The ecological consequences of stream sedimentation for freshwater macroinvertebrates in the Te Awanui/Tauranga Harbour catchment

A thesis

submitted in partial fulfilment

of the requirements for the degree

of

Master of Science in Ecology and Biodiversity

[Faculty of Science and Engineering]

at

The University of Waikato

by

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2024

Abstract

Sedimentation is a significant stressor in streams and rivers globally. The natural sediment regime paradigm posits that human activities have greatly altered the natural transport of sediment in river networks and that land-uses leading to increased bank erosion and surface runoff have resulted in large quantities of fine sediment being deposited in streams and rivers. In New Zealand, land conversion from native forest to pastoral agriculture has dramatically increased riverine sediment loads, making deposited fine sediment (DFS) a major stressor that negatively impacts stream ecosystem health. Consequently, DFS is an important attribute managed under the National Policy Statement for Freshwater Management 2020. However, uncertainties remain in how best to monitor the effects of DFS, and what impacts different land-use types have on stream ecosystems in addition to the adverse influence of sedimentation.

In my study, I surveyed 24 stream sites allocated evenly to three land-use types (native forest, pastoral agriculture, and horticulture, e.g. kiwifruit orchards) in the Tauranga Harbour basin. I measured habitat attributes, including DFS, and sampled stream macroinvertebrate communities at each site. Using macroinvertebrate data, I calculated and compared taxonomic and trait-based metrics as indicators of ecosystem health across land-use types. I used a variety of metrics, including the Macroinvertebrate Community Index (MCI) and its quantitative equivalent, the QMCI, the Average Score Per Metric (ASPM) index, and community-weighted mean trait abundances, including facets of functional diversity to assess impacts on stream health.

I found that human land uses negatively influenced stream health, indicated by greatly reduced MCI, QMCI, and ASPM scores. Although DFS explained most of the negative influences on macroinvertebrate communities in pastoral and horticultural streams, there were differences between native forest and horticultural streams not accounted for by DFS that contributed to declines in ecosystem health. I also found that sediment-specific macroinvertebrate metrics (sediment ‘decreasers’) were the best indicators of DFS impacts on stream health. Contrary to my predictions, I did not find that functional diversity was strongly reduced by land use or DFS, although several key traits responded to these stressors. Notably, life history trait modalities involving the number of reproductive cycles per year and oviposition site responded strongly to DFS and land use, indicating potential mechanisms contributing towards the impacts of these stressors on macroinvertebrate communities.

My study points to the need to continuously manage the deposition of fine sediment in streams to help mitigate the adverse effects of human land use. Future research should further explore the role that fine sediment and other stressors associated with human land uses play in the degradation of streams ecosystems in the Tauranga Harbour basin. Quantifying land use and land cover (LULC) attributes using geospatial tools will help further explore the relationship between catchment properties and macroinvertebrate responses. Additionally, issues regarding trait syndromes, trade-offs, and database inconsistencies must be addressed to better understand the mechanistic links between specific macroinvertebrate traits and fine sediment.

Acknowledgements

I want to begin first by acknowledging the tangata whenua and kaitiaki of Te Awanui/Tauranga Moana (Ngāi Te Rangi, Ngāti Pūkenga, Ngāti Ranginui) and their deep connection to the taiao of their kāinga tupu. With this recognition I hope that my thesis adheres to the principles of ki uta ki tai, in that it provides invaluable knowledge that can be used to help manage and restore the wai māori and mahinga kai of the Tauranga Moana area from the mountains to the sea.

I would also like to thank my supervisor Frank Burdon who has shown nothing but absolute dedication and support to me throughout my Master's journey. It's important to me that I acknowledge the vast amount of time you've spend helping me to complete my thesis over these last two years and guiding me into the world of freshwater ecology. I will always be deeply appreciative of the mentorship you have offered me, and I will aim to carry all of the knowledge and experience you have given me into my professional life as a freshwater ecologist.

I would also like to thank Elizabeth Copeland and Bex Gibson for taking the time to help me with sampling and with labwork along with all of the post-graduate students who helped support me through my thesis as well. I enjoyed my time with you guys.

I would like to thank the Bay of Plenty Regional Council Toi Moana for awarding me with the He Toka Tū Moana Scholarship. I also thank Te Papa Atawhai Department of Conservation for awarding me a 2023 Te Papa Atawhai Postgraduate Scholarship and the University of Waikato for a Research Masters Scholarship. This financial support was invaluable in helping me to complete my thesis.

I would also like to thank my friends and family for supporting me throughout the completion of my Master's thesis.

I also want to acknowledge the many trials and challenges I have faced over the last two years. I feel that it is important for me to mention this as I want to offer some solace to anyone undertaking their Master's thesis or another ambitious venture — although there may be many setbacks and you may feel discouraged, it is possible to achieve your goals despite your circumstances with perseverance, faith, and support from whoever and whatever you can draw it from. With this said, I feel that I have gained valuable lessons from the stream ecosystems I have studied in going with the flow, enduring adverse conditions, and adapting to continuous change just like the macroinvertebrates I encountered through my journey.

A final thanks to everyone who helped me to complete my thesis, I can’t thank you enough.

“When you pass through the waters, I will be with you; and when you pass through the rivers, they will not sweep over you”

Isaiah 43:2

Table of Contents

Chapter 1: Introduction ***10***

1.1 Importance of freshwater ecosystems 10

1.2 Human impacts on stream ecosystems 10

1.2.1 Sedimentation 11

1.2.2 Nutrient enrichment 12

1.2.3 Contaminant pollution 13

1.2.4 Hydrological alteration 14

1.2.5 Clearing of riparian vegetation 15

1.2.6 The loss of large wood 16

1.2.7 Other impacts 17

1.3 Land-use impacts on streams and rivers in New Zealand 18

1.4 Assessment of stream ecosystem health 19

1.5 Stream macroinvertebrates as a biomonitoring tool 23

1.6 Functional traits of stream macroinvertebrates 26

1.7 Aims and hypotheses 27

Chapter 2: Methods ***31***

2.1 Study area 31

2.1.1 Tauranga Harbour catchment 31

2.1.2 Significance to Tangata Whenua 34

2.1.3 Land uses 36

2.1.4 Geology 36

2.1.5 Climate 37

2.1.6 Topography 38

2.2 Sampling design 38

2.3 Survey methods 39

2.3.1 Water quality 39

2.3.2 Discharge, flow, and depth 39

2.3.3 Light 39

2.3.4 Bed stability 39

2.3.5 Benthic substrate and fine sediment 41

2.3.6 In-stream habitat conditions 42

2.3.7 Riparian habitat 42

2.3.8 Macroinvertebrates 45

2.4 Laboratory analyses 45

2.4.1 Nitrate-nitrogen 45

2.4.2 Macroinvertebrates 46

2.5 Data analysis 47

2.5.1 Taxonomic-based indicators 47

2.5.2 Functional diversity indicators 48

2.5.3 Mixed models and linear regression 49

2.5.4 Community analyses 50

Chapter 3: Macroinvertebrate responses to land use and sedimentation........ *51*

3.1 Introduction 51

3.2 Results 53

3.2.1 Environmental indicators 53

3.2.2 Macroinvertebrate community composition 57

3.2.3 Macroinvertebrate diversity indices 58

3.2.4 Macroinvertebrate ranked abundances 62

3.2.5 National Objective Framework Bands ... 64

3.2.6 NPS-FM metrics regarding land-use type and sediment ... 66

3.2.7 NPS-FM macroinvertebrate metrics and sediment ... 71

3.2.8 Sediment-specific macroinvertebrate metrics and sediment ... 72

3.3 Discussion ... 74

3.3.1 Sediment ... 74

3.3.2 Land-use ... 79

3.3.3 Riparian vegetation ... 80

3.3.4 Nitrate ... 82

3.3.5 Pesticides ... 84

3.3.6 Stream flow velocities ... 88

Chapter 4: Land use impacts on invertebrate traits and functional diversity ... *95*

4.1 Introduction ... 95

4.1.1 Metrics of functional diversity ... 97

4.1.2 Objectives of this chapter ... 98

4.2 Results ... 99

4.2.1 Functional diversity indicators ... 99

4.2.2 Macroinvertebrate trait abundances in relation to land use and sediment ... 103

4.2.3 The relationship between macroinvertebrate traits and sediment ... 137

4.3 Discussion ... 154

4.3.1 Metrics of functional diversity ... 154

4.3.2 Trait relationships with sediment ... 165

4.3.3 Trait relationship with land-use ... 168

Chapter 5: Synthesis and conclusions ... *172*

5.1 Hypothesis outcomes ... 172

5.2 Comparison of taxonomic and trait-based approaches 175

5.3 Limitations 178

5.4 Implications and proposed actions 179

5.5 Areas for further research.. 181

Cited literature 186

Appendices 208

7.1 Appendix 1 208

7.2 Appendix 2 210

Figures

Figure 1. a) The location of the 24 study sites in the Te Awanui/Tauranga Harbour catchment area. Dark green labels indicate native forest land use. Yellow labels indicate pastoral land use and light/lime green labels indicate horticultural land use. b) The study location in relation to New Zealand (North Island)……………………………………………………………………………………………..35

Figure 2. Changes in key environmental indicators across streams in three local land-use types in the Te Awanui/Tauranga Harbour basin. a) logit-transformed deposited fine sediment (%) on the stream bed; b) log-transformed concentrations of nitrate nitrogen (mg/); c) riparian habitat integrity indicated by the Riparian Condition Inde; and d) square-root transformed flow velocities (m/s). All indicators are standardized (centred on the mean and scaled to unit variance)……………………………………….57

Figure 3. The substrate composition of a) native forest streams, b) pastoral streams, c) horticultural streams and d) overall. Substrate composition was measured using the SAM3 method (Wolman walk)………….58

Figure 4. Non-metric multidimensional scaling (NMDS) plot of macroinvertebrate communities across streams in three local land-use types (green, native forest; khaki, pastoral; olive, horticultural) in the Te Awanui/Tauranga Harbour basin. Indicator taxa are shown by coloured text: green, native forest; khaki, pastoral; brown, pastoral and horticultural. Taxa in the grey text are the ten most abundant found across all sites (including two indicator taxa). Lines indicate convex hulls for each land use type…………………..61

Figure 5. Boxplots showing the (a) total abundance, (b) taxa richness, and (c) Simpson's Diversity Index of macroinvertebrates sampled across 24 sites in the Te Awanui catchment (Bay of Plenty). Local land-use type is indicated (Native forest, Pastoral, Horticultural………………………………………………..63

Figure 6. Barchart shows the (a) total abundance, (b) taxa richness, and (c) Simpson's Diversity Index of macroinvertebrates sampled across 24 sites in the Te Awanui catchment (Bay of Plenty). Local land-use type is indicated (Native forest, Pastoral, Horticultural)……………………………………………….64

Figure 7. The top ten most abundant stream macroinvertebrate taxa at sites sampled in the Te Awanui catchment: a) overall, b) in native forest streams, c) pastoral streams and d) horticultural streams……….66

Figure 8. Macroinvertebrate indices from stream sites in the Te Awanui/Tauranga Harbour basin. a) the Macroinvertebrate Community Index, and b) its quantitative equivalent, the QMCI; c) the Average Score Per Metric (APSM) index, including its constituent indices d) EPT taxa richness, and e) EPT relative abundance. Indices are ranked from highest to lowest, and the local land-use type is indicated. The horizontal lines indicate the biocriteria stated in the NPS-FM; the dotted lines indicate bands A-C; the solid line indicates a D status or below the national bottom line. Care is needed to interpret this data since it reflects only one sampling date and not the median values of five years of sampling required in the NPS-FM……………………………………………………………………………………………….68

Figure 9. Changes in key macroinvertebrate indicators across streams in three local land-use types in the Te Awanui/Tauranga Harbour basin. a) Macroinvertebrate Community Index (QMCI), b) Quantitative macroinvertebrate Community Index (MCI), c) EPT relative abundance (%EPT Abundance), d) EPT taxa richness (EPT richness), e) Average Score Per Metric (ASPM) index…………………………………...71

Figure 10. Boxplots of the changes in sediment-specific macroinvertebrate indicators across streams in three local land-use types in the Te Awanui/Tauranga Harbour basin. a) % sediment 'decreasers' taxa richness, and b) % sediment 'decreasers' abundances…………………………………………………...72

Figure 11. Barcharts of the changes in sediment-specific macroinvertebrate indicators across streams in three local land-use types in the Te Awanui/Tauranga Harbour basin. a) % sediment 'decreasers' abundances and b) % sediment 'decreasers' taxa richness……………………………………………….72

Figure 12. Boxplots of the changes in sediment-specific macroinvertebrate indicators across streams in three local land-use types in the Te Awanui/Tauranga Harbour basin. a) % sediment 'increaser' taxa richness, and b) % sediment 'increaser' abundances…………………………………………………….73

Figure 13. Barcharts of the changes in sediment-specific macroinvertebrate indicators across streams in three local land-use types in the Te Awanui/Tauranga Harbour basin. a) % sediment 'increaser' taxa richness, and b) % sediment 'increaser' abundances…………………………………………………….73

Figure 14. Linear regressions showing the relationships between Deposited Fine Sediment (DFS) and a) QMCI, b) MCI, c) (logit) EPT abundance, d) EPT Richness and e) ASPM in the Te Awanui/Tauranga Harbour basin. The grey ribbon indicates the 95% confidence interval for the regression line……………75

Figure 15. The relationships between Deposited Fine Sediment (DFS) and a) decreaser abundance, b) decreaser richness……………………………………………………………………………....76

Figure 16. Linear regressions showing the relationships between Deposited Fine Sediment (DFS) and a) increaser abundance, b) increaser richness…………………………………………………………....77

Figure 17. Boxplots of the changes in functional diversity metrics across streams in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) Functional richness, b) Functional evenness, c) Functional Divergence, d) functional dispersion, e) Rao's Q ………103

Figure 18. Barchart of the changes in functional diversity metrics ranked from highest to lowest across streams in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) Functional richness, b) Functional evenness, c) Functional Divergence, d) functional dispersion, e) Rao's Q……………………………………………………………………….....104

Figure 19. Linear regression plots displaying the relationship between Deposited Fine Sediment (DFS) and a) Functional richness, b) Functional evenness, c) Functional divergence, d) Functional dispersion & e) Rao's Q...…………………………………………………………………………………..106

