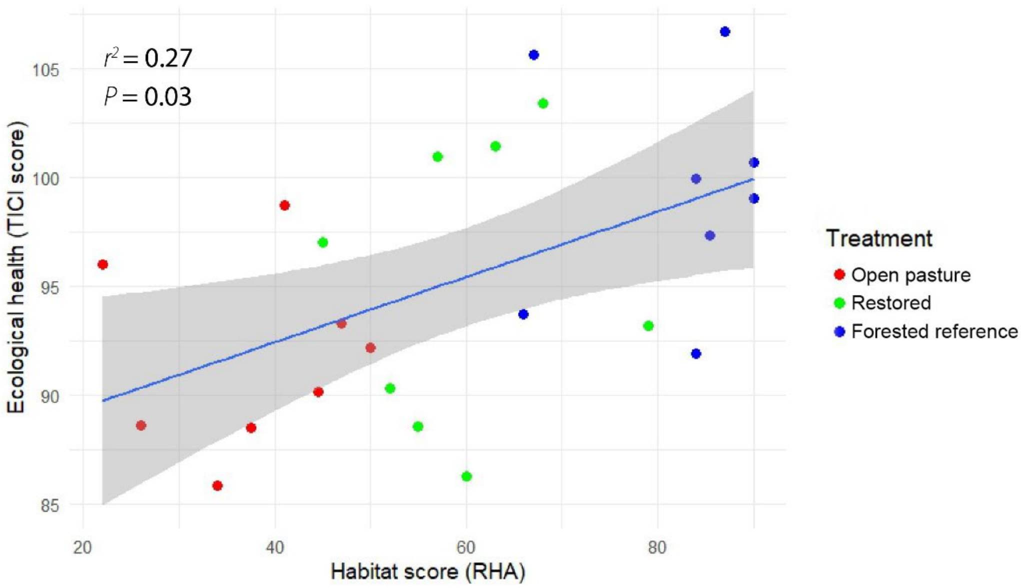


Figure 20. TICI score plotted against habitat score (RHA). Grey bands represent 95 per cent confidence intervals.

4.4.2.2 Kākahi and kōura – freshwater mussels and crayfish

Kākahi and kōura were detected using eDNA in both 2024 and 2025. These species are endemic to Aotearoa New Zealand and are taonga that perform important ecosystem functions, and they are sensitive to poor water quality, making them useful indicators of ecological health.

One kākahi species, *E. menziesii,* was detected at two reference sites (7 and 9), and the prevalence index calculated for this species at these sites was low to very low (Melchior & Baker, 2023). At the time of writing this species was declining and considered at risk (Grainger, et al., 2018). Kōura were detected in six streams, including one open pasture site (stream 3), two restored sites (streams 5 and 6) and all three forested reference sites (streams 7, 8 and 9). The prevalence index for kōura at the six sites ranged from trace levels to low, and at the time of writing they were listed as not threatened (Grainger, et al., 2018).

4.4.2.3 Freshwater fish

Seven native freshwater fish species were detected across the nine streams. Median species richness by stream ranged from three to five species. Streams 1 and 4 recorded the greatest number of fish species (six), while streams 2 and 5 recorded the lowest number of fish species (three). Total species richness was similar across the three treatment groups but was highest in open pasture streams (13), followed by forested reference streams (12) and then restored streams (10).

Calculated F-IBI scores returned positive results across all nine streams. Four streams (1, 3, 4 and 8), representing at least one from each treatment group, had F-IBI scores greater than or equal to 34, falling within A band – when graded against NPS-FM attribute bands – indicating high fish community integrity with minimal impact to habitat and migratory access. The remaining five sites had F-IBI scores between 28 to 30, falling within B band, indicating moderate integrity with potential for some habitat or migratory access impacts (see Appendix A, Table A-5).

Species detected over the nine sites (from most to least frequent) were shortfin eel (*Anguilla australis*), longfin eel (*A. dieffenbachii*), banded kōkopu (*Galaxias fasciatus*), common bully (*Gobiomorphus cotidianus*), redfin bully (*G. huttoni*), giant kōkopu (*G. argenteus*) and giant bully (*G. huttoni*); refer to Figure 21. Notably, īnanga (*G. maculatus*) were not detected, suggesting the presence of partial fish barriers downstream of all sites. No introduced fish species were detected at any site either. Of the species detected, three are recognised as nationally at risk (Dunn, et al., 2025), four as regionally threatened or at risk (Bloxham, et al., 2023), and two are not considered regionally threatened (Bloxham, et al., 2023; Dunn, et al., 2025).

Calculated prevalence indexes for most fish species were in the 'Moderate' to 'Low' range. This represents a relatively strong level of detection given that prevalence indexes in this study were based on three replicate samples, rather than the six replicates on which the index categories were developed from (Melchior & Baker, 2023). Giant kōkopu and giant bully, however, were detected at much lower levels. Both species returned a 'Very low' prevalence index in stream 1, and giant kōkopu were detected at 'Trace' levels in stream 8, reflecting only low DNA reads in a single replicate sample at each site.

A summary of the fish species detected at each site are shown in Figure 21 and their conservation status and proportional representation of their prevalence index – indicating the strength of that detection across the sites detected – are shown in Table 4.

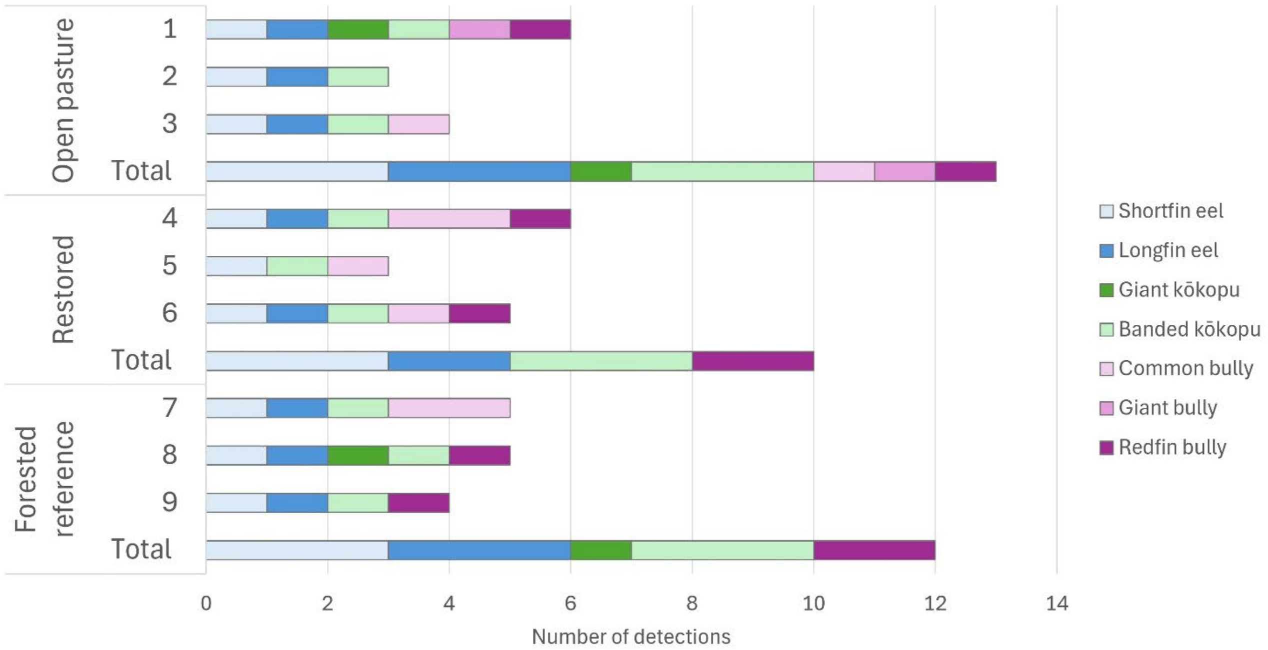


Figure 21. Freshwater fish species detected using eDNA analysis in each of the nine streams alongside the total number of positive detections (indicating species presence) per treatment group.

Table 4. The regional and national conservation status of freshwater fish detected in the nine Mahurangi streams and the number of streams each was detected in. Regional conservation status is defined in Bloxham, et al. (2023) and national conservation status is defined by Dunn et al. (2025). The proportional representation of the different prevalence indexes (Melchior & Baker, 2023) are presented for each species.

Common name	Māori name	Scientific name	Conservation status		Streams detected	Proportional representation of prevalence indexes			
			Regional (2023)	National (2025)		Moderate	Low	Very low	Trace
Shortfin eel	Tuna, Hau	*Anguilla australis*	Regionally Not Threatened	Not Threatened	9	46%	37%	12%	6%
Longfin eel	Tuna	*Anguilla dieffenbachii*	At Risk – Regionally Declining	At Risk – Declining	8	19%	75%	0%	6%
Banded kōkopu	Kōkopu, Kōkopu taiwhara	*Galaxias fasciatus*	Regionally Not Threatened	Not Threatened	9	56%	44%	0%	0%
Common bully	Tīpokopoko	*Gobiomorphus cotidianus*	Regionally Not Threatened	Not Threatened	4	25%	75%	0%	0%
Redfin bully	Kōkopu urutira whero	*Gobiomorphus huttoni*	At Risk – Regionally Declining	Not Threatened	5	13%	58%	29%	0%
Giant kōkopu	Kōkopu, Tītarakura	*Galaxias argenteus*	Threatened – Regionally Critical	At Risk – Declining	2	0%	0%	50%	50%
Giant bully	Pōrohe, Paraki	*Gobiomorphus gobioides*	Threatened – Regionally Vulnerable	At Risk – Naturally Uncommon	1	0%	0%	100%	0%

5 Discussion

The aim of this study was to determine whether riparian planting and stock exclusion in restored headwater streams was associated with measurably lower fine sediment and greater ecological health, compared to unrestored open pasture streams. Because the MLRP builds on work that began in the catchment some 20 years ago, and the restoration process takes time, this assessment has been inferred by monitoring a subset of streams that had been fenced and planted prior to the beginning of the programme, as projects completed over the past six years are still establishing.

The results from nine monitoring sites showed that riparian habitat quality was inversely related to sediment measurements and positively associated with ecological health indicators. Reach-scale habitat quality was significantly greater in restored streams compared with open pasture streams. These findings indicate that riparian management is contributing to measurable reductions in fine sediment and improvements in stream health in the Mahurangi catchment. In addition, the influence of wider catchment-scale characteristics, particularly the degree of forest cover and integrated land management practices, cannot be overlooked. Sites with less catchment-scale forest cover and grazed riparian margins had higher levels of fine sediment and poorer ecological conditions. The diagram below illustrates the key findings of this study (**Figure 22**).

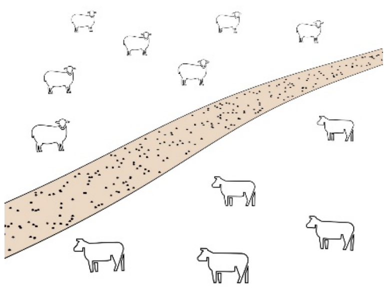
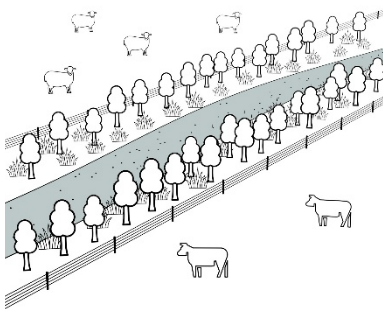
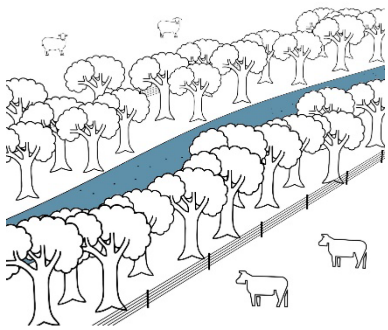
Open pasture	Restored	Forested reference
Low riparian habitat quality and low catchment-scale forest cover	Moderate riparian habitat quality and low catchment-scale forest cover	High riparian habitat quality and catchment-scale forest cover
		
Instream conditions: High levels of fine sediment and poor ecological health	Instream conditions: Moderate levels of fine sediment, ecological health is impacted but improving	Instream conditions: Low levels of fine sediment and the greater ecological health scores

Figure 22. Illustrated summary of findings from the MLRP stream monitoring programme.

5.1 Riparian restoration improves stream health

Riparian restoration is widely applied in New Zealand to improve water quality and stream ecological health (McKergow, et al., 2016). Potential benefits operate through multiple pathways. The riparian zone is a critical interface between the land and receiving environments: it can be a potential source for contaminants, but if protected it serves as a buffer against physical disturbances (preparatory processes), caused by livestock (Hughes, 2016). Riparian vegetation also supplies leaf litter and woody debris that provide food and habitat for aquatic fish and invertebrates. In addition to riparian protection, increasing forest cover in a catchment, through partial or full afforestation of erosion-prone slopes and gully systems, can improve hydrological functions and flood resilience in streams by enhancing infiltration and evapotranspiration rates, thereby helping to moderate both peak and low flow events (Brown, et al., 2005).

Both riparian planting and catchment landcover (e.g. permanent forest, pasture, and impervious surfaces associated with urban development) strongly influence catchment hydrology and soil disturbance regimes, with land use change often exacerbating natural (background) erosion rates. A recent analysis covering 657 catchments, examining interactions between forest loss, rainfall, and short-term water inputs to rivers via "leaky" watersheds, found that for every 1% reduction in forest cover across a catchment, the proportion of "young" rain-derived water[2] entering the river network increased by approximately 0.17%. This can have profound effects on peak flow and flood events, and was reported to be most pronounced in catchments with shallow groundwater, where rapid connections to surface flow accelerate runoff and shallow subsurface lateral flow (Qiu, et al., 2026). The authors also highlighted the role of forest edge density (associated with an arrangement of smaller forest fragments). In catchments with low overall forest cover (< 40%), increased forest edge density significantly reduced young water inputs by facilitating evapotranspiration (Qiu, et al., 2026). This emphasises the importance of integrated catchment management strategies including riparian planting, which will increase forest edge density over time and can serve to mediate peak flow events.

Similar catchment-scale results have been reported in New Zealand. In the upper Waipori catchment in Otago, 34 years of hydrological monitoring showed that partial afforestation of small catchments in exotic pine (*P. radiata*) reduced average peak flows by 78% for small rainfall events and by 37% during larger storms (Fahey & Payne, 2017). In the Mangaotama catchment in the Waikato, 23 years of monitoring following partial afforestation (62%) with *P. radiata* showed reductions of approximately 50% in peak flows and around 30% in large storm induced flows (Hughes, et al., 2020). In the Mahurangi catchment, Te Muri-o-Tarariki stream's peak discharge rates decreased from 313 L/s (pre-restoration) to 135 L/s (several years after fencing and native planting) representing a 57% reduction in peak flows (Tsyplenkov & Neverman, 2025).

The wide variation in sediment values measured during the MLRP study locations reflects both the natural variability of fine sediment (Clapcott, et al., 2011; Davis, 2023) and inherent site-specific differences (e.g. grazing intensity, riparian vegetation establishment and understorey intactness). For

[2] Young water refers to the portion of streamflow made up of rainfall that entered the catchment within the past 2 to 3 months.

example, site 6 (restored) and site 7 (forested reference) had lower RHA scores and higher fine sediment concentrations than the other streams in the same riparian treatment group. Observations made during monitoring suggested that these differences were primarily related to local soil characteristics and ground cover conditions which contributed to higher rates of sediment mobilisation and reduced bank stability. Consequently, riparian treatment group was not necessarily the strongest predictor of sediment or ecological health conditions. Instead, reach scale habitat quality (shaped by a combination of vegetation type, riparian width, shading, understorey density, flow complexity, and grazing intensity) was a reliable predictor of sediment and ecological conditions. Similar findings have been reported elsewhere. Riparian fencing in the Waikakahi stream in South Canterbury was found to reduce deposited fine sediment cover by approximately 20% when at least 300 meters of fencing was established upstream with a minimum five-meter setback along both stream margins (Holmes, et al., 2016).