Figure 20. Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'maximum potential size' across three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) size1, b) size2, c) size3, d) size4, e) size5…..109

Figure 21. Changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'body flexibility' across three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) noflex, b) lowflex, c) highflex………………………………………………………………………………………111

Figure 22. Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'body form' across three different local land-use types (native forest, pastoral,

horticultural) in the Te Awanui/Tauranga Harbour basin. a) streamlined, b) flattened, c) cylindrical, d) spherical…………………………………………………………………………………………...113

Figure 23. Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'respiration of aquatic stages (not including eggs)' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) tegument, b) gill, c) plastron, d) aerial...……………………………………………………………………………115

Figure 24. Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'maximum number of descendants per reproductive cycle' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) desc1, b) desc2, c) desc3, d) desc4…………………………………………………………………………...117

Figure 25. Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'maximum number of reproductive cycles per year' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) semi, b) univ, c) pluriv………………………………………………………………………………………119

Figure 26. Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'number of reproductive cycles per individual' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) CPI, b) cpi2…………………………………………………………………………………………..120

Figure 27. Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'life duration of adults' across three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. lda1, b) lda2, c) lda3, d) lda4………………...122

Figure 28. Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'reproductive techniques' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) single, b) herma, c) two………….124

Figure 29. Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the trait 'oviposition sites' modalities across three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) surface, b) submerged, c) terrestrial, d) eggendo………………….126

Figure 30. Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the trait 'egg/egg mass' modalities across three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) eggfree, b) eggcement, c) eggprotected…………………………..128

Figure 31. Boxplots of the changes in the Community Weighted Mean (CWM) abundances of the trait 'aquatic stages' modalities across three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) aduandlar, b) andorlar, c) larandpup……………………………...130

Figure 32. Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'dissemination potential (all stages)' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) disslow, b) dissmedium, c) disshigh……………………………………………………………………………………132

Figure 33. Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'attachment to substrate of aquatic stages (excluding eggs)' across three different local

land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) swimmer, b) crawler, c) burrower, d) attached……………………………………………………………134

Figure 34. Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'feeding habits' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) shredder, b) scraper, c) deposit, d) filterfeed, e) predator, f) algalp…………………………………………………………………………………..137

Figure 35. Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'dietary preferences' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) specialist, b) moderatespe, c) generalist…....139

Figure 36. The relationship between DFS and the modalities of the trait 'body flexibility' in the Te Awanui/Tauranga Harbour basin. a) a) noflex, b) lowflex, c) highflex………………………………….142

Figure 37. The relationship between DFS and the modalities of the trait 'body form' in the Te Awanui/Tauranga Harbour basin. a) flattened, b) cylindrical, c) spherical, d) streamlined………………143

Figure 39. The relationship between DFS and the trait 'respiration of aquatic stages (not including eggs)' modalities in the Te Awanui/Tauranga Harbour basin. a) tegument, b) gill, c) plastron, d) aerial……....144

Figure 40. The relationship between DFS and the modalities of the trait 'maximum number of descendants per reproductive cycle' in the Te Awanui/Tauranga Harbour basin. a) desc1, b) desc2, c) desc3, d) desc4………………………………………………………………………………………145

Figure 41. The relationship between DFS and the modalities of the trait 'maximum number of reproductive cycles per year' in the Te Awanui/Tauranga Harbour basin. a) semi, b) univ, c) pluriv………………….146

Figure 42. The relationship between DFS and the modalities of the trait 'number of reproductive cycles per individual' in the Te Awanui/Tauranga Harbour basin. a) cpi1, b) cpi2……………………………....147

Figure 43. The relationship between DFS and the modalities of the trait 'life duration of adults' in the Te Awanui/Tauranga Harbour basin. a) lda2, b) lda3, c) lda4, d) lda5…………………………………....148

Figure 44. The relationship between DFS and the trait 'reproductive techniques' modalities in the Te Awanui/Tauranga Harbour basin. a) single, b) herma, c) two……………………………………….149

Figure 45. The relationship between DFS and the trait 'oviposition sites' modalities in the Te Awanui/Tauranga Harbour basin. a) surface, b) submerged, c) terrestrial, d) eggendo…………………150

Figure 46. The relationship between DFS and the trait 'egg/egg mass' modalities in the Te Awanui/Tauranga Harbour basin. a) eggfree, b) eggcement, c) eggprotected…………………………..151

Figure 47. The relationship between DFS and the trait 'egg/egg mass' modalities in the Te Awanui/Tauranga Harbour basin. a) aduandlar, b) aduorlar, c) larandpup…………………………....152

Figure 48. The relationship between DFS and the trait 'egg/egg mass' modalities in the Te Awanui/Tauranga Harbour basin a) disslow, b) dissmedium, c) disshigh……………………………153

Figure 49. The relationship between DFS and the trait 'egg/egg mass' modalities in the Te Awanui/Tauranga Harbour basin a) swimmer, b) crawler, c) burrower, d) attached…………………….154

Figure 50. The relationship between DFS and the modalities of the trait ‘feeding habits’ in the Te Awanui/Tauranga Harbour basin. a) shredder, b) scraper, c) deposit, d) filterfeed, e) predator, f) algalp...155

Figure 51. The relationship between DFS and the trait ‘dietary preferences’ modalities in the Te Awanui/Tauranga Harbour basin. a) specialist, b) moderatespe, c) generalist…………………………..156

1 Chapter 1: Introduction

1.1 Importance of freshwater ecosystems

Freshwater ecosystems are profoundly important to New Zealand. They are essential in sustaining biodiversity and provide multiple ecosystem services encompassing recreational, cultural and economic uses (Department of Conservation, 2023). However, New Zealand's freshwater ecosystems are under immense pressure from human activities. From the period 2016 to 2021, almost two-thirds of freshwater lotic ecosystems in New Zealand were ecologically compromised when compared to the National Objective Framework (NOF) guideline values listed in the New Zealand National Policy Statement for Freshwater Management 2020 (NPS-FM) (LAWA, 2023b; "NPS-FM 2020,"). Modelled Macroinvertebrate Community Index (MCI) scores indicate that 55% of New Zealand's stream length is in the NOF bands of C and D, reflecting moderate to severe ecological impairment (LAWA, 2023b). Data released in the reports 'Our Freshwater 2023 & 2020' and 'Environment Aotearoa 2022' produced by the Ministry for the Environment indicate what has caused this widespread degradation (Ministry for the Environment, 2022, 2023; Ministry for the Environment & NZ, 2020). Monitored sites with a low median score falling within the NOF band of D are located where approximately 80% of the surrounding land-use has been modified by humans (LAWA, 2023b). Whilst this statistic clearly indicates that anthropogenic changes are the leading cause of decline in the ecological health of New Zealand's streams and rivers, less certain is what key stressors or combination of stressors are driving this degradation.

1.2 Human impacts on stream ecosystems

Internationally, the impact of human land-uses on stream ecosystems is well established (Allan 2004). The principal mechanisms by which land use influences stream ecosystems are sedimentation, nutrient enrichment, contaminant pollution, hydrological alteration, clearing of riparian vegetation, and the loss of large wood (Allan 2004).

1.2.1 Sedimentation

Although sediment is a natural geological component of a stream ecosystem, human land-uses can contribute excessive amounts of sediment when managed improperly, leading towards many adverse effects (Allan, 2004). Sedimentation negatively affects stream ecosystems by degrading and altering the physical heterogeneity of benthic habitats, which has multiple negative implications for aquatic organisms. Inputs of fine inorganic sediment also cause additional adverse effects by directly impacting aquatic organisms (Allan, 2004).

Firstly, sediment erosion and runoff increase the water's turbidity due to increased suspended particle concentration, reducing water clarity (Shaw & Richardson, 2001). This diminished water clarity can make it difficult for aquatic organisms to perform activities essential for survival, such as foraging, mating and evading predators (Rowe & Dean, 1998).

Sedimentation also reduces light penetration into a stream's water column due to the increased turbidity (Davies-Colley et al., 1992). This results in a decrease in primary production due to the inhibition of photosynthesis by macrophytes and algae. Potentially, if the rate at which primary production occurs decreases significantly enough, it could result in a bottom-up trophic cascade, reducing the abundance and diversity of organisms in higher trophic levels (Osmundson et al., 2002).

Sedimentation can also cause direct physical harm to organisms within a stream ecosystem (Allan, 2004). The abrasive properties of sediment can scour algae (Francoeur & Biggs, 2006) and cause injury to invertebrates, fish, and amphibian species (McKenzie et al., 2020). Fine sediment particles can cause the gills of aquatic organisms such as fish and macroinvertebrates to become clogged, impairing their respiration ability (Lemly, 1982). Additionally, filter-feeding organisms such as freshwater mussels are also affected due to the impairment that fine sediment causes (Forbes-Green & Cyr, 2023). Sedimentation can also impact reproduction by burying eggs and impeding the porosity of gravels where fish spawn (Wood & Armitage, 1997).

Sediment deposition can smother and degrade streambed habitats by infilling riffles, runs, pools and interstitial spaces in gravel beds (Burdon et al., 2013). This habitat degradation can reduce the availability of suitable habitat space for macroinvertebrates and spawning grounds for fish species. Also, the availability of suitable streambed substrate can be reduced for macrophytes and periphyton, potentially exacerbating the damage that sedimentation causes by reducing the available habitat for primary production. Additionally, finer substrates can increase instability during high flows and reduce coarser substrates, further contributing to a decrease in the availability of suitable habitat space (Allan, 2004).

The impacts of sedimentation can cause the loss of macroinvertebrate taxa sensitive to pollution, such as those belonging to the orders of Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies), commonly known as the EPT taxa (Burdon et al., 2013). As a result, these sensitive taxa are replaced by pollution-tolerant taxa, such as the true flies belonging to the order Diptera (Allan, 2004; Myslinski & Ginsburg, 1977). Other fish and macrophyte species sensitive to sedimentation may also be lost (Allan, 2004; Myslinski & Ginsburg, 1977)

The effects of sedimentation are often cumulative, as streams impacted by sedimentation are usually affected through the frequent but small additions of sediment over time (Lake, 2000). This ‘ramp’-type disturbance can eventually cause harmful effects to become more pronounced when the accumulation of sediment exceeds the ability of a stream to store and process it. Unfortunately, when stream assimilative capacity is exceeded, it may lead to non-linear sedimentation impacts, making it difficult to identify this dominant stressor until the adverse effects overwhelm biological communities (Burdon et al., 2013).

1.2.2 Nutrient enrichment

Nutrient enrichment within streams and rivers typically occurs through poorly managed agricultural land-uses, as they allow high concentrations of dissolved inorganic nitrogen (DIN) and dissolved reactive phosphorus (DRP) to enter them via run-off during heavy periods of rain (Allan, 2004). Anthropogenic nitrogen may also enter streams through groundwater inputs (Burdon et al., 2013). Nutrient enrichment negatively affects stream ecosystems through the proliferation of periphyton and harmful/invasive macrophyte species, which cause multiple adverse effects through eutrophication (Smith et al., 2006).

Nutrient enrichment can rapidly accelerate the growth of periphyton and harmful/invasive macrophyte species. It also can cause a significant increase in primary production and biomass, leading to an overabundance of these plant species (Dodds & Smith, 2016). The proliferation of algae can cause a significant shift in the community composition of stream organisms from pollution-sensitive taxa to species that tolerate nutrient enrichment and organic pollution (Allan, 2004). Under eutrophic conditions, periphyton grows in thick, dense mats that smother and degrade benthic habitats (Allan, 2004). These mats can rapidly deplete dissolved oxygen levels within stream ecosystems through respiration and decomposition (Allan, 2004). Although algal and macrophytes produce oxygen through photosynthesis during daylight hours, these primary producers can deplete the quantity of oxygen through respiration overnight (Wilcock & Nagels, 2001). Also, as stream plants die through natural means, large quantities of organic matter can be deposited on the streambed. This creates a high Biological Oxygen Demand (BOD) due to the high rate at which microorganisms consume oxygen, decomposing this organic matter (Dodds, 2006). These impacts may be particularly relevant in larger, more open streams where abundant available light does not limit photosynthesis (Yuan, 2010). However, hydrological events, including bed movement and sediment scouring, may limit the effects of nutrients and other stressors (e.g., light, temperature) on primary production and stream eutrophication (Francoeur & Biggs, 2006).

1.2.3 Contaminant pollution

Human land-uses can also pose another stressor to stream ecosystems in the form of contaminant pollution (Allan, 2004). Because of their diverse modes of action and potency, toxic pollutants such as pesticides (e.g., herbicides, fungicides, insecticides), heavy metals, and other chemicals can negatively affect stream ecosystems in various ways (Knillmann & Liess, 2019).

Contaminant pollution's main direct effects on sensitive species are impaired performance and altered behaviour due to sublethal effects or elimination by lethality (Fleeger et al., 2003). At lower concentrations, toxins can interfere with aquatic organisms' such as macroinvertebrates and fish species endocrine and nervous systems. Such interference can impair reproductive success, growth rates and ability to progress through life stages, such as the emergent life stage of macroinvertebrates transitioning from an aquatic to a terrestrial environment (Allan, 2004). Aquatic organisms can exhibit behavioural changes in response to contaminant pollution, such as avoidance. For example, macroinvertebrate drift can increase in response to the presence of toxins, causing them to move downstream (Allan, 2004). In higher concentrations, contaminants have the potential to decrease the survival rate of aquatic organisms, leading to increased mortality, which is highlighted by the numerous studies that exhibit the extremely low LD50 (lethal dose to kill 50% of organisms in a test group) values (Knillmann & Liess, 2019).

Some of these pollutants, particularly heavy metals, can accumulate in biofilms and fine sediment on the streambed (Mahler et al., 2020; Suedel et al., 1994). These pollutant reservoirs can cause contaminants to be ingested or passively absorbed into the bodies of aquatic organisms as they leach into the surrounding water. Depending on the concentration and degree of exposure, this may cause acute or chronic effects. Bioaccumulation is when toxins are absorbed into an organism's body throughout its lifespan if continually exposed and can occur (Goodyear & McNeill, 1999). If the frequency of exposure and or concentration of toxins is high enough, it can lead to biomagnification, the process in which the concentration of toxins increases as the trophic level increases (Suedel et al., 1994). In other words, biomagnification occurs when toxins present in organisms at lower trophic levels are transferred to organisms at higher trophic levels through consumption, resulting in apex predators having the highest concentration of toxins.