A large proportion of annual sediment yield occurs during high rainfall events, when soils are near or at saturation. Under these conditions, runoff and groundwater inputs are highest, stream flow increases, and the additional energy mobilises and transports fine sediment downstream. Reducing the impact of these peak events can disproportionately reduce the quantity of sediment mobilised (Basher, et al., 2011; Temple & Parsonson, 2014).

Sediment variables are not only influenced by weather events, but also by changes in stream morphology, which can be altered through planting. In some cases, sediment inputs may temporarily increase following riparian planting as stream channel morphology adjusts and widens in response to increased shading and the transition in groundcover vegetation from pasture to forest understorey. Stream channel width has been found to be greater in forested streams compared to open pasture streams in other studies from New Zealand. For example, a paired stream study of second order streams reported significantly greater channel width in native forest catchments compared with pasture catchments (Davies-Colley, 1997). Similar results were documented during the MLRP study, where open pasture streams had the narrowest average channel width (0.62 m) and forested reference streams had the greatest average width (1.05 m). This difference is linked to vegetation type and cover; with exotic pasture grasses providing a more continuous groundcover and interwoven shallow root structure that binds stream banks and limits gradual natural erosion, resulting in narrower channels with steeper banks. Steep banks are more prone to erosion and mass failure events that release pulses of sediment. Conversely, when pastured streams are planted, the shade created by establishing shrubs and trees causes pasture dieback and subsequently increased erosion as the stream channel begins to adjust accordingly. These morphological changes may temporarily increase sediment delivery before longer-term stabilisation and ecological benefits emerge. These processes will differ in each system and over the longer-term, it is expected that riparian planting reduces fine sediment inputs by reinforcing stream banks through a complexity of root structures, canopy cover and native groundcover. Studies that demonstrate this, however, are still lacking.

When livestock access streams, erosion and sedimentation rates generally increase as a result of animals (particularly cattle, deer, horses and pigs) physically damaging banks and stream beds. This leads to the mobilisation of sediment alongside other contaminants like *E. coli* and nutrients (Davies-

Colley, et al., 2004; Trimble & Mendel, 1995). A lack of woody riparian vegetation increases the risk of erosion by removing the integrated root structures that stabilise soil in forested catchments. This is especially important on steeper slopes subject to higher land use classes (>25° LUC 6e+) where they are most vulnerable to amplified soil loss and erosion processes (Quinn & Stroud, 2002). Conversely, establishing native riparian vegetation reduces sediment mobilisation and improves ecological health by dissipating rainfall, increasing infiltration, improving bank stability over time via root tensile strength (Simon, et al., 2023), and enhancing habitat complexity, thermal regulation and food supply for native fauna (McKergow, et al., 2016).

There are few studies that demonstrate the consistent effectiveness of riparian restoration because outcomes vary in response to differences in riparian management (such as setback distance, vegetation maintenance and addressing point-source inputs of contaminants) and stream conditions. In general, riparian restoration has been found to reduce water temperature, improve visual clarity and enhance bank stability with associated improvements in macroinvertebrate communities linked to cooler water temperatures following canopy closure (Parkyn, et al., 2003; Quinn, et al., 2009). The results presented in this report align with other studies showing that stream habitat quality, upstream forest cover and catchment size influence aquatic macroinvertebrate indices like MCI, QMCI and pollution sensitive EPT taxa (Davis, et al., 2022; Reid, et al., 2010).

Macroinvertebrate indicators for ecological health have been documented as responding positively to riparian and catchment restoration within relatively short timeframes. An example of this are the ecological gains observed six years after the implementation of the Integrated Catchment Management project at the Whatawhata research station in Waikato (Quinn, et al., 2009). In the MLRP study, where streams had been retired for between 6 to 10 years, restored streams generally had a greater abundance and richness of pollution sensitive mayfly and caddisfly taxa than in open pasture streams. In addition, MCI scores were classed as either ‘Fair’ in two out of three restored streams, corresponding to NPS-FM band C (the same band as reference streams), while open pasture streams were classed as either ‘Fair’ or ‘Poor’, all of these sites fell into the NPS-FM band D, indicating severe pollution (NPS-FM, 2020). QMCI scores indicated ‘Fair’ conditions in two out of three restored streams (sites 5 and 6) and ‘Poor’ conditions in open pastures streams. TICI scores calculated for all sampled streams indicated ‘Average’ ecological health in most restored streams and ‘Poor’ health in open pasture streams. For comparison, recent State of the Environment (SOE) monitoring results from eight streams across the Mahurangi catchment showed that MCI classes range from ‘Excellent’ to ‘Poor’ corresponding to NPS-FM attribute bands A to D with Mahurangi River having the highest scores and Duck creek having the lowest, while QMCI scores were generally lower, with most sites classed as ‘Fair’ or ‘Poor’, corresponding to attribute bands C and D (Surrey & Storey, 2025).

The relatively low ecological health indices recorded during the MLRP could be related to the extreme weather events experienced shortly before and during the monitoring period. Two big cyclones hit the Auckland Region in early 2023 (Cyclones Hale and Gabrielle), followed by an extended period of hot, dry weather over summer 2024. Such events can substantially affect aquatic invertebrate communities by causing concentrated pulses of sediment and increased rates of invertebrate drift during storm events and low flow conditions coupled with higher water temperatures during summer droughts (Davis, 2023). Although macroinvertebrate communities in Aotearoa New Zealand are resilient to one-off sediment pulses, regular sediment pulses can deplete invertebrate communities (Davis, et al., 2024). SOE monitoring covering 25 sites across the wider Auckland region in 2023 (three months after flooding) and repeated in 2024, found that the average score in four aquatic macroinvertebrates metrics in 2023, post-flooding, was greater than both the year before flooding and the year after in 2024 (Surrey & Storey, 2025). This suggests that macroinvertebrate communities in Tāmaki Makaurau / Auckland's streams did not lose taxa sensitive to habitat degradation long enough to be detected through annual monitoring. Given their small size, however, headwater streams are more exposed to extreme weather events, particularly heatwaves, droughts and low-flow events (Parkyn, et al., 2006).

Fencing and planting riparian margins are important components of stream restoration because stream bank and channel erosion can be major contributors to sediment loads (Basher, 2013). Research conducted both overseas and in Aotearoa New Zealand report up to 90% of sediment transport originating from instream erosion processes (Kronvang, et al., 2013; Hughes, 2016; Hughes, et al., 2022). Research in the Hōteo River catchment, neighbouring the Mahurangi catchment, estimates that up to 72% of the total specific sediment yield originates from stream bank erosion (Simon, et al., 2016). Together with our finding that stock exclusion correlates with lower fine sediment concentrations, this evidence supports excluding livestock from riparian margins and establishing diverse riparian plantings to develop successional, forest like habitats.

Differences in fine sediment between restored streams and open pasture streams were subtle at the site level. Collectively, however, restored streams contained almost a third less deposited sediment than open pasture streams. In addition, mud particles (< 63 μm), which are highly mobile and ecologically detrimental, accounted for more than twice the volume of inorganic fine sediment in open pasture streams compared to restored streams.

Inorganic particles (minerals and non-biological material) indicate erosion; sediment sources from land erosion contain a higher proportion of inorganic particles (Gibbs, 2006). Organic particles (comprised of living or decaying biological material such as leaf litter, woody debris, animal waste, algae and microorganisms) are typically higher in forested and productive rural catchments. Inorganic fine sediment values were highest in open pasture streams, both in absolute terms and as a proportion of total sediment, indicating a larger share of sediment in unfenced streams originates from land or instream erosion.

Ultra fine sediment particles, mud (< 4 μm) and silt (4 μm – 63 μm), have the greatest potential to cause environmental degradation because they remain in suspension longer and can be transported further under lower flow conditions (Gupta, et al., 2022; Wood & Armitage, 1997), allowing them to travel more readily from small headwater streams into larger receiving environments like the Mahurangi Harbour. Once fine sediment is deposited in the environment it can profoundly alter habitat structure by smothering benthic habitats, reducing substrate porosity and interstitial flow and ultimately altering channel morphology. Ecological impacts are wide ranging, including clogging the gills for fish and pollution sensitive invertebrates, increasing invertebrate drift, reducing predator foraging efficiency (via lower water clarity), lowering oxygen concentrations in the benthic environment, degrading food quality for aquatic invertebrates – particularly grazers and deposit feeders – and loss of interstitial habitat between coarse substrates (Blöcher, et al., 2020; Davis, 2023; Ryan, 1991; Wood & Armitage, 1997).

Ecological health was higher in protected, forested streams and lower in open pasture streams. Invertebrate indicators linked to sensitive taxa (QMCI and EPT richness and abundance) showed a stronger relationship with habitat quality than MCI and the TICI derived from eDNA. Upstream catchment factors (catchment size, stock exclusion and native forest cover) were more consistently related to ecological indicators than to sediment metrics, suggesting ecological responses integrate broader catchment influences.

Ecological indicators of stream health can respond faster to riparian restoration than sediment and water quality metrics. Storey and Cowley (1997) reported ecological recovery in three rural streams in the Auckland Region while water quality remained variable (Storey & Cowley, 1997). Scarsbrook and Halliday (2010) found channel morphology and epilithon biomass returned toward reference conditions within 300 m of native forest, while deposited fine sediment and water quality lagged (Scarsbrook & Halliday, 1999). These changes arise from multiple interacting drivers, including increased habitat availability and complexity, improved food quality and cooler water temperatures under a closed canopy of riparian vegetation (Parkyn, et al., 2003). Lower light levels under riparian cover also serves to supress excessive growth of aquatic plants and algae biomass which can choke small streams (Parkyn, 2004). Conversely, sediment and nutrient inputs can continue to enter the system via point source contributions from runoff and unprotected drains and streams (AgResearch Ltd, 2025).

In summary, restored streams in the MLRP study had lower fine sediment measurements, particularly for mud (< 63 um) particles, alongside greater ecological health indicators compared to open pasture streams. Forested reference streams exhibited the lowest fine sediment measures and highest ecological health, although ecological health indicators appeared to be relatively low in these streams, from a regional perspective. Catchment-scale characteristics, particularly native forest cover, the degree of stock exclusion and catchment size also play a role. The underlying mechanisms are complex and changes in erosion processes and fine sediment mobilisation after riparian restoration follow a non-linear response. Reducing physical disturbances to stream channels by excluding livestock and planting riparian margins serves to protect stream banks and filter out sediment and other contaminants from overland run-off and may help moderate peak flows over time.

5.2 Protecting headwater streams facilitates catchment restoration

Headwater streams offer valuable restoration and monitoring opportunities to help better understand and document the process of catchment restoration over achievable timescales. The Mahurangi monitoring programme focused on headwater streams because they are the most numerous waterbodies in the catchment and many discharge directly into the harbour inlets, including Dyers Creek, Cowen Bay, Pukapuka Inlet, Te Kapa River and the Te Muri Inlet. Collectively, headwater streams can be important contributors of fine sediment to the catchment and provide valuable habitats for native wildlife. From a land management perspective, they present achievable restoration opportunities.

A key point from the MLRP study is that fencing and planting headwater streams can measurably reduce fine sediment deposition and that the cumulative ongoing fencing and planting of headwater streams is likely to reduce overall sediment yields to the Harbour. This is supported by observations of sediment yield during high flow events in Te Muri-o-Tarariki stream, a small hill-country stream (with a catchment area of 0.27 km^2) which discharges directly into the Te Waihē / Mahurangi Harbour. Between 2014 to 2019 the stream returned the highest sediment yield (172 $t/km^2/yr$) across eleven monitored sites in the Auckland region (Hicks, et al., 2021). At that time, the stream's catchment was grazed. Between 2020 to 2024, however, approximately 70% of the catchment has subsequently been retired from grazing and planted with mixed native species. This corresponded to a 9.7% decrease in annual average sediment yields measured between 2014 to 2024 and much lower yields were recorded for the period 2020 to 2024 (132 $t/km^2/yr$), though this result was still regionally high (Tsyplenkov & Neverman, 2025). Protecting headwater streams, therefore, is expected to have a cumulative, downstream benefit for Te Waihē / Mahurangi Harbour.

Sediment processes operate at different scales across a catchment driven largely by fluvial processes, land cover and channel morphology. For example, headwater streams are more susceptible to physical disturbances from livestock, weather events and localised changes in land cover. In contrast, the mid-reaches of a catchment are more impacted by fluvial processes arising from changes in stream flow and water level, whereas the lower reaches often have higher banks that are susceptible to continuous erosion as well as sporadic bank slumping, potentially releasing large volumes of sediment in a single event (Hughes, 2016). Headwater streams offer significant opportunities to reduce sediment loading, particularly during peak flow events and provide valuable habitat for native (and potentially under-represented) freshwater invertebrate species (Parkyn, et al., 2006; Tsyplenkov & Neverman, 2025). Moreover, these systems serve as microcosms in the landscape and provide insight into sediment sources governed by larger-scale macro processes, i.e. land derived sediment inputs to the Te Waihē / Mahurangi Harbour.

In addition, because headwater streams have smaller flood zones and require less land to be retired to achieve desired ecological and water quality outcomes, they often represent more manageable riparian restoration solutions for landowners compared with larger streams. They also provide important habitat and refugia for native and even threatened species (including giant kōkopu, kākahi, longfin eel and redfin bully) as evident from the eDNA samples collected during the MLRP monitoring programme.

Both the size and location of restored areas matter: restoration is measurable along a continuum of forested riparian margin, but areas must be large enough to influence stream processes. Proximity to healthy, forested streams also speeds recovery by providing source populations of invertebrates and native fish; small, isolated reaches will recover more slowly and less completely (Quinn, et al., 2009).

5.3 Considerations for riparian restoration

In the Mahurangi catchment, restoration efforts have largely focused on supporting landowners to retire riparian margins, wetlands and steep slopes, excluding livestock, and plant native species. Results from the MLRP study indicate that riparian restoration and increased forest cover work in concert to measurably reduce fine sediment levels in streams and support better ecological health, which can have cumulative benefits to the wider catchment and Te Waihē / Mahurangi Harbour. Programmes like MLRP seek to support and enable the local community to engage with and learn about the restoration process. Field days, community planting days and stalls at public events, are some examples of the different types of community engagement methods supported during the programme. These events created opportunities for people to meet others in their community who are also doing the mahi (work) to restore catchment health, creating a ripple effect from person to person.

The programme has greatly benefited from the partnership with Ngāti Manuhiri. Cultural monitoring has encouraged a more holistic view of catchment health, restoration opportunities and the recovery process. The multi-generational lens held by kaitiaki aligns with the decadal timeframes required to build resilience in the landscape and restore ecosystems. This highlights the ongoing commitment required by local and central government to support local communities if stated catchment restoration outcomes are to be achieved.

Key lessons from the Mahurangi Land Restoration Programme:

- Map and plan with landowners: Conduct site assessments to identify erosion prone slopes and likely sediment pathways. Encourage the Landowner to retire riparian margins that maximise ecological health and sediment reduction, ideally a minimum of 10 m either side.
- Tailor planting and fencing: Use eco-sourced native species suited to the site's hydrology, soil and exposure; include canopy, mid-storey and understorey species. Engage mana whenua to ensure species mixes are ecologically and culturally appropriate.
- Include long-lived native species: Include a 20-25% mix of native long-lived species (those expected to survive 100 years or more) in planting plans to establish long-term canopy cover and kick-start forest regeneration. Adjust this proportion based on the proximity of existing mature forest and local sources of seed.
- Select plant species which offer a range of ecological functions: For example, varied root structures to help stabilise soil, growth forms to increase habitat complexity and species that provide a range of food sources for birds and aquatic fauna across different seasons.