Contaminant-induced changes in behaviour, competition, and predation/grazing rate can alter species' abundances or community composition (Fleeger et al., 2003). If toxic effects are sufficient to eliminate secondary consumers, a trophic cascade may occur where primary producers increase due to the loss of grazing organisms that consume them (Fleeger et al., 2003). These indirect effects mediated through biotic interaction may enhance, mask or spuriously indicate direct contaminant effects.

1.2.4 Hydrological alteration

Human land uses negatively affect stream ecosystems via hydrological alterations (Allan, 2004). Changes in drainage patterns through catchment modification, dam construction, water diversion and channelization can alter the natural flow regimes of streams (Poff et al., 1997). Flow is considered a master variable affecting multiple stream ecosystem health components (Allan, 2004; Poff et al., 1997).

The altered balance between runoff and evapotranspiration due to changes in catchment land uses can significantly alter stream hydrology (Allan, 2004). The loss of woody vegetation and replacement with pasture or impervious surfaces mean water can run off the land more freely, increasing flood magnitude and frequency (Allan, 2004). Also, impervious surfaces prevent rain from infiltrating into the ground at a sufficient rate, and in combination with the loss of vegetation, the rate of evapotranspiration is reduced. Less infiltration can reduce groundwater recharge and lower base flows (Allan, 2004). Base flow is defined as the sustained flow in streams between precipitation events. Another human activity affecting base flow is excessive water abstraction for agricultural purposes, which removes water from the stream faster than it can be recharged (Allan, 2004).

Changes to catchment hydrology can alter channel morphology and the natural dynamics of streams (Poff et al., 1997; Wohl et al., 2015). Increased runoff can increase stream erosion rates, alter hydrogeomorphology, and lead to less resilient riverine habitats (Allan, 2004). Natural flooding events play an essential role in maintaining the ecological health of not only in-stream ecosystems but also riparian ecosystems through processes such as nutrient cycling, sediment transport, and the removal of contaminants. Due to hydrological alterations caused by human activities, these processes can be disrupted. Flood regimes may become more severe with slower community recovery (Collier & Quinn, 2003). Diminished flows during summer may lead to eutrophication, excessive sedimentation, and contamination within stream ecosystems (Allan, 2004).

These hydrological alterations can lead to habitat degradation, including sedimentation, eutrophication and warming (Allan, 2004). Anthropogenic structures that disrupt longitudinal connectivity, such as dams, can create migratory barriers to aquatic organisms like fish. These migratory barriers can interrupt life cycles, reduce genetic diversity, and decrease the abundance of aquatic species (Fausch et al., 2002; Joy & Death, 2013). This issue is particularly problematic for New Zealand freshwater fish species since diadromy (needing passage between freshwater and saltwater to complete their life cycle) is thought to be obligatory in 13 species and facultative in six out of approximately 52 genetically distinct species (Joy & Death, 2013).

1.2.5 Clearing of riparian vegetation

Human land-uses frequently negatively affect stream ecosystems by removing riparian vegetation, significantly reducing stream shading and increasing stream water temperatures (Allan, 2004). Naturally, the shade provided by riparian vegetation helps regulate the stream's water temperature (Quinn et al., 1997). However, removing riparian vegetation reduces shading and, in turn, leads to elevated water temperature levels. This can result in increased stress on aquatic organisms, leading to mortality, impairment of their ability to perform ecological functions and a disruption of the life cycles of temperature-sensitive species (Allan, 2004). Increased light and temperatures due to the loss of riparian shading in small streams can exacerbate the effects of eutrophication by allowing primary producers to accrue excessive biomass (Quinn et al., 1997).

Removing riparian vegetation can cause a loss of bank stability, leading to increased erosion (Allan, 2004). Riparian vegetation naturally helps to stabilize streambanks by increasing soil stability through their root structure. However, removing riparian vegetation causes bank instability, increasing its susceptibility to erosion and, in turn, increasing sedimentation rates (Allan, 2004). Removing riparian vegetation can alter trophic connectivity between stream and terrestrial food webs (Champagne et al., 2022). Vegetation loss reduces organic matter inputs, such as leaf litter, a critical food source and habitat for aquatic organisms (Allan, 2004). This can cause further changes in stream trophic structure by reducing the detrital food web channel reliant on allochthonous inputs. The loss of terrestrial organic matter leads to changes in the abundance and distribution of species that use this food source. Additionally, removing riparian vegetation can alter the quality and quantity of dissolved organic carbon present in the water body of the stream (Pisani et al., 2020). This can change stream water chemistry and, in turn, further alter ecological processes such as carbon cycling (Allan, 2004).

1.2.6 The loss of large wood

Another negative consequence of human land use is the loss of large wood in streams (Allan, 2004). Although this is related to the removal of riparian vegetation, the loss of large wood has lasting negative impacts on streams, including loss of habitat and altered geomorphology (Allan, 2004). Large wood provides a stable habitat for aquatic organisms that can be particularly important in North Island streams (Collier & Halliday, 2000). Woody material naturally provides attachment sites for algae, macroinvertebrates, and fish. Xylophagous organisms can consume the wood, and macroinvertebrates and fish can use the large wood as a stable structure for reproduction and refugium. However, the loss of large wood reduces habitat and niche availability for aquatic organisms, reducing biodiversity and abundance. This may negatively affect the structure and function of stream communities at larger spatial scales through the disruption of food webs and altered nutrient processing (Allan, 2004).

The loss of large woody material also impacts stream geomorphology. Large wood is an important structural element that enhances a stream's natural capacity to store organic matter and sediment (Allan, 2004). The loss of this natural storage element can diminish a stream's ability to effectively store and trap sediment and organic material while also reducing habitat heterogeneity (Bisson et al., 1992). This reflects the influence of large woody materials on the hydraulic characteristics of a stream (Allan, 2004). Naturally, large woody material plays a role in dissipating the energy contained in a stream's water flow. However, its removal can alter the flow patterns of a stream due to the increased velocity of water. This can cause stream bank erosion and disrupt sediment transport dynamics.

Increased water flow velocity can also cause channelization (straightening of the stream channel). Compounding these impacts is the loss of large woody material, reducing bank stability (Allan, 2004). Along with riparian vegetation, large woody material improves bank stability by providing a strong element that helps stabilise bank sediments. However, its removal can erode stream banks and increase sedimentation rates (Allan, 2004). Both streambank erosion and channelization negatively affect the distribution of habitats within the stream by increasing sedimentation and decreasing habitat heterogeneity (Allan, 2004). The loss of large woody material can also increase sedimentation further downstream (Allan, 2004).

1.2.7 Other impacts

Other human impacts included losses of native biodiversity through the introduction of exotic species (Mayfield et al., 2021; Milardi et al., 2019). These non-native species can aggressively displace native species by outcompeting them for the same resources, such as food and habitat, resulting in a reduction in the diversity and evenness of species as exotic species become more established within the ecosystem (Mayfield et al., 2021; Milardi et al., 2019).

Human impacts are not mutually exclusive and may interact in complex ways. For instance, the natural flow regime paradigm posits that flow is a ‘master’ variable controlling water quality, ecosystem functioning, physical habitat, and biodiversity, including biotic interactions that contribute to the ecological integrity of streams and rivers (Poff et al. 1997). An example of how one of these components can indirectly influence another can be observed in the relationship between the flow regime, physical habitat, and water quality of a stream ecosystem. Physical habitat indirectly affects water quality by removing riparian vegetation, which can cause increased erosion, releasing fine sediment into the water and increasing turbidity. Flow variability can induce and exacerbate such effects, including floods and low summer flows (Allan, 2004; Poff et al., 1997).

1.3 Land-use impacts on streams and rivers in New Zealand

In New Zealand, the proportions of catchment areas in urban and pastoral land uses have been consistently positively correlated with contaminant levels in water bodies and negatively correlated with ecological health indicators (Larned et al., 2019). Data released by Land Air & Water Aotearoa (LAWA) displaying trends in ecological health indicators (MCI scores) from the period 2016−2021 provide further insights into the effects that human land modifications are having on stream ecosystems in New Zealand (LAWA, 2021, 2023b). Approximately 80% of 604 monitoring sites situated in pastoral-land use fall within the National Objective Functions (NOF) band of C and D, along with 50% of 49 sites situated in exotic forest land-use and 78% of 52 sites situated in urban land-use (LAWA, 2021). In comparison, only approximately 30% of 312 sites situated in native land-use fall within the NOF band of C and D. To provide further context to these results, native vegetation comprises 48% of New Zealand's channel length, 45% is comprised of pastoral land-use, exotic forests comprise 5% and urban streams only comprise 1%. When comparing these results with the proportion of each land use, it becomes apparent that although all human-modified land-use types comprise a smaller area than native vegetation, they are significantly more degraded. This reinforces that anthropogenic drivers are the leading cause of ecological degradation in New Zealand streams and rivers (LAWA, 2021, 2023b).

These results have been projected to worsen. Results published by LAWA (2021) on the national trends for MCI scores over 10 years from 2012 to 2021 show a decline in the number of sites within the NOF band of A and B and an increase in the number of sites in the C and D band (LAWA, 2021, 2023b). One major contributor to this decline is land-use intensification, which involves increasing the intensity and or frequency of activities such as irrigation, fertilization or increasing stock numbers to enhance the land's productivity (Ministry for the Environment, 2022, 2023; Ministry for the Environment & NZ, 2020; Snelder et al., 2022; Snelder et al., 2020). New Zealand's water quality changes were closely associated with the proportion and intensity of pastoral, agricultural, plantation, and forestry land uses (Snelder et al., 2022).

The Ministry for the Environment's annual environmental report 'Our Freshwater 2023' provides key statistics highlighting the issue of land-use intensification in New Zealand (Ministry for the Environment, 2023). New Zealand had one of the highest rates of agricultural land intensification in the Organization for Economic Cooperation and Development (OECD) countries between 2015 and 2022. Specifically, from 1996 to 2018, approximately 60,000 hectares of exotic grassland shifted from being classified as low-producing to high-producing. In comparison, only 3,500 acres of high-producing grassland were converted to low-producing. The number of dairy cattle also increased by 61% between 1996 and 2014, falling by only 5% by 2018. Additionally, from 2002 to 2019, the area of irrigated land significantly increased by 91% (384,000 to 735,000 hectares), with 73% of the increase related to dairy farming, 18% to grain, fruit and vegetable farming and 9% to sheep and beef. These statistics highlight the prominence of the rise in land-use intensification in New Zealand (Ministry for the Environment, 2023).

1.4 Assessment of stream ecosystem health

The deterioration of ecosystems worldwide, including in New Zealand, has created the need for effective environmental monitoring and the development of ecosystem health indicators that can be used widely (Costanza & Mageau, 1999). Costanza and Mageau (1999) provide a practical conceptual framework for understanding ecosystem health. They recognise that whilst the concept of ecosystem health represents a desired endpoint of environmental management, it requires adaptive, ongoing definition and assessment. Thus, Costanza and Mageau (1999) sought to define ecological health in a way that is more conducive and applicable to environmental monitoring. They define ecological health as a 'comprehensive, multiscale, dynamic and hierarchical measure of system organization, vigour and resilience'. Resilience refers to an ecosystem's ability to maintain its organization (structure) and vigour (function) in response to disturbance. There are two principal metrics of an ecosystem's resilience. The first metric is the magnitude of a disturbance that an ecosystem can endure (resistance). The second metric is the duration of time in which an ecosystem can recover from a disturbance (recovery). By incorporating resilience, among other factors, Costanza and Mageau's (1999) definition of ecosystem health is broad. It acknowledges that multiple aspects contribute to an ecosystem's ecological integrity (Costanza & Mageau, 1999).

Ecosystem health concepts have also turned to non-Western traditions. In New Zealand, a concept of ecological health that guides the implementation of the National Policy Statement for Freshwater Management 2020 is called *Te Mana o Te Wai* (Ruru et al., 2022; Tipa & Teirney, 2006). This term is grounded in Te Ao Māori (the Māori worldview) and recognises the vital importance of clean, healthy water for protecting biodiversity and ecosystem services. Implicitly, *Te Mana o Te Wai* refers to the importance of adopting a holistic approach when managing freshwater ecosystems by acknowledging their complexity and connections with land and sea. Another complementary principle derived from Te Ao Māori is *ki uta ki tai,* which translates to 'from the mountains to the sea' (Virens, 2023). This principle has been adopted as a framework for recognising water's complex interactions as it flows down the catchment from its upland sources to the coast. *Ki uta ki tai* also reflects a holistic form of kaitiakitanga (stewardship) that recognises the journey wai (water) takes from its mātāwai (headwaters) to the taiāpure (coastal fishing grounds) that form an important part of customary food gathering for Māori (Virens, 2023). Both principles (*Te Mana o Te Wai* and *ki uta ki tai*) incorporate mātauranga Māori (Māori knowledge) and a Western scientific worldview since aspects of ecosystem health can not only be quantified by established norms but also through the spiritual and cultural aspects important to Māori (Ruru et al., 2022; Tipa & Teirney, 2006). The desire of Māori to undertake freshwater biomonitoring and see their cultural values reflected in these approaches has led to monitoring tools that explicitly characterise ecosystem health using mātauranga Māori (Tipa and Teirney, 2006).

Bioassessment remains a particularly effective assessment tool for ecosystem health, as it provides physical, chemical, and biological measurements necessary to determine ecological integrity (Costanza & Mageau, 1999; Jha et al., 2022; Lu et al., 2015; Ulrich et al., 2023). In a review of freshwater and estuarine studies (O'Brien et al., 2016), the authors found a strong bias towards fish and macroinvertebrate community metrics (e.g., diversity, abundance, and composition) used in bioassessment. Given the human impacts on ecosystems globally, there is a strong need for the development of effective and accurate biomonitoring tools (Costanza & Mageau, 1999; Fahrig, 1992; Gray et al., 2014; Holt & Miller, 2010; Makiola et al., 2020; Menezes et al., 2010; Parmar et al., 2016). The accuracy of a biomonitoring tool depends on its sensitivity to stressors (Fahrig, 1992; Makiola et al., 2020). This accuracy can be significantly increased by ensuring that biomonitoring is applied at the correct spatio-temporal scales (Fahrig, 1992; Jackson et al., 2021; Makiola et al., 2020). Drivers of ecological degradation may go undetected because organisms respond to disturbances at a different scale than observed (Jackson et al., 2021; Sheldon et al., 2012).

Biomonitoring must be conducted at appropriate spatial and temporal scales to adequately capture ecological degradation and organisms' responses to it (Fahrig, 1992; Makiola et al., 2020; Sheldon et al., 2012). As drivers of ecological degradation such as pollution or impoundments (e.g., dams) can influence an ecosystem over varying spatial and temporal scales, it is crucial to identify the maximum extent they hold influence over an ecosystem and, in turn, its organisms (Fahrig, 1992; Krajenbrink et al., 2022; Villeneuve et al., 2018; Waite, 2013; Waite et al., 2014). Spatial and temporal scales above the maximum extent of disturbance will not accurately capture its effects on an ecosystem, but the scales below this maximum extent will (Fahrig, 1992; Jackson et al., 2021; Sheldon et al., 2012). This problem means that determining the optimal biomonitoring scale can be difficult, expensive, and time-consuming (Makiola et al., 2020). However, in the context of my study, I have selected the most appropriate spatial scales for detecting anthropogenic drivers of ecological degradation by examining a range of relevant literature (Makiola et al., 2020; Villeneuve et al., 2018; Waite, 2013).

Annual sampling is the most appropriate temporal scale to sample macroinvertebrates in New Zealand (Clapcott et al., 2017; Harding et al., 2009; Stark et al., 2001; Stark & Maxted, 2007; Storey & Quinn, 2011; Winterbourn et al., 1981). This is for two main reasons: generation time and seasonal variability (Fahrig, 1992; Storey & Quinn, 2011; Winterbourn et al., 1981). An organism's generation time is defined as the ‘average interval between an individual's birth and its offspring’ (Fahrig, 1992). Seasonal variability in this context is defined as the ‘presence or absence of a macroinvertebrate species across seasons due to their respective life-history patterns’ (Storey & Quinn, 2011; Winterbourn et al., 1981).

As most New Zealand macroinvertebrate species have an average generation time of one year, it means that this is the most appropriate temporal scale to sample, as it captures the majority of responses exhibited by macroinvertebrate communities in response to anthropogenic stressors in the preceding year (Fahrig, 1992; Stark et al., 2001; Storey & Quinn, 2011). Also, previous studies (Harding et al., 2009; Stark et al., 2001; Stark & Maxted, 2007; Storey & Quinn, 2011; Winterbourn et al., 1981) have shown that New Zealand macroinvertebrate communities show little seasonal variability in their species composition. This can be explained by the finding that New Zealand macroinvertebrate community composition is more significantly influenced by changes in their environmental conditions than those caused by seasonal variation (Storey & Quinn, 2011; Winterbourn et al., 1981). This finding is reinforced by Winterbourn (1981): ‘New Zealand stream insects have broad habitat requirements, lack feeding niche differentiation, and possess flexible, poorly synchronised life histories’ and ‘abiotic factors have been of paramount importance in the shaping of New Zealand stream communities’ (Winterbourn et al., 1981). New Zealand macroinvertebrate species have life-history patterns that are flexible enough to allow them to be present and sampled throughout all seasons annually, with any differences in species composition being more likely influenced by changes in their environmental conditions than those caused seasonal variations (Storey & Quinn, 2011; Winterbourn et al., 1981).

Previous studies have shown that a reach (e.g., 10^1 m) is the most appropriate spatial scale to sample macroinvertebrates, but depending on the disturbance and/or source of anthropogenic stress, an ‘eco-region’ spatial scale may be the best to observe changes in macroinvertebrate community composition (Frissell et al., 1986; Harding, 1994; Winterbourn & Harding, 1997).

Streams exhibit high geomorphological variability due to their hierarchical, linear and dendritic nature (Fitzpatrick et al., 1998; Parsons et al., 2003), but for simplicity, a reach system is typically defined at the 10^1 m scale (Frissell et al., 1986). However, to explicitly account for differences in stream size (i.e., channel width), the exact length of a stream reach can be determined by multiplying its wetted width (the length between one bank to another where it is in contact with water) by a constant of 20 (Harding et al., 2009; Stark et al., 2001). This constant was chosen to account for the ‘extra length’ that a meandering stream might add through its sinuosity (wavelength), which scales with stream size, thus ensuring the reach sampled represents all its different habitat types (Fitzpatrick et al., 1998).

The reach system is the most appropriate spatial scale to sample macroinvertebrates because it represents a stream community composition in terms of diversity and abundance by including all microhabitats (Barquín & Death, 2006; Clapcott et al., 2017; Sheldon et al., 2012; Stark et al., 2001). At a reach scale, stable microhabitats can retain a greater quantity and diversity of food resources and support greater significant habitat heterogeneity. Habitat and resource heterogeneity increases the number of available niches, supporting a higher abundance and diversity of macroinvertebrate species (Barquín & Death, 2006). From a management perspective, reaches are the most efficient spatial scale to sample as they are they encompass an area large enough to capture the true representation of a stream community but not so large that sampling would be expensive and time-consuming (Clapcott et al., 2017; Fitzpatrick et al., 1998; Harding et al., 2009; Stark et al., 2001; Stark & Maxted, 2007).

An eco-region is characterized by its six defining environmental aspects: meteorological, climatic region, rainfall, relief (slope), vegetation, soil and geology (Harding, 1994; Winterbourn & Harding, 1997). An eco-region is similar to the regional boundaries set by the New Zealand Government (such as the Bay of Plenty), which are based on the extent of the drainage basin area ("Local Government Act," 2002). However, some regional boundaries can occupy multiple eco-regions (Harding, 1994). For instance, Canterbury occupies 3 eco-regions, “East Coast Plains”, “Banks Peninsula”, “High Country” and the “Southern Alps” (Harding, 2004). An eco-region scale may be the most optimal spatial scale to observe changes in macroinvertebrate community composition, as it captures a greater representation of significant spatial heterogeneity (e.g., geology, climate, land use) contributing to environmental filtering whilst minimising responses by different macroinvertebrate species constrained by their dispersal abilities and biogeography (Harding, 1994; Li et al., 2021; Winterbourn & Harding, 1997).

This suggestion is evidenced in Harding (1994), who identified that, in New Zealand, the species assemblage of a macroinvertebrate metacommunity within an eco-region was most strongly affected by the following three factors within an eco-region: historical biogeographical events, climatic conditions, and; land use (Harding, 1994). Essentially, these three factors act as filters, selecting only species compatible with the environmental conditions of that eco-region (Harding, 1994; Li et al., 2021; Winterbourn & Harding, 1997).

However, changes in the assemblage of macroinvertebrate metacommunities can be better understood when comparing the metacommunities of multiple eco-regions (Harding, 1994; Li et al., 2021; Winterbourn & Harding, 1997). This is because assemblage changes caused by environmental filtering (biogeographical events, climatic conditions and land use) and dispersal limitations become more prominent when compared at a broader scale. For example, this study found that taxonomic diversity was significantly higher in eco-regions that had native forests as their predominant land use in comparison to those characterized by pastoral land use (Harding, 1994; Li et al., 2021; Winterbourn & Harding, 1997).

1.5 Stream macroinvertebrates as a biomonitoring tool

The appropriateness of a biomonitoring tool is strongly determined by its ability to measure ecological health. A tool that meets this criterion and can be applied to lotic ecosystems involves aquatic macroinvertebrates. Macroinvertebrates are found in almost every stream and river around the world. By convention, macroinvertebrates are defined as any water-dwelling invertebrate retained by a 0.5mm mesh (Hauer and Resh, 2007). However, this definition is often relaxed for early life stages; thus, macroinvertebrates can include finer mesh sizes such as 0.25 mm. A macroinvertebrate's maximum size can exceed 15 cm in length (Hauer & Resh, 2007), with the largest recorded taxon in New Zealand by length being kōura/freshwater crayfish (*Paranephrops* sp.).

There are over 675 species of macroinvertebrates within New Zealand belonging to 12 Linnaean orders: Ephemeroptera, Plecoptera, Trichoptera, Diptera, Coleoptera, Odonata, Megaloptera, Crustacea, Mollusca, Oligocheta, Platyhelminthes and Collembola (Grainger et al., 2018). The Ephemeroptera, Plecoptera and Trichoptera (EPT taxa) are the orders most significant to stream biomonitoring due to their high sensitivity to pollution (Clapcott et al., 2017; Stark et al., 2001; Stark & Maxted, 2007). Due to the degradation of New Zealand's freshwater habitats, 26% of New Zealand's macroinvertebrate fauna had a conservation status of critical, threatened, endangered or at risk as of 2018 (Grainger et al., 2018). Of the 177 Threatened and At Risk taxa, 48 were listed as Nationally Critical, 14 as Nationally Endangered, 16 as Nationally Vulnerable, 10 as Declining, and 89 as Naturally Uncommon (Grainger et al., 2018).

Macroinvertebrates are an effective bioindicator for assessing the ecological health of stream ecosystems (Charvet et al., 1998; Dolédec et al., 2011; Stark & Maxted, 2007). They are present in almost every stream type and play an essential role in transferring energy from basal resources to higher consumers (Wallace & Webster, 1996). Macroinvertebrates are strongly influenced by changes in local abiotic/biotic conditions (environmental filtering), making them excellent bioindicators for a wide variety of stressors (Liu et al., 2023; Menezes et al., 2010; Myslinski & Ginsburg, 1977; Rideout et al., 2022). Macroinvertebrates are often held in higher regard than physiochemical approaches for estimating stream health because they reflect environmental conditions over a larger temporal and spatial scale (Charvet et al., 1998; Dolédec et al., 1999).

Macroinvertebrates play an essential role in the nutrient cycling of stream ecosystems by processing various sources of allochthonous and autochthonous organic matter, releasing the nutrients contained, and transporting them laterally and longitudinally for other organisms to utilize (Vannote et al., 1980). Their widespread presence allows them to be used as a biomonitoring tool in any geographic region (Charvet et al., 1998; Dolédec et al., 1999; Menezes et al., 2010). They can be used to assess the ecological health of a stream ecosystem by allowing community organization (structure), vigour (function), and resilience to be measured (Costanza & Mageau, 1999). The organization of a macroinvertebrate community is represented by its abundance and species diversity, vigour is represented by its functional diversity, and resilience can be represented by the change in response to external stressors (Costanza & Mageau, 1999).

Macroinvertebrates are effective bioindicators because of their wide range of sensitivities towards pollution and habitat degradation (Clapcott et al., 2017; Myslinski & Ginsburg, 1977; Stark & Maxted, 2007) For example, macroinvertebrates species belonging to the orders Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies) are highly sensitive to pollution, thus, in general, the EPT taxa combined can be valuable indicators of ecosystem health (Clapcott et al., 2017). In contrast, macroinvertebrates belonging to the orders Mollusca and Oligochaetes are relatively tolerant of pollution. Due to differing tolerance levels to stressors, the absence or presence of specific species within a stream ecosystem (e.g., environmental filtering) can indicate how degraded it is (Clapcott et al., 2017).

Macroinvertebrates integrate physical, chemical, and biological measurements necessary to determine ecosystem health (Charvet et al., 1998; Menezes et al., 2010). This difference highlights several weaknesses in physicochemical indicators, such as water quality, also used to monitor stream ecosystems (Charvet et al., 1998; Keke et al., 2021). Water quality measurements only reflect changes over a small temporal scale. For example, environmental processes that occur over a short temporal scale, such as sunlight, can cause water quality to change over a day. Further, physico-chemical indicators do not accurately indicate the biological health of an ecosystem. A wide range of factors can determine community integrity, and water quality measurements that provide a ‘snapshot’ in time may fail to adequately characterise all the sources of degradation affecting an ecosystem. This is why it is essential to use biological indicators alongside environmental indicators, as they better reflect the ecological health of communities over larger temporal scales (Charvet et al., 1998; Keke et al., 2021).

Historically, the Macroinvertebrate Community Index (MCI) has been the primary biomonitoring metric involving the use of macroinvertebrates in New Zealand (Clapcott et al., 2017; Stark et al., 2001). The MCI is a metric that was developed to indicate organic and nutrient pollution and has similarities to other indices used around the world, such as the Biological Monitoring Working Party (BMWP) scoring system in Great Britain (Clapcott et al., 2017; Medupin, 2020) The MCI is an effective and invaluable tool for assessing the overall health of stream ecosystems (Stark et al., 2001). It functions by assigning a pollution tolerance value to each macroinvertebrate species ranging from 1 (highly tolerant of pollution) to 10 (highly intolerant of pollution), summing the value of each species’ tolerance value, dividing the total score by the number of species present and then multiplying it by a constant to provide a final score ranging between 0 (poor ecological health) to 200 (excellent ecological health); (Stark et al., 2001).

However, there are some fundamental issues with the MCI. Firstly, this approach requires a high level of taxonomic knowledge of New Zealand macroinvertebrates, as incorrect identification of a species can result in the wrong tolerance value being included in the final calculation, potentially invalidating the results (Wagenhoff et al., 2016). Although initially designed to measure organic pollution, the MCI struggles to identify specific pollutants or disturbances and works best as a general indicator of ecological impairment (Stark, 1985). Consequently, it is more phenomenological than mechanistic in its ability to describe invertebrate-stressor relationships (Clapcott et al., 2017). It is crucial that we develop more effective biomonitoring tools that successfully link indicators to different types of stressors, is underpinned by a mechanistic understanding of stream ecology, are robust to relatively low levels of taxonomic expertise, and are applicable across a broad range of environments (Wagenhoff et al, 2016; Menezes et al. 2010).

1.6 Functional traits of stream macroinvertebrates

An ideal biomonitoring tool could be created using functional traits of key freshwater organisms such as macroinvertebrates. Functional trait-based ecology describes the relationship between a species' traits and its environment (Menezes et al., 2010). Those authors define a trait as a "characteristic that reflects a species' adaptation to its environment". Traits are commonly separated into two groups: biological traits and ecological traits (Menezes et al., 2010). Biological traits affect an organism's life cycle, physiological and behavioural characteristics such as maximum body size, lifespan, feeding and reproductive strategies, and mobility. Ecological traits are related to habitat preferences, like pH and temperature tolerances, tolerance to organic pollution, and biogeographic distribution. Functional traits are defined as "morphological biochemical, physiological, structural, phenological or behavioural characteristics expressed in the phenotypes of individual organisms and considered relevant to the response of such organisms to the environment and/or their effects" (Menezes et al., 2010; Usseglio-Polatera et al., 2000a).