- **Invest in site preparation and maintenance:** Early and ongoing strategic weed and pest control reduces overall maintenance costs. Target invasive pest plants and browsing animals; plan biannual plant releases for at least two to three years after planting, with seasonal checks (spring and autumn) and additional releases as needed.
- **Use robust fencing:** Permanent fencing offers the best protection from livestock access.
- **Design for flood and erosion risk:** Where possible, locate fences outside of flood zones. Tailor planting plans to take into account local flow dynamics, seasonal soil moisture levels and potential changes to stream channels from scouring and/or widening. Take note of actively eroding sites near streams and exclude livestock from these, wherever possible.
- **Provide funding on a reimbursement basis:** Once works have been completed, with the option to reimburse at staged milestones to reduce the upfront financial burden on landowners.
- **Allow flexibility as to how landowners meet their contribution:** Including monetary, in-kind labour, donated materials such as plants, and volunteer support.
- **Retain a portion of the funding:** Until post-planting releasing has been completed to incentivise this critical activity and support higher plant survival rates.
- **Provide ongoing support and demonstration:** Continued advisory support, accessible maintenance funding, and visible demonstration sites increase community buy in and wider uptake.
- **Include monitoring and adapt management practices as needed:** Establish monitoring to measure the effectiveness of restoration actions and learn where and how to improve outcomes to support knowledge-based adaptive management.
- **Build trust with landowners:** Consider what's important to the local community and tailor engagement and support mechanisms to build trust and sustained relationships. Wherever possible, maintain continuity by assigning the same land advisor to the project for its duration to strengthen trust, improve knowledge retention, and support effective relationship building.

Riparian planting on its own is unlikely to restore catchment health without incorporating other strategically implemented catchment management actions, such as afforestation of erosion-prone slopes, good soil management, wetland protection and enhancement, and strategically placed interventions (such as constructed wetlands, buffer strips, and bunds) (Levine, et al., 2021; Hughes & Dang, 2026; Tanner, et al., 2022; Tomscha, et al., 2021). These realities mean that detecting changes in medium and larger waterways will take longer and the impacts of localised restoration projects can be diluted by sediment contributions from the wider catchment.

5.4 Limitations of this monitoring programme

This programme covered nine headwater streams (three riparian treatment groups) from mid-2023 to end 2025, providing a modest dataset for detecting differences in fine sediment composition, volume and ecological health. The nine streams did not represent the strong gradient of catchment characteristics that they did for reach-scale habitat quality. In addition, some potentially influential parameters were not captured (e.g. detailed site assessments of the soil type, slope, rainfall and stream flow), due to resourcing and time constraints. The small sample size also limited statistical power.

Hydrological extremes before and during monitoring added substantial background noise and further increased variability in the dataset. Cyclone Hale (12 Jan 2023) and Cyclone Gabrielle (12-16 Feb 2023) occurred only months prior to monitoring and likely generated large sediment pulses and stream flow leading to invertebrate drift. Over the three-year monitoring period, soil moisture conditions shifted markedly, from saturation in 2023 to drought in 2024. Stream flow measurements collected during monitoring were rudimentary and would represent low-flow levels. They were not collected concurrently with turbidity readings from sensors, constraining the ability to convert measured concentrations to catchment yields or link sediment peaks to specific flow events. Ecological responses may also have been impacted by multiple sediment pulses or other pollutants, which were not measured during the study.

Due to the short-term nature of the monitoring programme, space-for-time comparisons were necessary, as pre-restoration data were limited. This approach assumes comparability among sites and is sensitive to between site heterogeneity, including differences in grazing intensity, riparian conditions and upstream land cover, slope, and soil type.

Monitoring of streams fenced and planted ≥ 6 years earlier helped to characterise transitional states, but longer monitoring periods and more replicated controls would strengthen inference. Furthermore, disentangling the effects of riparian restoration from other land management practices and broader catchment-scale drivers (e.g. forest cover, land use change, urban development, wastewater events) requires denser temporal sampling and more detailed catchment assessments, which were beyond the scope of this study.

6 Conclusions and recommendations

This study shows that stream restoration in the Mahurangi catchment, through stock exclusion and native riparian planting, is likely to be reducing fine sediment and improving ecological health in the majority of monitored headwater streams. The findings also highlight the importance of native forest cover and stock exclusion in the upper catchment, particularly to aquatic invertebrate communities. Headwater streams represent high-value, practical targets for riparian restoration. The results support continued implementation of targeted, well-designed riparian restoration measures (stock exclusion and native planting) delivered in partnership with mana whenua and the community. These actions can reduce fine sediment accumulation and enhance important habitats for native wildlife.

Recommendations for ongoing and future restoration include ensuring that native plant species selection is well matched to site conditions and supported by follow-up maintenance for at least two years to aid establishment, and linking projects across the catchment to create wildlife corridors. However, riparian restoration should not be treated as a 'silver bullet' to achieve catchment recovery. Developing a catchment-scale approach which identifies hotspots for contaminate loss (like sediment) and transportation pathways, in conjunction with high-value sites for biodiversity, water quality and culturally important sites will reinforce overall restoration outcomes. The role of catchment-scale factors like forest cover in mediating peak-flow events must not be overlooked. A range of complementary interventions, including sustainable soil management (to reduce compaction and the occurrence of bare ground), wetlands protection and enhancement, partial or full retirement and afforestation of steep-sided gullies and hillsides, and the installation of strategically located structures to intercept runoff – such as small detainment bunds, planted swales and drainage ditches vegetated with native sedges and rushes to slow runoff and intercept contaminants – will further support catchment resilience.

This monitoring programme has provided valuable insights for the Mahurangi catchment. Given its modest sample size, however, the results should be read as indicative of what is being achieved by restoring riparian margins. Wider catchment restoration outcomes unfold over decadal timescales. Future monitoring is recommended following sediment measurements protocols presented here to track changes in restored streams over longer time periods. Collecting information on catchment-scale characteristics alongside peak-flow data for first order streams will help address current knowledge gaps regarding the relative importance of catchment-scale landcover versus riparian management. This would be best supported by detailed records of the location and nature of restoration initiatives across the catchment to guide future restoration priorities and monitoring opportunities.

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Appendix A: Scoring and grading for habitat and ecological health indicators

Table A-1. Interpretation of Rapid Habitat Assessment (RHA) (Clapcott, 2015).

RHA score	Habitat Quality	Description
> 75	Excellent	Stream in excellent condition and close to reference state.
51 - 75	Good	Stream in good condition, low level of impact only deviating slightly from a reference state.
25 - 50	Fair	Stream in fair condition, showing some signs of impairment and impact, deviating moderately from reference state.
< 25	Poor	Stream in poor condition, multiple habitat indicators are impaired and deviate substantially from reference state.

Table A-2. Interpretation of macroinvertebrate indices, Macroinvertebrate Community Index (MCI), Quantitative Macroinvertebrate Community Index (QMCI) attribute bands as defined in the National Policy for Freshwater Management (NPS-FM, 2020).

MCI score	QMCI score	Attribute band	Description
≥130	≥6.5	A	Macroinvertebrate community, indicative of pristine conditions with almost no organic pollution or nutrient enrichment.
≥110 - <130	≥5.5 - <6.5	B	Macroinvertebrate community indicative of mild organic pollution or nutrient enrichment. Largely composed of taxa sensitive to organic pollution/nutrient enrichment.
≥90 - <110	≥4.5 - <5.5	C	Macroinvertebrate community indicative of moderate organic pollution or nutrient enrichment. There is a mix of taxa sensitive and insensitive to organic pollution/nutrient enrichment.
90	4.5	National bottom line	
<90	<4.5	D	Macroinvertebrate community indicative of severe organic pollution or nutrient enrichment. Communities are largely composed of taxa insensitive to inorganic pollution/nutrient enrichment.