Functional traits comprise functional diversity, which is defined as the 'diversity of functional identities of organisms in a community based on morphological, behavioural and life-history traits, which are directly linked to an organism's ecological performance (Nock et al., 2016).

Historically, there has been extensive research into the relationship between a species' traits and its habitat (Charvet et al., 1998; Dawson et al., 2021; Dolédec et al., 2011; Menezes et al., 2010). There is a growing interest in the potential for a species' traits to be used as a biomonitoring tool to detect the effects of anthropogenic stressors (Menezes et al., 2010). The origin of functional traits-based research broadly began in the 1970s when the plant ecologist Phil Grimes hypothesised that plants developed different strategies to cope with competition, stress and disturbance, strongly influencing community composition (Grime, 1974). Southwood (1977) used these ideas to help develop the habitat templet model, which hypothesises that the characteristics of a habitat select species with the most compatible traits as a consequence of past evolutionary processes (Southwood, 1977).

These ideas have strongly influenced subsequent theoretical developments in stream ecology, with the seminal River Continuum Concept (RCC) drawing on the habitat templet model to explain how changing environmental conditions moving down a river network predictably influence the composition of stream communities (Vannote et al., 1980). The habitat templet model has been further tested in the Rhône (France & Switzerland) and Taieri River (New Zealand), with these studies providing strong supporting evidence (Bournaud et al., 1992; Chevene et al., 2006; Dolédec & Chessel, 1994; Statzner et al., 1994; Townsend et al., 1997). It was these studies that lead to the foundation of the use of functional traits as a biomonitoring tool (Charvet et al., 1998; Chevene et al., 2006; Dolédec et al., 2011; Dolédec et al., 1999; Menezes et al., 2010).

The work needed to develop the trait-based framework motivated the study by Doledec et al. (1999), which compared taxonomic approaches to freshwater monitoring with a functional trait-based approach. They proposed that an ideal biomonitoring tool must be applicable across geographic regions, able to identify specific stressors, reliable, and be developed using sound ecological concepts grounded in theory. Their study was successful as functional traits provided a better indication of anthropogenic stressors than taxonomic indicators since the former was not as affected by changes in spatial scales (Dolédec et al., 1999). A similar result was observed in New Zealand streams affected by agriculture, as functional traits could discriminate the impacts of pastoral land uses better than taxonomic indicators (Dolédec et al., 2006). Using traits can help disentangle stressor impacts and provide a more general understanding of macroinvertebrate community structure (Lange et al., 2014).

1.7 Aims and hypotheses

New Zealand's freshwater ecosystems are increasingly in poor ecological health (LAWA, 2021; Ministry for the Environment, 2022, 2023; Ministry for the Environment & NZ, 2020). One area that has been particularly affected by recent land-use intensification is the Te Awanui or Tauranga Harbor catchment located in the Western Bay of Plenty region (Bay of Plenty Regional Council, 2023b; Sinner et al., 2011). Te Awanui has strong cultural significance to local Māori and is a traditional source of food for whānau, hapū and iwi living in the area. The harbour covers an area of 210 square kilometres and is one of New Zealand's largest estuaries (Bay of Plenty Regional Council, 2023b; Sinner et al., 2011). It has been identified as an outstanding natural feature with high ecological significance (Van Meeuwen-Dijkgaar et al., 2010). The wider catchment covers an area of almost 1,300 square kilometres and contains 27 major rivers and 46 minor streams (Bay of Plenty Regional Council, 2023b; Sinner et al., 2011). The streams in the Te Awanui catchment provide habitat for a wide range of native freshwater species, including diadromous galaxiids such as kōaro (*Galaxias brevipinnis*) and īnanga (*G. vulgaris*) (Goodman, 2018) and the rare leptophlebiid mayfly species *Isothraulus agnitus* (Snelder et al., 2019). The Kamai and Mamaku ranges inland of Te Awanui hold important native forest types (Singers & Rogers, 2014) that support populations of endangered native birds (Van Meeuwen-Dijkgaar et al., 2010), such as North Island brown kiwi (*Apteryx mantelli*) and kōkako (*Callaeas wilsoni*).

However, these ecological values are under threat by rapid population growth and land-use intensification. The Bay of Plenty had the greatest population growth out of all regions in New Zealand (2.8% or 9,100 people) between the period 2019 to 2020 (StatsNZ, 2020). The Western Bay of Plenty has contributed significantly towards this growth (Houpt, 2022), which has been driven by economic factors, including the ongoing success of kiwifruit (*Actinidia* spp.) orcharding (Zespri, 2021). There has been an increasing trend for pastoral land to be converted to horticultural land uses, including kiwifruit orchards (Miller et al., 2015; Van Rossen & Adam, 2015). The effects of kiwifruit orcharding on stream ecosystems relative to other activities, such as pastoral agriculture in the Bay of Plenty, remain poorly understood (Van Rossen & Adam, 2015).

One of the most significant issues faced by the Te Awanui catchment is sedimentation caused by the rapid intensification of land uses between the forested uplands and the coast (Bay of Plenty Regional Council, 2023b; Hume et al., 2010a; LAWA, 2018, 2021; Parshotam et al., 2008; Sinner et al., 2011; Snelder et al., 2019; Van Rossen & Adam, 2015). A report published by the National Institute for Water and Atmospheric Research (NIWA) in 2010 explored the issue of sedimentation within Tauranga Harbour (Hume et al., 2010a). The report identified increased sedimentation rates, such as in the Waimapu River sub-catchment of Te Awanui, which contributes 99% of the sediment load in Waimapu Bay and 26% in Welcome Bay (Hume et al., 2010a). Sedimentation has adverse effects on not just stream ecosystems but also receiving estuarine/marine ecosystems in the Te Awanui/Tauranga Harbor (Hume et al., 2010a; Parshotam et al., 2008; Sinner et al., 2011). Furthermore, climate change impacts such as increased rainfall intensity likely exacerbate sedimentation rates caused by land-use intensification within the Te Awanui catchment (Hume et al., 2010a). Sedimentation rates within the Te Awanui catchment have been modelled to increase by 40% by 2051 (Hume et al., 2010a).

The impact of pastoral land uses on streams in New Zealand is well recognized, including their role in increased sedimentation and nutrient enrichment (Allan, 2004; Burdon et al., 2013; Dolédec et al., 2011; Hume et al., 2010a; Parshotam et al., 2008; Phillips & Smith, 2018; Quinn et al., 1997; Rowe & Dean, 1998; Scarsbrook & Townsend, 1993; Sinner et al., 2011; Snelder et al., 2022; Snelder et al., 2020; Townsend et al., 2008). In contrast, little is known about how horticultural activities such as kiwifruit orcharding impact streams and rivers in New Zealand, including those in the Te Awanui catchment (Van Rossen & Adam, 2015). Considering the rapid expansion of horticultural land uses in the Bay of Plenty, it is important that this knowledge gap is further explored (LAWA, 2018; Van Rossen & Adam, 2015; Zespri, 2021). For example, many studies have identified that pesticide use associated with horticultural land use poses a significant threat to stream ecosystems. However, little is known about pesticides' impacts on stream ecosystems in New Zealand (Close, 1993; Hageman et al., 2019; Rajmohan et al., 2020; Syafrudin et al., 2021). As overseas studies have identified the threat that pesticides pose, and more than a ton of highly bioactive pesticides are used in New Zealand each year, it is important to explore further the potential role they play in the degradation of stream ecosystem health (Bighiu et al., 2020; Hageman et al., 2019; Knillmann & Liess, 2019b; Mac Loughlin et al., 2022; Sharma et al., 2019; Stone et al., 2014; Tudi et al., 2021)

Other potential impacts could include nutrient enrichment, loss of riparian vegetation and water abstraction for irrigation. Multiple stressors may have cumulative and interactive effects on stream ecosystems, such as sedimentation and flow reductions (Jackson et al., 2021; Lange et al., 2014; Matthaei et al., 2010; Townsend et al., 2008; Villeneuve et al., 2018; Wagenhoff et al., 2011). Understanding the key drivers of ecological degradation in the Tauranga Harbor catchment will enable the design and implementation of effective management strategies that mitigate and help restore these water bodies (Matthaei et al., 2010).

In this context, the main aim of my study was to assess if sedimentation is the dominant driver of stream ecological degradation in the Te Awanui catchment. I hypothesized that deposited fine sediment would explain the most variation between stream sites in different land uses but that any remaining differences not explained by sediment would indicate the influence of other stressors such as contaminant pollution (pesticides), nutrient enrichment, flow alteration, and loss of riparian vegetation (Jackson et al., 2021; Lange et al., 2014; Matthaei et al., 2010; Townsend et al., 2008; Villeneuve et al., 2018; Wagenhoff et al., 2011). I also hypothesized that the influence of these alternative stressors would be less apparent using macroinvertebrate indicators designed specifically to measure sediment impacts since they should be specific to this stressor (Davis et al., 2022).

In addition, I hypothesized that non-random species losses would lead to negative impacts of deposited fine sediment (DFS) and land-use on functional diversity (Jonsson et al., 2017). Lastly, I hypothesized that specific macroinvertebrate trait modalities would respond to sediment stress and land use, respectively (Dolédec et al., 2011; Phillips & Reid, 2012b; Phillips & Smith, 2018). Following previous studies (Ladrera et al. 2019), I predicted that life history traits (number of reproductive cycles per year <1; 'semivoltine'), morphology (gill respiration of aquatic stages; 'gill'), locomotion (crawling of aquatic stages; 'crawler'), and resource acquisition (feeding habits such as grazing; 'grazer', 'filter-feeding') would all be negatively impacted by Deposited Fine Sediment (DFS). I also predicted that the following modalities would have negative responses to sediment: 'submerged', 'univoltine', 'size1', 'surface', and 'eggprotected' (Clapcott et al., 2017; Dolédec et al., 2011; Philips & Smith, 2012; Phillips & Reid, 2012b). Regarding land-use change (native forest to pasture, and native forest to horticulture), I predicted that the following modalities would respond: 'pluriv', 'single', 'submerged', 'scraper', 'algalp', 'eggprotected', 'plastron', 'lda2', 'lda3', 'surface' and 'lowflex' (Clapcott et al., 2017; Dolédec et al., 2011; Philips & Smith, 2012; Phillips & Reid, 2012b).

2 Chapter 2: Methods

2.1 Study area

2.1.1 Tauranga Harbour catchment

The Tauranga Harbour catchment is located within the Western Bay of Plenty region of New Zealand. It encompasses an area of 1,300 square kilometres, with its boundaries being the Kaimai Ranges in the west, the suburbs of Welcome Bay to the south, Matakana Island to the east, and Waihi Beach to the north. The Tauranga Harbor encompasses 218 square kilometres (Bay of Plenty Regional Council, 2016, 2023b). The Tauranga Harbor is one of New Zealand's largest estuaries and has been identified as an outstanding natural feature (Bay of Plenty Regional Council, 2023b). Within the Te Awanui/Tauranga Harbor catchment, there are 16 main sub-catchments: Aongātete, Kaitemako, Kopurererua, Mangatawa, Oturu, Te Mania, Te Puna/Waipapa, Te Rereatukāhia, Tuapiro, Uretara, Waiau, Waimapu, Wainui, Wairoa, Waitekohe, and Waitao (Bay of Plenty Regional Council, 2016). Twenty-four sites on ten sub-catchments of the Tauranga Harbour basin were sampled, spread among three different local land uses (native forest, pastoral agriculture, and horticulture) (Figure 1a; Table 1)

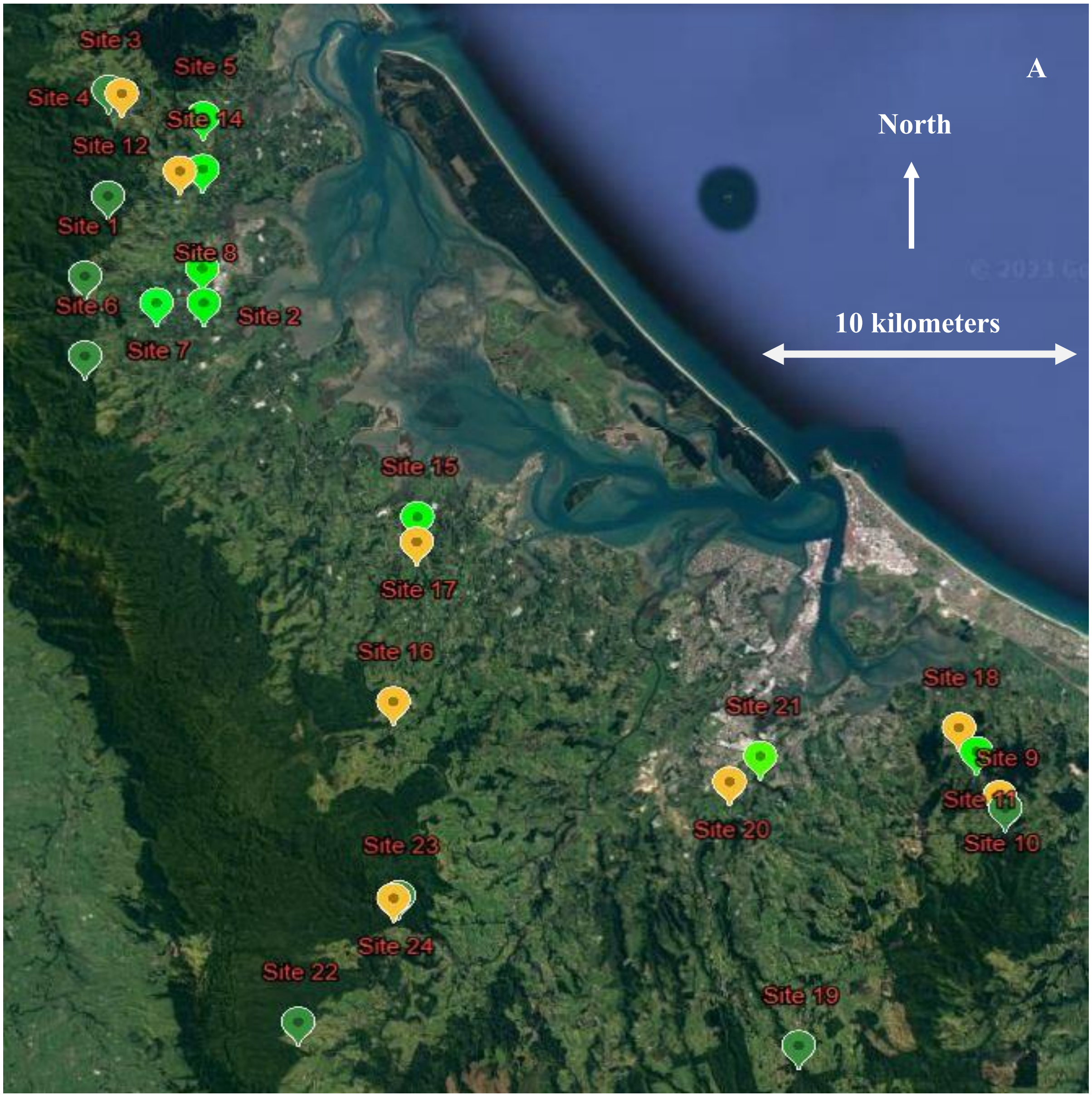

Figure 1 *a) The location of the 24 study sites in the Te Awanui/Tauranga Harbour catchment area. Dark green labels indicate native forest land use. Yellow labels indicate pastoral land use and light/lime green labels indicate horticultural land use.*

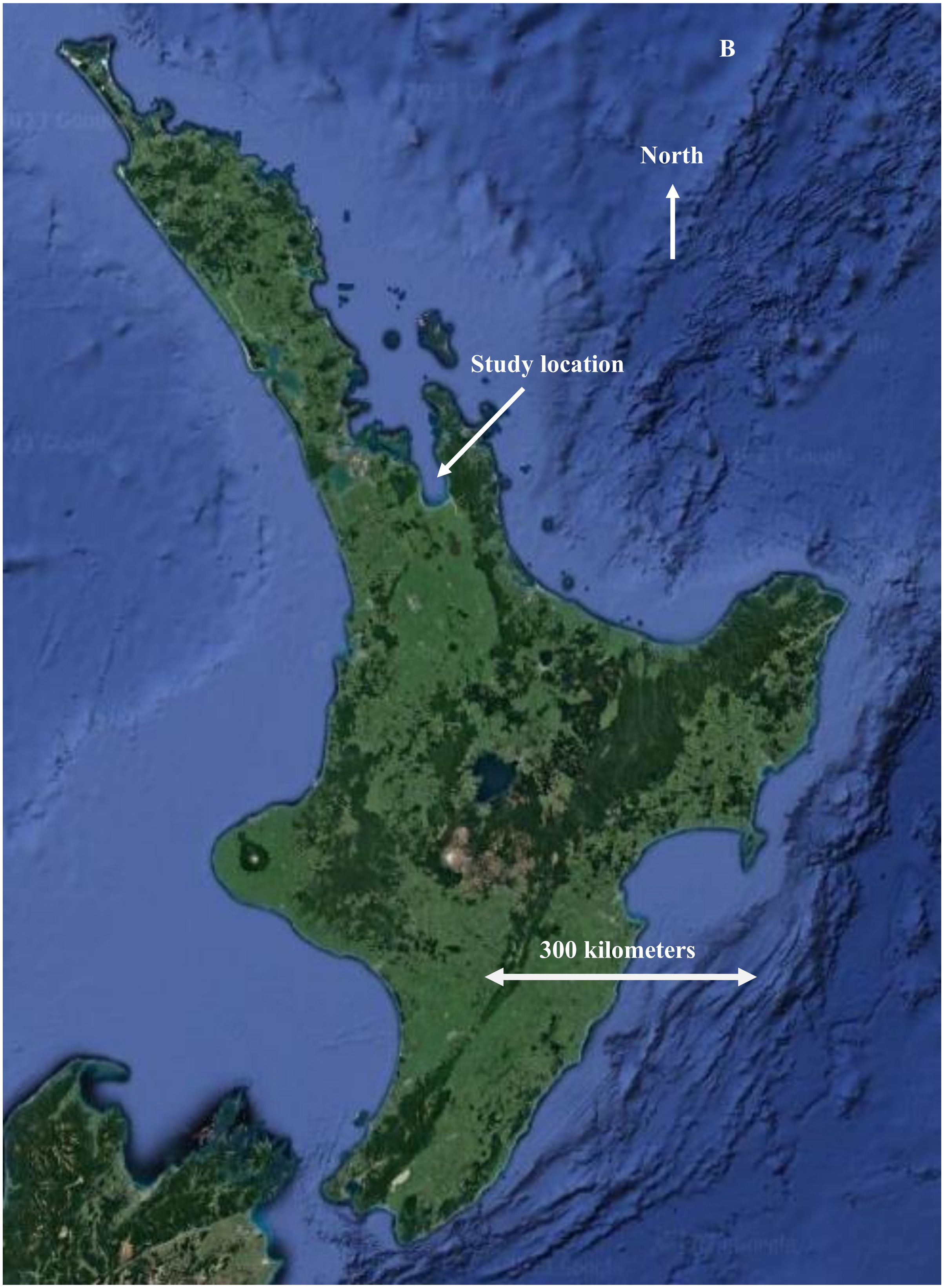

***Figure 1** b) The study location in relation to New Zealand (North Island).*

Site	Stream	Catchment	Type	Latitude	Longitude	Order	Elevation	Climate	Source	Geology	Landcover
1	Uretara	Uretara	N	-37.57	175.86	4	133	WX	H	VA	IF
2	Uretara	Uretara	H	-37.57	175.91	4	19	WW	L	VA	P
3	Waitengaue	Tuapiro	N	-37.5	175.87	3	80	WX	L	VA	IF
4	Tamaki	Tuapiro	P	-37.5	175.88	3	65	WX	L	VA	P
5	Tuapiro	Tuapiro	H	-37.51	175.91	4	16	WX	L	VA	IF
6	Tuahu	Te Rereatukāhia	N	-37.6	175.86	2	154	WW	L	VA	IF
7	Ngututuru	Te Rereatukāhia	H	-37.58	175.89	2	60	WX	L	VA	P
8	Te Rereatukāhia	Te Rereatukāhia	H	-37.58	175.91	4	16	WX	L	VA	P
9	Owairoa	Waitao	N	-37.77	176.25	2	156	WW	L	VA	IF
10	Otawera	Waitao	P	-37.77	176.25	2	139	WW	L	VA	P
11	Kaiate	Waitao	H	-37.75	176.24	3	13	WW	L	VA	P
12	Tahawai	Tahawai	N	-37.54	175.87	1	177	WX	L	VA	IF
13	Willoughby	Tahawai	P	-37.53	175.9	3	15	WX	L	VA	P
14	Tahawai	Tahawai	H	-37.53	175.91	2	15	WX	L	VA	P
15	Waipapa	Waipapa	H	-37.66	176	3	19	WW	L	VA	P
16	Waione	Te Puna	P	-37.73	175.99	2	256	WX	L	VA	P
17	Waipapa	Waipapa	P	-37.67	176	2	15	WW	L	VA	P
18	Waitao	Waitao	P	-37.74	176.23	3	16	WW	L	VA	P
19	Waiorohi	Waiorohi	N	-37.86	176.16	3	299	WX	L	VA	IF
20	Waiorohi	Waiorohi	P	-37.76	176.13	3	15	WW	L	VA	P
21	Pukekonui	Waiorohi	H	-37.75	176.15	2	13	WW	L	VA	P
22	Piako	Wairoa	N	-37.85	175.95	2	413	CX	H	VA	IF
23	Ngamuwahine	Wairoa	N	-37.8	175.99	1	178	CX	H	VA	IF
24	Mangatotara	Wairoa	P	-37.8	175.99	4	177	CX	H	VA	IF

Table 1. *Study sites in the Te Awanui/Tauranga Harbour catchment area. Site type indicates local land use: N, native forest; P, pasture; H, Horticulture. The elevation is measured as metres above sea level. Climate, Source (of flow), Geology and Landcover refer to River Environment Classification categories (Snelder & Biggs, 2002).*

[1] Climate: WX, Warm-Extremely Wet; WW, Warm-Wet; CX, Cold-Extremely Wet

[2] Source (of Flow): H, Hill; L, Lowland

[3] Geology: V, Volcanic

[4] Land Cover: IF, Indigenous Forest; P, Pastoral

2.1.2 Significance to Tangata Whenua

The Te Awanui catchment surrounds Tauranga Harbour (also known as Tauranga Moana). This region has strong cultural and spiritual significance for Māori, whose ancestors arrived between 1200 and 1300 AD (McKinnon, 2016). There are three main iwi (tribes) within the Te Awanui catchment surrounding Tauranga Moana: Ngāi Te Rangi, Ngāti Pūkenga, and Ngāti Ranginui (Bay of Plenty Regional Council, 2023b).

The iwi of Ngāi Te Rangi settled in the Te Awanui catchment after journeying from the East Coast, where they arrived in Aotearoa via the Mataatua waka (Bay of Plenty Regional Council, 2023b;

Ngāi Te Rangi Iwi, 2023). Ngāi Te Rangi has 10 hapū (subtribe) and 14 maraes (meeting house) across their rohe (customary territory), which runs from Ngā Kurī ā Whārei in the north to Mount Te Aroha then along the Kaimai ranges to Puwhenua, Otānewainuku, and Otawa, connecting with Te Tumu near Maketū. They are the largest of the three main iwi in the Te Awanui/Tauranga Moana catchment (Bay of Plenty Regional Council, 2023b; Ngāi Te Rangi Iwi, 2023).

The iwi of Ngāti Pūkenga also arrived in the Te Awanui catchment via the Mataatua Waka (Bay of Plenty Regional Council, 2023b; Ngāti Pūkenga, 2023). Ngāti Pūkenga have eight hapū across their rohe, with four main settlements situated at Manaia in the Coromandel, Ngapeke in Tauranga, Maketū, and Pakikaikutu in the Far North (Bay of Plenty Regional Council, 2023b; Ngāti Pūkenga, 2023).

The iwi of Ngāti Ranginui arrived in the Te Awanui catchment via the Takitimu waka. Ngāti Ranginui have seven affiliated hapū and ten maraes across their rohe, which ranges between Tuapiro and Waimapu (Bay of Plenty Regional Council, 2023b; Ranginui, 2023).

The significance of the Te Awanui catchment to Ngāi Te Rangi, Ngāti Pūkenga, and Ngāti Ranginui can be better understood through the concept of whakapapa (Bay of Plenty Regional Council, 2023b; Taonui, 2015). Whakapapa is the concept of genealogy in Te Ao Māori (the Māori worldview). Whakapapa not only explains the genealogy of an iwi but also their strong spiritual connection to the environment. In the Māori creation story, the universe and everything within it stems from the separation of Ranginui (Sky Father) and Papatūānuku (Earth Mother) by their descendants who became Atua (Gods) of the natural world. For example, Tāne-Mahuta is the God of the forests, Tawhirimatea is the God of wind, Tangaroa is the God of the ocean, and Maru is the God of freshwater. The separation of Ranginui and Papatūānuku eventually led to the creation of the Māori people by Tāne-mahuta. As Māori originate from these Atua, it explains their strong connection to their environment and in this context, the connection of these three iwi (Ngāi Te Rangi, Ngāti Pūkenga and Ngāti Ranguinui) to the Te Awanui/Tauranga Moana catchment. Consequently, the region provides a source of identity and ancestral lore, a homeland, and a traditional source of kai (food) for these iwi. Food sources include kaimoana (seafood) of kahawai, mussels and cockles that can be collected from Tauranga Harbor (Bay of Plenty Regional Council, 2023b; Taonui, 2015).

2.1.3 Land uses

The overall land use types in the Bay of Plenty are provided for context, although these do not necessarily represent the land cover in the Te Awanui catchment. The dominant land-use types in the Bay of Plenty are forestry (69%) and grassland (21%) (Bay of Plenty Regional Council, 2023a; LAWA, 2023a). The two areas that contribute most towards forestry lands are the native (indigenous) forests of Te Urewera National Park (Eastern Bay of Plenty) and the pine plantation forests of the Kāingaroa forest. Grasslands are comprised of primarily agricultural farmlands spread across the Bay of Plenty Region (Bay of Plenty Regional Council, 2023a; LAWA, 2023a).

Forestry comprises 69% (851,705 ha) of the land cover in the Bay of Plenty, with moderate-level land classes that comprise this land class are indigenous forest (46% or 569,791 ha) and exotic forest (23% or 281,915 ha) (Bay of Plenty Regional Council, 2023a; LAWA, 2023a). Grassland and other herbaceous vegetation comprise a total of 21% (264,819 ha) of the land cover. This comprises exotic grassland (~21% or 259,899 ha), other herbaceous vegetation (<1% or 4,302 ha) and tussock grassland (<1%, 617 ha). Cropland comprises 2% of the total land cover with 30,282 hectares. However, in the Western Bay of Plenty, this figure is likely to be higher (e.g., ~5.6%), given the prominence of kiwifruit orcharding in this part of the region (Bay of Plenty Regional Council, 2023a; LAWA, 2023a).

Scrub/shrubland comprises a further 3% (37,311 ha) of the land cover, with indigenous scrub/shrubland ~3% (33, 931 ha) and exotic scrub/shrubland <1% (3, 379 ha) (Bay of Plenty Regional Council, 2023a; LAWA, 2023a). Urban/bare/lightly vegetated surfaces comprise <1% (20,822 ha) of the total land cover. Urban areas contribute 1% (16 631 ha), naturally bare/lightly vegetated surfaces contribute <1% (3, 003 ha), and artificial bare surfaces contribute <1% (1, 188 ha). Water bodies comprise a total of 2% (29,312 ha) of total land cover in the Bay of Plenty (Bay of Plenty Regional Council, 2023a; LAWA, 2023a).

2.1.4 Geology

The geology of the Te Awanui/Tauranga Harbor is dominated by volcanic rocks and volcanogenic sediments (Briggs et al., 1996). These rocks originate from volcanic activity during the late Pliocene (23 to 5.3 mya) to the Pleistocene epoch (2.6 to 1.8 mya). The oldest geological formations are comprised of andesitic lavas and volcanic breccias from the late Pliocene. Younger geological formations are comprised of rhyolite and dacite. The sequence of geological formations in the Western Bay of Plenty from youngest to oldest are Minden Rhyolite, Matakana Basalt, Kopukairua Dacite, Waiteariki Ignimbrite, Papamoa Ignimbrite, Ongatiti Ignimbrite, Te Puna Ignimbrite, Te Ranga Ignimbrite, Waimakariri Ignimbrite and Mamaku Ignimbrite. Out of these formations, the Waiteariki Ignimbrite holds the most significance as it forms the basement layer of the Te Awanui catchment and has been tilted with the tectonic uplift of the Kaimai Ranges during the Pleistocene. The volcanism associated with the Waiteariki Ignimbrite formation also created the peninsulas of Tauranga Harbor. It is also responsible for the northeast direction in which the major rivers flow through a series of striking non-stratified normal faults (downward movement of hanging wall block relative to the footwall block) (Briggs et al., 1996).