Table A-3. Quality classes for macroinvertebrate indices MCI and QMCI following the method outlined in (Stark & Maxted, 2007).

MCI score	QMCI score	Quality Index
> 119	> 5.99	Excellent
100 - 119	5.00 - 5.99	Good
80 - 99	4.00 - 4.99	Fair
< 80	< 4	Poor

Table A-4. Interpretation of the Taxonomic Independent Community Index (TICI). The TICI is calculated from species data extracted from eDNA barcoding as described by (Wilkinson, et al., 2024).

TICI score	Ecological Health	Description
>120	Pristine	Stream is in pristine condition, with minimal impact.
110 - 120	Excellent	Stream in excellent condition and close to reference state.
100 - 110	Good	Stream in good condition, low level of impact only deviating slightly from a reference state.
90 - 100	Average	Stream in fair condition, showing some signs of impairment and impact, deviating moderately from reference state.
80 - 90	Poor	Stream in poor condition, multiple habitat indicators are impaired and deviate substantially from reference state.
<80	Very Poor	Stream is heavily polluted.

Table A-5. Fish Index of Biotic Integrity (F-IBI) and NPS-FM attribute bands and corresponding descriptions (NPS-FM, 2020).

F_IBI score	Attribute Band	Description
≥34	A	High integrity of fish community. Habitat and migratory access have minimal degradation.
<34 - ≥28	B	Moderate integrity of fish community. Habitat and/or migratory access are reduced and show some signs of stress.
<28 - ≥18	C	Low integrity of fish community. Habitat and/or migratory access is considerably impairing and stressing the community.
<18	D	Severe loss of fish community integrity. There is substantial loss of habitat and/or migratory access, causing a high level of stress on the community.

Appendix B: Correlations between sediment indicators

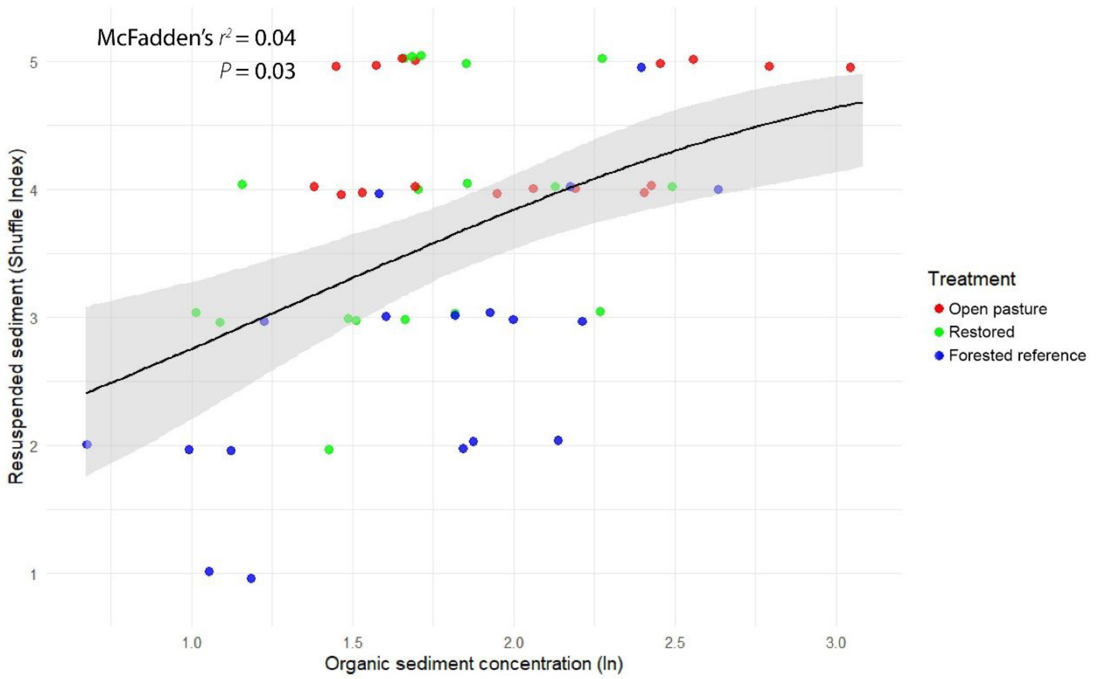

Figure B-1. Correlation between the shuffle index and the concentration of total organic sediment.

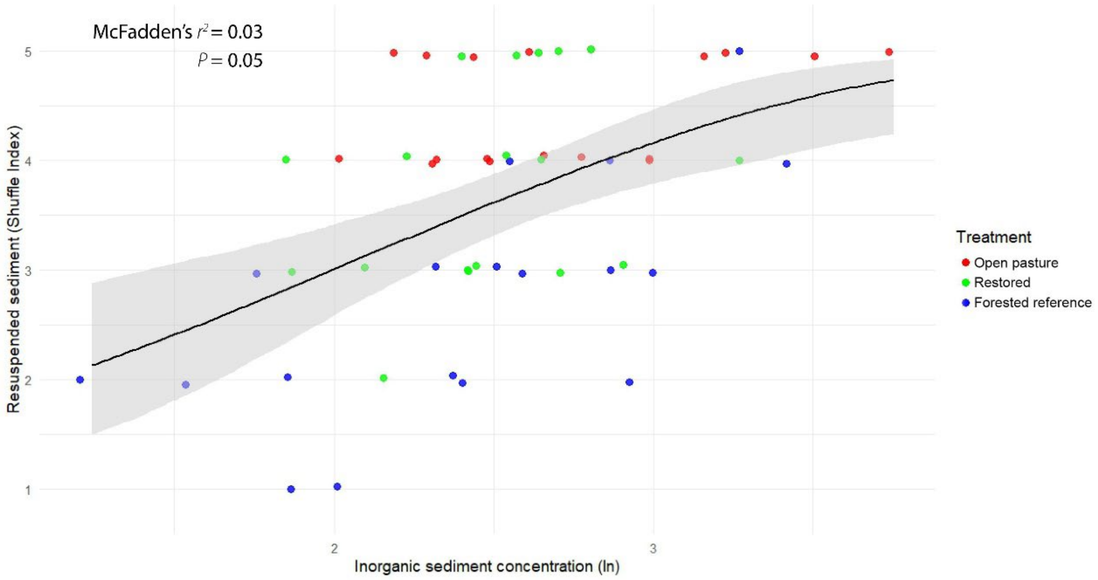

Figure B-2. Correlation between the shuffle index and the concentration of total inorganic sediment.

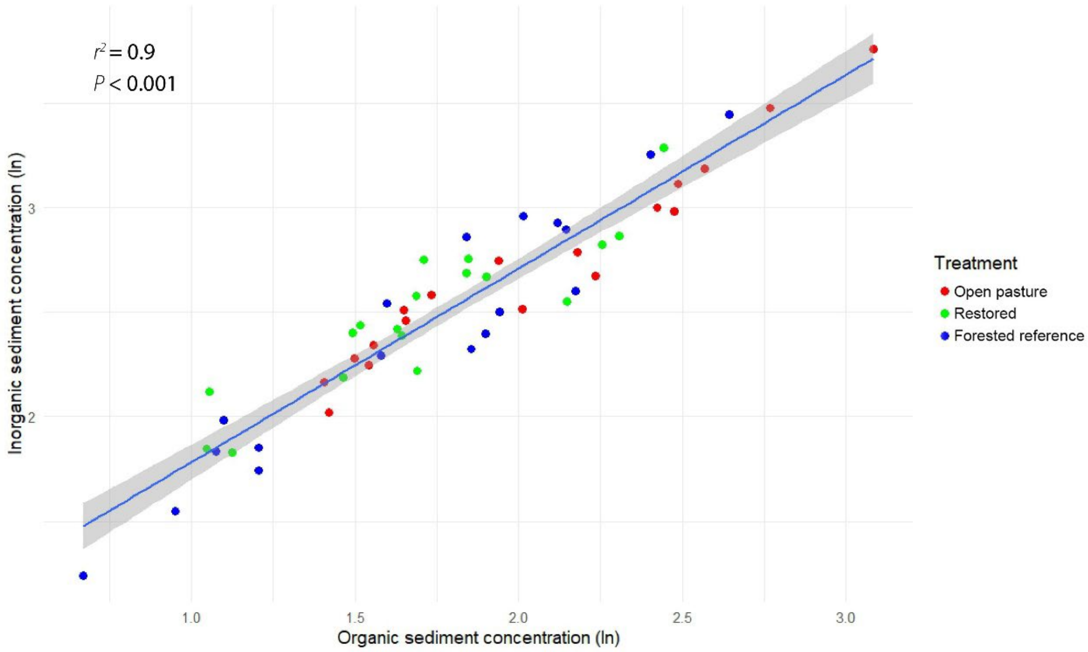

Figure B-3. Correlation between the total concentration of organic and inorganic sediment collected using the Quorer sampling method.