Volcanogenic sediments were intercalated (inserted) among these volcanic formations following the deposition of the Waiteariki Ignimbrite (Briggs et al., 1996). This intercalation occurred in response to fluvial, lacustrine, and estuarine erosion processes, transportation and redeposition of volcanic debris. These volcanogenic sediments are terrestrial and estuarine in nature and originate from the Matua geological sub-group. The lithology of the Matua sub-group is comprised of rhyolitic sands, silts and gravels, ignimbrite, tephra, fluvial pumice, lacustrine silts, estuarine sands and lignites. Much of the Tauranga region is covered with a deep layer of Late Pleistocene and Holocene tephras following volcanism in the Central North Island (Briggs et al., 1996).

2.1.5 Climate

Like much of New Zealand, the Bay of Plenty has an oceanic climate (Cfb) in the Köppen-Geiger climate classification (Climate Data, 2023). Using the climate category in the New Zealand River Environment Classification indicates that most study sites were Warm-Wet or Warm-Extremely Wet (Table 1) (Ministry for the Environment, 2010; Snelder & Biggs, 2002). These climatic conditions are influenced by the region's topography (Chappell, 2014). The Bay of Plenty is sheltered on three sides by the Kaimai and Coromandel ranges to the west/north-west, the Mamaku Plateau to the south and the Raukumara Ranges of the East Coast peninsula. This sheltering causes the Bay of Plenty to have low average annual wind speeds, with the average annual wind speed in the Te Awanui catchment being 14 kilometres per hour. The sheltering from strong winds to the west, south and east influences cloud dispersal patterns, makes the Bay of Plenty one of the sunniest regions in New Zealand, with an average of 2000 hours of sunlight annually. The northerly exposure also influences air temperatures. The annual air temperature of the Bay of Plenty ranges from below 0°C to over 30°C, with its average annual air temperature being 9.5°C. The average annual air temperature of the Te Awanui catchment is 14°C (Chappell, 2014).

However, the annual rainfall of the Bay of Plenty is also very high due to its exposure to moisture-laden north-easterly weather systems (Chappell, 2014). This causes the Te Awanui catchment to have average annual rainfalls of 1,362 mm near the coast (mean value from Waihi Beach, Kaitikati and Tauranga) to around 2,000 mm in the Kaimai and Mamaku Ranges (Chappell, 2014).

2.1.6 Topography

The Tauranga Harbor catchment is constrained by the Kamai-Mamaku Range, which rises steeply from the coast and reaches a maximum height of 953 metres above sea level at Mount Te Aroha. The elevation of the 25 sites sampled in my study ranged from 2 to 413 m a.s.l. (Table 1). The median elevation of all sites was 60 m a.s.l, but this varied with site type owing to the development of the coastal zone. The median elevation of native forest sites was 156 m a.s.l., which dropped to 41 m a.s.l. for pastoral sites and 16 m a.s.l. for horticultural sites, respectively.

2.2 Sampling design

I selected 24 stream sites across ten catchments (listed above) in the Tauranga Harbor catchment. Each of these sites was selected on their dominant local land use (over the segment scale of 1km) determined *a priori* by visually examining surrounding land uses using aerial photography on Google Earth. Three land-use types were used: native forest (mixed broadleaf podocarp forest), pastoral agriculture (sheep and beef, deer, and dairying), and horticulture (mainly kiwifruit orcharding, but also featuring avocado groves). The study design was balanced, with eight stream sites surrounded by predominately native forest, eight by pasture, and eight by horticulture.

The exact location of each site (latitude/longitude) can be viewed in Table 1. I also used the MfE portal (https://data.mfe.govt.nz/) to extract the River Environment Classification (REC) data, which includes land cover characterizations over the entire upstream catchment (Ministry for the Environment, 2010).

The stream's width was multiplied by 20 at each site to determine the reach length (Harding et al., 2009). This sampling unit was chosen to ensure that the full variation of different habitats, such as riffles, runs, and pools, were included when sampling (Harding et al., 2009; Stark et al., 2001).

2.3 Survey methods

2.3.1 Water quality

Dissolved oxygen (mg/L, % saturation), water temperature (°C), ambient conductivity (µS/cm), and specific conductivity (µS/cm temperature adjusted to 25 °C) were measured at each site using a YSI Pro2030 Dissolved Oxygen and Conductivity Meter (YSI Inc., Yellow Springs, OH, USA). The YSI Pro2030 was calibrated at each site prior to sampling. pH was measured using an EC-PCTestr35 handheld probe (Eutech Instruments Pte Ltd, Paisley, UK). The sampling time was recorded, but efforts were made to standardise sampling between the daylight hours of 10 am and 4 pm. An unfiltered water sample (250mL) was collected for nitrate-nitrogen analysis; samples were placed on ice until refrigeration in the laboratory (Jowett et al., 2008).

2.3.2 Discharge, flow, and depth

The wetted width of the stream at each site was measured on a single transect. At five evenly spaced sampling points along the transect, the depth and velocity of the water were measured (Jowett et al., 2008). Velocity was measured using a Marsh-McBirney Flo-Mate 2000 (HACH Company, Frederick, MD, USA). Width, depth, and flow were used to estimate stream discharge (cumecs) (Harding et al., 2009).

2.3.3 Light

Photosynthetically available light (µmol m^{-2} s^{-1}) was measured at ten random locations throughout each study reach using a Licor LI-250A light meter (LI-COR Biosciences, Lincoln, NE, USA).

2.3.4 Bed stability

The bed stability of the stream at each site was assessed using the streambed component of the Pfankuch Stability Index, comprised of six attributes, with stability decreasing as the overall score increases (Collier, 1992). Each attribute is weighted differently; therefore, some have larger maximum scores than others. Visual observations were made while conducting the Wolman walk to help score each attribute of the index (Collier, 1992).

Rock angularity: The first attribute measured was rock angularity, scored on a scale of 1 to 4. The overall stability of a stream increases as rock angularity increases, as more angular rocks can resist erosion due to their shape's ability to 'cut through' the water, reducing hydraulic stress. A score of 1 is indicative of a site with rocks with predominately sharp edges and corners and rough plane surfaces. A score of 4 is indicative of a site with rocks that are well-rounded and have smooth surfaces (Collier, 1992).

Rock brightness: The second attribute measured is rock brightness, scored on a scale of 1 to 4 (Collier, 1992). The overall stability of a stream increases as its brightness decreases, indicating low erosion levels occurring due to the build-up of organic matter on the rocks. A score of 1 is indicative of a site with rocks with predominately dull or dark surfaces. A score of 4 is indicative of a site with predominately bright rocks, or more erosion is present in more than 65% of rocks (Collier, 1992).

Consolidation or particle packing: The third attribute was consolidation or particle packing, measured on a scale of 1 to 8, increasing in increments of 2 (e.g., 2, 4, 6, 8) (Collier, 1992). The stream's overall stability increases as the particles' packing increases as it decreases the likelihood of being moved by hydraulic force. A score of 2 is indicative of a site that has a large size range of substrates that are tightly packed. A score of 8 is indicative of a site that has a very loose substrate that is easily moved (Collier, 1992).

Bottom size distribution and percent stable materials: The fourth attribute that was measured is the distribution of the smallest substrate (e.g., sand/silt) and the percentage of stable substrate present, which is scored on a scale of 4 to 16, increasing on increments of 4 (Collier, 1992). The overall stability of the stream increases as the percentage of small-sized substrate decreases and the percentage of stable substrate increases. A score of 4 is indicative of a site where between 80 to 100% of the substrate is stable, and there is no evident change in the size distribution of the substrate. A score of 16 is indicative of a site where less than 20% of the substrate is stable, and there is a significant change in the size distribution of the substrate (Collier, 1992).

Scouring and deposition: The fifth attribute measured is the degree of scouring and deposition present in a site's streambed, scored on a scale of 6 to 24, increasing in increments of 8 (Collier, 1992). The overall stability of the stream increases as the percentage of scouring and deposition decreases. A score of 6 is indicative of a site where less than 5% of its streambed has been affected by scouring and deposition. A score of 24 is indicative of a site where more than 50% of its streambed has been affected by erosion and deposition (Collier, 1992).

Clinging aquatic vegetation: The sixth attribute measured is the abundance of clinging aquatic vegetation present at a site, scored on a scale of 1 to 4, increasing in increments of 4 (Collier, 1992). Overall stream stability increases as the abundance of macrophytes increases and the percentage of algae decreases. A score of 1 is indicative of a site that has a high abundance of dark green, moss-like macrophytes in perennial growth, even in swift water. A score of 4 is indicative of a site where the perennial growth of macrophytes is absent or scarce and algal growth is abundant (Collier, 1992).

2.3.5 Benthic substrate and fine sediment

Benthic substrate composition and deposited fine sediment (DFS) were determined using two main Sediment Assessment Methods (SAM): the SAM2 (visible streambed cover) and SAM3 (the Wolman walk) protocols (Clapcott et al., 2011).

The SAM2 method relies on visual observations of DFS on the streambed. It involves five randomly selected transects across the stream channel within the study reach (Clapcott, Harding, et al., 2011). Four randomly selected points were chosen along each transect, and the streambed was observed using a bathyscope (underwater viewer). Observations began at the downstream transect to prevent accidental releases of DFS influencing results. The visible percentage of DFS cover observed through each of the four quadrants in the bathyscope was then estimated and recorded. This process was then repeated for the next four transects, resulting in a total of 20 independent observations taken (Clapcott, Harding, et al., 2011).

For the SAM3 protocol, 100 individual substrate samples were selected randomly (i.e., using the Wolman walk) at each site throughout the study reach (Clapcott, Harding, et al., 2011). Each substrate sample was recorded and categorized based on its size. In order from largest to smallest, these were the substrate size categories recorded: concrete/artificial, bedrock, boulder (>264 mm), cobble (64 – 264 mm), gravel (8 – 64 mm), fine gravel (2 – 8 mm), silt/sand/mud (<2 mm) (Clapcott, Harding, et al., 2011).

2.3.6 In-stream habitat conditions

Aspects of instream habitat, including substrate composition, were also observed using Rapid Habitat Assessment (P2c) approaches (Harding et al. 2009). The percentage of riffle, run, and pool habitat present in a stream was assessed visually, with various parameters of stream bed substrate composition, organic matter and fish habitat measured for each of the three habitat types (Harding et al., 2009)

In riffle, run, and pool sections, the same substrate size categories used in the SAM3 protocol were estimated as a proportion of the streambed using visual assessment following a stream walk along the study reach (Clapcott, Harding, et al., 2011). Substrate compactness, which measures the degree of 'packing' or closeness of substrate in proximity to each other, was recorded as loose, moderate, or high (Clapcott, Harding, et al. (2011). Embeddedness was measured as the percentage of larger substrate types (e.g., cobbles) smothered in fine sediment (silt and/or sand). Zones of deposition and scouring were estimated as a proportion of the streambed for each section. Different types of organic matter present were estimated as a proportion of the streambed. These were macrophytes, moss, algae, woody debris and leaf packs (Clapcott, Harding, et al., 2011).

Lastly, the condition of a stream's fish habitat was assessed too (Clapcott, Harding, et al., 2011). The first metric was the percentage of area within a stream obstructing fish passage. This metric is important as nineteen of New Zealand's 52 fish species are diadromous, meaning they require adequate fish passage between freshwater and marine environments to complete their life cycle. The second metric measured the percentage of bank cover within a stream available to fish as a habitat/shelter (Clapcott, Harding, et al., 2011).

2.3.7 Riparian habitat

2.3.7.1 Riparian Condition Index

At each site, the riparian conditions, substrate composition and hydrology were all assessed using semi-quantitative protocols (P2) outlined in Harding et al. (2009). The assessment sheet that was used can be viewed in Appendix I. The quality of riparian conditions was assessed using a semi-quantitative method that scored 11 different attributes of riparian conditions on a scale of 1 (worst) to 5 (best). The riparian zone of a stream's left and right banks were scored separately (Harding et al., 2009).

Shading: The first attribute measured was water shading, which measures the degree of shading at the water's surface provided by banks, vegetation, and hill slopes (Harding et al., 2009). A score of 1 reflects less than 10% of the water's surface being shaded, whereas a score of 5 reflects more than 80% of the water's surface being shaded (Harding et al., 2009).

Buffer width: The second attribute measured is buffer width, which considers the width between the edge of the stream bank and the buffer (Harding et al., 2009). A buffer is any area that aims to prevent or reduce the effects of land use from impacting the stream. A typical example of a buffer is the planting of native riparian vegetation adjacent to the stream bank. A score of 1 is indicative of a buffer that is less than 1 meter from the stream bank, whereas a score of 5 is indicative of a buffer that is more than 30 meters from the stream bank (Harding et al., 2009).

Buffer intactness: The third attribute that was measured is the intactness of the buffer, which estimates the gaps present in the buffer as a percentage (Harding et al., 2009). This attribute helps estimate the buffer's effectiveness, as a completely intact buffer will be more effective at preventing pollutants from entering the stream and providing habitat. A score of 1 is indicative of a buffer that is not present at all, whereas a score of 5 is indicative of a buffer that is 100% intact (Harding et al., 2009).

Vegetation composition: The fourth attribute measured is the buffer's vegetation composition and the dominant land use within 30 meters of the stream bank (Harding et al., 2009). The composition of the buffer and the adjacent land use were scored separately on both sides of the stream (left & right). This metric also helps to estimate the effectiveness of the buffer as the vegetation composition of the buffer influences how effective it is in providing habitat and preventing pollutants from entering the stream. A score of 1 is indicative of short, grazed pasture or impervious surfaces such as concrete. A score of 5 is indicative of a mature native forest (Harding et al., 2009).