Appendix C: Statistical model outputs

Tables C-1 to C-3 summarise the results of cumulative link mixed models and linear mixed models. The models test the relationship between reach-scale habitat quality (RHA) and catchment characteristics (land cover and catchment size) against three types of environmental response variable: a qualitative resuspended sediment indicator using the shuffle index method, quantitative resuspended sediment indicators collected using the Quorer method, and freshwater invertebrates. Statistically significant models are shown in **bold**. The best performing models (i.e. the strongest predictor set) are highlighted in **bold and red**. Models which were not statistically significant are noted as n.s.

Reported statistics for cumulative link mixed models include, Akaike information criterion (AIC), delta Akaike (ΔAIC), Akaike weights (*wi*), pseudo *r* squared (McFadden's r^2) and significance values (*P*). Reported statistics for linear mixed models include, degrees of freedom (d.f.), Akaike information criterion corrected for small sample size (AICc), Akaike weights (w_i), *r* squared (r^2)

Table C-1. Cumulative link mixed model results for resuspended sediment assessed using the shuffle index. Model set one contains the predictor variables, treatment group, catchment characteristics, reach-scale habitat quality (RHA), and a Full Model containing the best-performing predicator variables (indicated with an asterisk '*'). Model set two assess qualitative shuffle index results against quantitative concentrations of inorganic and organic resuspended sediment collected using the Quorer method.

Response variable	Model Set	Predictor variables	AIC	ΔAIC	w_i	McFadden's r^2	*P*
Resuspended sediment – Shuffle index (n = 53)	I	**Treatment Group**	**136**	**6.7**	**0.018**	**0.06**	**0.001**
		Native Bush %	142	12.5	< 0.001	0.005	n.s.
		*Stock Exclusion %	142	12.5	0.001	0.005	n.s.
		Catchment Size	142	12.9	< 0.001	0.002	n.s.
		***RHA**	**130**	**0.3**	**0.456**	**0.10**	**< 0.001**
		Full Model	**130**	**0**	**0.523**	**0.12**	**< 0.001**
	II	**Inorganic sediment concentration**	**139**	**1.05**	**0.372**	**0.03**	**0.047**
		Organic sediment concentration	**138**	**0**	**0.628**	**0.04**	**0.027**

Table C-2. Linear mixed model results for resuspended sediment parameters collected using the Quorer method, tested against treatment group, catchment characteristics, reach-scale habitat quality (RHA), and a Full Model containing the best-performing predicator variables (indicated with an asterisk '*').

Sediment Response Variable	Predictor variables	d.f.	Log likelihood	AICc	w_i	r^2	P
Inorganic sediment (g/m^3) n = 54	Treatment Group	4	-33.45	75.7	0.018	0.07	n.s.
	Native Bush %	4	-32.67	74.1	0.040	0.10	n.s.
	***Stock Exclusion %**	**4**	**-31.85**	**72.5**	**0.090**	**0.14**	**0.03**
	Catchment Size	4	-34.21	77.2	0.009	<0.001	n.s.
	*RHA	4	-30.38	69.6	0.392	0.23	0.008
	Full Model	5	-29.03	69.3	0.451	0.30	n.s.
Organic sediment (g/m^3) n = 54	Treatment Group	4	-37.11	83.0	0.024	0.10	n.s.
	Native Bush %	4	-36.99	82.8	0.027	0.08	n.s.
	***Stock Exclusion %**	**4**	**-36.24**	**81.3**	**0.058**	**0.11**	**0.04**
	Catchment Size	4	-38.48	85.8	0.006	<0.001	n.s.
	*RHA	4	-33.95	76.7	0.569	0.20	0.004
	Full Model	5	-33.32	77.9	0.315	0.22	n.s.
Inorganic mud < 63 µm (g/m^3) n = 27	Treatment Group	4	-44.66	102.2	0.028	0.08	n.s.
	Native Bush %	4	-45.31	100.4	0.066	0.02	n.s.
	*Stock Exclusion %	4	-44.87	99.6	0.102	0.06	n.s.
	Catchment Size	4	-45.55	100.9	0.052	0.004	n.s.
	*RHA	4	-43.11	96.0	0.597	0.18	0.02
	Full Model	5	-42.93	98.7	0.155	0.19	n.s.

Table C-3. Linear mixed model results for invertebrate response variables (MCI, QMCI, EPT richness and EPT abundance) tested against treatment group, catchment characteristics, reach-scale habitat quality (RHA), and a Full Model containing the best-performing predicator variables (indicated with an asterisk '*').

Invertebrate Response Variable	Predictor variables	d.f.	Log likelihood	AICc	w_i	r^2	P
MCI n = 36	Treatment	5	14.67	-16.5	0.030	0.50	0.002
	*Native Bush %	4	15.36	-20.9	0.275	0.50	< 0.001
	Stock Exclusion %	4	14.33	-18.8	0.099	0.43	0.001
	Catchment Size	4	14.48	-19.1	0.115	0.49	0.002
	*RHA	4	11.57	-13.3	0.006	0.25	0.03
	Full Model	5	17.42	-22.0	0.475	0.60	≤ 0.05
QMCI n = 36	Treatment	5	9.71	-6.6	0.739	0.58	< 0.001
	*Native Bush %	4	4.85	0.1	0.026	0.35	0.004
	Stock Exclusion %	4	3.68	2.5	0.008	0.25	0.02
	Catchment Size	4	2.70	4.4	0.003	0.22	n.s.
	*RHA	4	4.89	0.0	0.027	0.38	0.004
	Full Model	5	8.38	-3.9	0.197	**0.54**	≤ 0.01
EPT richness n = 36	Treatment	5	-43.04	98.9	0.040	0.38	0.01
	Native Bush %	4	-44.93	99.7	0.028	0.18	n.s.
	*Stock Exclusion %	4	-44.70	99.2	0.035	0.22	0.03
	Catchment Size	4	-44.28	98.4	0.053	0.27	0.048
	*RHA	4	-42.20	94.2	0.423	0.38	0.006
	Full Model	5	-40.69	94.2	0.421	0.49	n.s.
EPT abundance n = 36	Treatment	5	-31.14	75.1	0.672	0.68	<0.001
	*Native Bush %	4	-36.93	83.7	0.009	0.28	0.01
	Stock Exclusion %	4	-37.44	84.7	0.006	0.20	0.03
	Catchment Size	4	-37.12	84.1	0.008	0.32	0.04
	*RHA	4	-34.41	80.6	0.043	0.43	0.003
	Full Model	5	-32.08	77.0	0.262	**0.63**	≤ 0.01

Appendix D: Continuous records of total suspended sediment and rainfall

The three graphs show continuous readings of total suspended sediment (TSS) and rainfall recorded at 15-minute intervals. TSS was measured with turbidity sensors installed in three streams: representing one of the three riparian treatment groups in the monitoring programme. Rainfall data for the same period was taken from the Auckland Council rain gauges at Satellite Station Road and the Warkworth wastewater treatment plant. For each graph, the rainfall record from the gauge located nearest to each respective sensor was used.

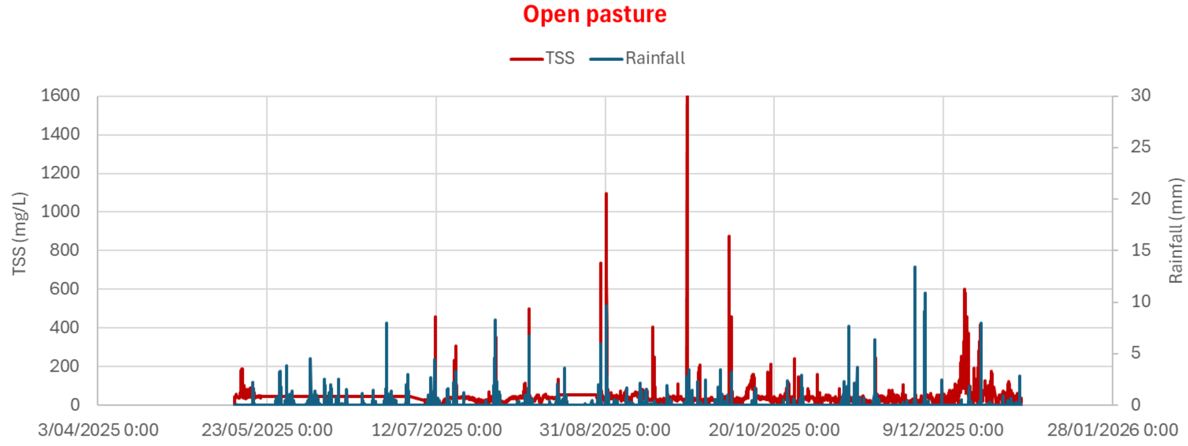

Figure D-1. Sediment sensor readings total suspended solids (TSS) and rainfall, Open pasture stream, 14 May to 31 December 2026.

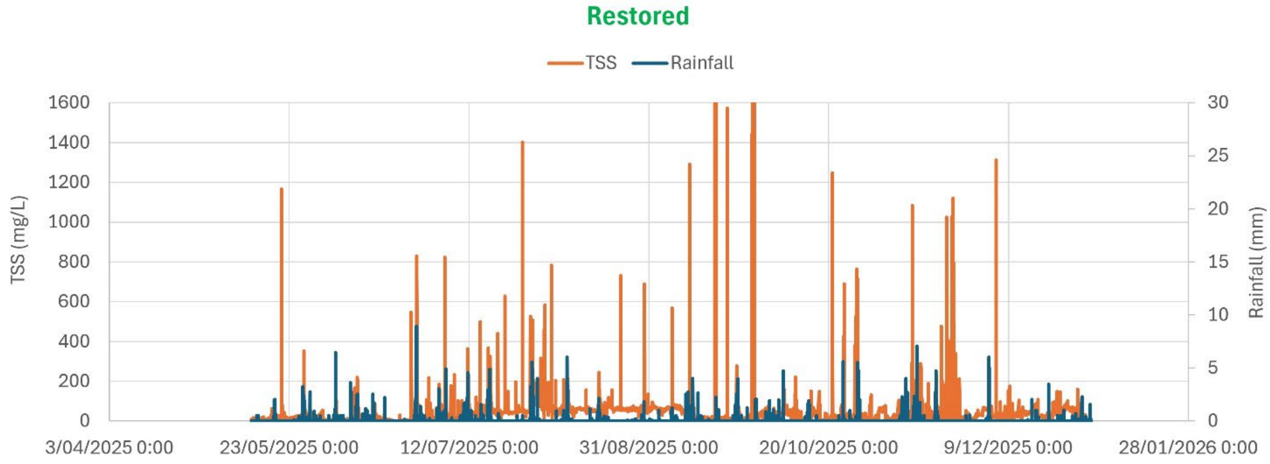

Figure D-2. Sediment sensor readings for total suspended solids (TSS) and rainfall, for restored streams (fenced with native planted native scrub), 14 May to 31 December 2026.

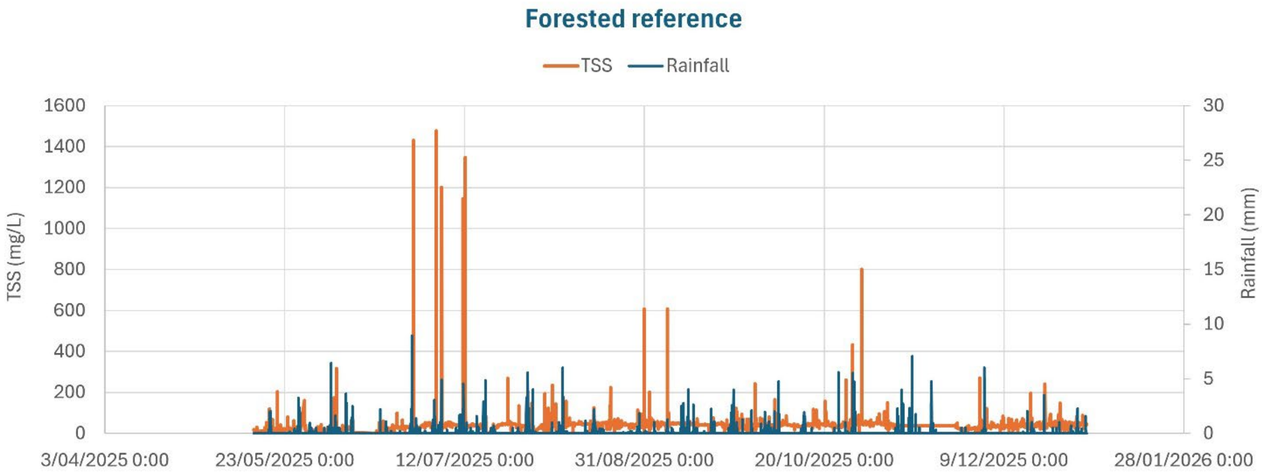

Figure D-3. Sediment sensor readings total suspended solids (TSS) and rainfall, forested reference streams (fenced stream under mature native forest), 14 May to 31 December 2026.

Appendix E: Graded macroinvertebrate results by site

Table E-1. Macroinvertebrate Community Index (MCI), Quantitative Macroinvertebrate Community Index (QMCI), and Ephemeroptera Plecoptera Trichoptera (EPT) taxon richness and abundance scores across all monitoring sites. Interpretation of MCI and QMCI scores are graded against quality classes following Stark and Maxted (2007) and the attribute states defined in the National Policy for Freshwater Management (2020).

Site ID	Treatment	Median Score ± 1 Standard deviation						
		Number of samples	MCI score Quality classes	MCI NPS-FM band	QMCI score Quality classes	QMCI NPS-FM band	% EPT richness	% EPT abundance
1	Open pasture	3	80 ± 10	D	3.1 ± 1.0	D	14.3 ± 11.9	1.4 ± 0.1
2		3	66 ± 6	D	3.1 ± 0.6	D	0 ± 5.8	0 ± 0.3
3a		1	68	D	2.4	D	14.3	1.0
3b		2	83 ± 10	D	3.5 ± 0.6	D	14.9 ± 12.2	3.5 ± 4.2
4	Restored	3	65 ± 5	D	3.5 ± 0.7	D	8.3 ± 5.1	0.9 ± 3.8
5		3	92 ± 9	C	4.3 ± 1.4	D	23.5 ± 3.6	3.0 ± 22.9
6		3	93 ± 21	C	4.0 ± 0.2	D	21.4 ± 7.9	8.4 ± 5.0
7	Forested reference	3	99 ± 14	C	5.7 ± 1.3	B	26.7 ± 3.8	34.8 ± 18.0
8		3	108 ± 11	C	5.7 ± 1.0	B	32.0 ± 9.8	41.5 ± 24.7
9		3	100 ± 7	C	4.4 ± 0.8	D	21.4 ± 2.5	12.8 ± 1.2

Quality classes following Stark & Maxted (2007)	Excellent	Good	Fair	Poor
NPS-FM (2020) Attribute bands	A	B	C	D

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