Bank stability: The fifth attribute measured is the stability of stream banks. This metric helps assess the degree of erosion present in each streambank, which is important as it indicates a site's contribution towards instream sedimentation (Harding et al., 2009). A score of 1 is indicative of a site where more than 40% of it has been eroded, has incohesive sediment, and has few plant roots. A score of 5 is indicative of a site where less than 1% of it has been eroded and has geology that provides high stability, such as bedrock, along with a high proportion of soil-stabilizing vegetation cover (Harding et al., 2009).

Livestock access: The sixth attribute measured is livestock's accessibility to streams. This metric helps assess the potential impact of surrounding land use on the stream ecosystem via livestock defecating into the stream or trampling organisms within it (Harding et al., 2009). A score of 1 is indicative of a site that has active livestock use with no fencing present. A score of 5 is indicative of a site that has either permanent fencing or no livestock (Harding et al., 2009).

Riparian soil denitrification: The seventh attribute that was measured is the denitrification potential of the soil (Harding et al., 2009). This metric helps assess the soil's capacity in the riparian zone to denitrify anthropogenic nitrogen sources such as fertilizer or livestock excrement and process it via the nitrogen cycle before it can enter the stream water and cause eutrophication. A score of 1 is indicative of a site that has soil that is dry and firm or is moist-wet but has a drain tile bypass riparian soils. A score of 5 is indicative of a site where more than 30% of the soil in the riparian soil is water-logged, or the surface is moist and extrudes fluid when stepped upon and has no drains (Harding et al., 2009).

Land slope: The eighth attribute measured was the degree of slope present across 30 meters from the stream bank (Harding et al., 2009). This metric helps to assess the rate at which potential pollutants can run off into the stream, with runoff rates increasing as the slope increases. A score of 1 is indicative of a site that has a slope that is more than 35 degrees. A score of 5 is indicative of a site that has a slope that is less than 5 degrees (Harding et al., 2009).

Groundcover composition: The ninth attribute that was measured was the composition of the groundcover present in the buffer and the land adjacent up to 30 meters from the streambank (Harding et al., 2009). The groundcover of the buffer and adjacent land-use are scored separately for both sides of the stream (left & and right). This metric helps to assess the capacity of the ground cover to process pollutants and prevent them from entering the stream. A score of 1 is indicative of a site that has bare groundcover. A score of 5 is indicative of a site that has high-density long grass (Harding et al., 2009).

Soil drainage: The tenth attribute measured is the drainage potential/permeability of the soil in the riparian zone (Harding et al., 2009). This metric assesses the soil's ability to absorb pollutants such as effluent runoff and prevent them from accessing the stream's water. A score of 1 is indicative of impervious surfaces such as concrete or is extensively compacted to the extent that its permeability is extremely low. A score of 5 is indicative of a site that has a soil composition with very high permeability, such as pumice or sand and is not compacted (Harding et al., 2009).

Rills/channels: The 11th attribute that was measured is the number and size of rills and or channels present in the riparian zone (Harding et al., 2009). This metric assesses the potential for pollutants to enter the stream via runoff. A score of 1 is indicative of a site that has more than 9 rills per 100 meters or more than 2 large channels carrying runoff. A score of 5 is indicative of a site that has no rills or channels (Harding et al., 2009).

2.3.7.2 *Riparian vegetation*

At each site, a qualitative assessment of the floral biodiversity was conducted (Harding et al., 2009). The species present were recorded from both banks along a reach length that was 20 times the wetted width of the stream (Harding et al., 2009).

2.3.8 Macroinvertebrates

Macroinvertebrate samples were collected following the National Environmental Monitoring Standards (NEMS) protocol recommended for kicknet sampling in wadeable streams (Milne et al., 2022). I used a kicknet with a mesh size of 0.5 mm for macroinvertebrate sampling. The reach sampled at each stream site contained habitat representative of the entire stream segment, and sampling targeted the most commonly available wadeable mesohabitats (e.g., riffles, runs) within the reach. The length of the reach sampled was approximately 20 times the average channel width with a minimum length of 20 m. A single composite sample was comprised of 4–8 unit efforts (subsamples), with effort split based on the proportional contribution of each mesohabitat in the sampling reach. Each unit effort was approximately 0.1–0.15 m^2 in area, and the total sampling area sampled was 0.6–0.9 m^2. Samples were preserved in 70% ethanol and stored for further processing (Milne et al., 2022).

2.4 Laboratory analyses

2.4.1 Nitrate-nitrogen

Nitrate-nitrogen concentrations (mg/L) were measured using a ‘colourimetric’ method (microtest® Nitrate-N NED^{HS}, Product no. 654100, Aquaspex Water Testing Products, Edwardstown, SA, Australia); (NIWA, 2020). This test uses an acidic reagent to reduce nitrate to nitrite by metallic zinc. Subsequent diazotation/azocoupling forms an intensely coloured pink azo dye, the intensity of which is related to the nitrite concentration. In this approach, the colour intensity is read by the eye using a colour comparator. It is designed for rapid assessment and is less accurate than alternative methods using a photometer (NIWA 2019). The standard measuring range is 0 – 0.8 mg/L nitrate-nitrogen. The colour comparator has 5 colour fields: 0.05, 0.1, 0.2, 0.4, and 0.8 mg/L. The sample was diluted for higher nitrate concentrations (e.g., a 1:4 sample dilution can extend the measuring range to 18 mg/L). Water samples were refrigerated and processed at room temperature within 48 hours of collection (NIWA, 2020).

2.4.2 Macroinvertebrates

Macroinvertebrate processing followed the NEMS 200+ fixed count and scanned for missed taxa protocol (Milne et al., 2022). The macroinvertebrate samples were processed in the laboratory for taxonomic identification and enumeration. Firstly, the sample was rinsed through a clean 0.5 mm sieve to remove the ethanol preservative, fine sediment, and any remaining unwanted organic material (e.g., whole leaves and twigs). The sample was then divided equally into four large white plastic sorting trays. Individual macroinvertebrates were removed from a randomly selected tray using forceps and placed in a petri dish. If macroinvertebrates remained in a tray after 200 individuals were removed, these individuals were also removed for identification. If <200 individuals were recovered from a tray, the same steps were repeated on another randomly selected tray. The macroinvertebrate individuals removed were separated into different Petri dishes based on their taxonomic order (e.g., Trichoptera and Ephemeroptera individuals were placed in different petri dishes). Once macroinvertebrate extraction was completed, all individuals were viewed under a stereo microscope and identified down to the species level where possible using standard taxonomic keys (e.g., Winterbourn et al. 2006, Winterbourn 2023). The abundances of taxa identified were then recorded. Finally, the remaining trays were scanned for rare or missed taxa that had not yet been recorded. Rare or missed taxa were removed for identification and enumeration. The remaining contents of each macroinvertebrate sample (i.e., the residual) were then stored in its original container and preserved with 70% ethanol (Milne et al., 2022).

2.5 Data analysis

2.5.1 Taxonomic-based indicators

Multiple taxonomic-based indicators were used to describe changes in stream macroinvertebrate communities (Burdon et al., 2013; Clapcott et al., 2017). These included the Macroinvertebrate Community Index (MCI) and its quantitative equivalent (QMCI), the species richness and relative abundance of Ephemeroptera, Plecoptera and Tricoptera (EPT), the Average Score Per Metric (ASPM) using the MCI, EPT richness and relative abundance indicators described above, and the percentage of sediment 'decreasers' in the abundance and richness of taxa across all sampled sites (Burdon et al., 2013; Clapcott et al., 2017).

The Macroinvertebrate Community Index (MCI) helps assess the impact of anthropogenic pollution on a stream ecosystem by accounting for the presence or absence of macroinvertebrate species (Clapcott et al., 2017). It assigns a pollution tolerance value to macroinvertebrates ranging from 1 (less sensitive to pollution) to 10 (most sensitive to pollution). The MCI score for a site was calculated by calculating the average tolerance values of all taxa present multiplied by 20. The MCI score of a site/stream can range between 0 and 200, with ecological health increasing as the MCI score increases. I used the hard-bottomed version of the MCI for my calculations (Clapcott et al., 2017).

The Quantitative Macroinvertebrate Community Index (QMCI) functions similarly to the MCI but also accounts for the abundance of macroinvertebrates (Burdon et al., 2013; Clapcott et al., 2017). It is calculated by multiplying each species' abundance by its tolerance value, summing these, and dividing the combined value by the total number of individuals present in the sample. The QMCI is scored on a scale of 1 to 10, with the ecological health of a site/stream increasing as its QMCI increases (Burdon et al., 2013; Clapcott et al., 2017).

The relative abundance of EPT taxa was calculated by dividing the abundance of EPT taxa by the total abundance of a site/stream (Burdon et al., 2013; Clapcott et al., 2017). This was repeated for all sites and displayed as a percentage. The species richness of EPT taxa was calculated by simply counting the number of species belonging to the stream insect orders Ephemeroptera, Plecoptera and Tricoptera present in the sample of a site, which was repeated for all sites (Burdon et al., 2013; Clapcott et al., 2017).

The Average Score Per Metric (ASPM) is a multi-metric index derived from the MCI score, %EPT abundance, and EPT species richness of a site (Collier, 2008). Both EPT metrics exclude the microcaddisfly family Hydroptilidae. Firstly, values for the three metrics were standardised by the maximum values stated in the National Policy Statement for Freshwater Management 2020 (200 for the MCI, 29 for EPT richness, and 100 for %EPT abundance). This results in values for each site that range between 0 and 1. The mean value of these standardized metrics was then used to calculate the ASPM at each site (Collier, 2008).

The percentage (%) abundance and richness of sediment ‘decreasers’ were also used to assess changes in the macroinvertebrate communities of stream ecosystems sampled (Clapcott et al., 2017). The sediment ‘decreasers’ metrics assume that taxa decreasing in response to deposited fine sediment are less tolerant of this pollutant. Conversely, taxa that increase (‘increasers’) in response to this stressor are more tolerant of sediment pollution. These two metrics were calculated using the sediment ‘decreasers’ taxa identified by Clapcott et al. (2017). Their approach used partial dependence plots from random forest analyses and expert opinion to identify 25 sediment ‘decreasers’ and 12 ‘increasers’. Twenty-eight taxa were classified as ‘unclear’. (Clapcott et al. 2017) analysed 1,861 samples collected from 973 sites across New Zealand, including data from (Burdon et al., 2013).

2.5.2 Functional diversity indicators

I used functional biological traits to describe changes to stream macroinvertebrate communities. Biological traits include life cycle, physiological and behavioural characteristics, such as maximum body size, lifespan, feeding and reproductive strategies, and mobility. In contrast, ecological traits are related to habitat preferences, like pH and temperature tolerances, tolerance to organic pollution, and biogeographic distribution (Usseglio-Polatera et al., 2000a). Sixteen biological traits with 59 modalities were sourced from v.19.2.18 of the New Zealand Freshwater Macroinvertebrate Trait Database (Phillips & Smith, 2018). The 16 traits were: maximum potential size; maximum number of descendants per reproductive cycle; maximum number of reproductive cycles per year; number of reproductive cycles per individual; life duration of adults; reproductive technique; oviposition site; egg-type; dissemination potential (all stages); attachment to substrate of aquatic stages (excluding eggs); body flexibility; body form; feeding habits; dietary preferences; respiration of aquatic stages (not including eggs); and aquatic stages (Phillips & Smith, 2018). The trait database assigns trait affinity scores (0-3) using ‘fuzzy-coding’ (Chevene et al., 2006) for 495 freshwater macroinvertebrate taxa. I used median values at the taxon level to account for trait variability among species of the same genera since the majority of my macroinvertebrate identifications were made at this coarser level of taxonomic resolution.

Functional diversity (FD) metrics and community-weighted means of traits were obtained using the dbFD function in the 'FD' R package (Laliberté & Legendre, 2010; Villéger et al., 2008). Five metrics of macroinvertebrate FD were calculated: functional richness (FRic), evenness (FEve), divergence (FDiv), dispersion (FDis), and Rao's Q functional diversity index. FRic describes the volume of the functional space occupied by the community, FRic the regularity of the distribution of abundance in this volume, and FDiv the divergence in the abundance distribution in this volume. FDis is similar in concept and highly correlated to Rao's Q functional diversity index and simultaneously quantifies both trait dissimilarity and evenness within communities (Laliberté & Legendre, 2010; Villéger et al., 2008). Accordingly, the most functionally dispersed assemblage (high FDis) comprises evenly distributed, dissimilar traits. Firstly, the Gower distance matrix was calculated from the invertebrate taxa functional traits. This matrix was then used to calculate a centroid for each invertebrate sample after accounting for variation in species abundances. FDis is the sum of the distances of each species to that centroid, weighted again by taxa relative abundances (Laliberté & Legendre, 2010; Villéger et al., 2008).

2.5.3 Mixed models and linear regression

To test for differences in macroinvertebrate responses to local land-uses (native forest, pasture, horticulture), I used linear mixed-effects models. The first set of models included a fixed effect for land use and a random effect for sub-catchment since some study sites were located on the same stream. The second set of models added deposited fine sediment (SAM2) as a covariate to test the hypothesis that any differences between local land-uses uses not explained by sediment would indicate the influence of other stressors such as contaminant pollution (pesticides), nutrient enrichment, flow alteration, and loss of riparian vegetation. Mixed models were fitted with the *lme4* and *blme* packages in R. Analysis of Variance (ANOVA) was performed on mixed models using the *car* package in R. Post-hoc tests were conducted using the *lsmeans* package with Tukey's correction for multiplicity.

I also fitted linear regression models with deposited fine sediment (SAM2) as a predictor to assess the response of macroinvertebrate indicators to this stressor. In all analyses, I used Shapiro tests and quantile-quantile (Q–Q) plots to assess normality and heteroscedasticity. The proportion of deposited fine sediment (SAM2) was logit-transformed to prevent over-fitting. I used the arc-sine square root transformation for % 'Decreaser' abundances. For taxa richness responses, a generalised linear mixed-model approach assuming a Poisson distribution was used for count data.

2.5.4 Community analyses

I used unconstrained ordination techniques to assess differences in community composition across local land-use types. I conducted a Non-metric Multi-Dimensional Scaling (NMDS) analysis on Hellinger-transformed macroinvertebrate relative abundance data using the Bray-Curtis distance metric. I tested the differences in community composition between site types using Permutational M with the 'adonis' and 'pairwiseAdonis' R functions (Anderson, 2001). I also assessed the taxa that contributed the most to differences across site types using indicator species analysis (ISA). Formal ISA was conducted using the R package 'interspecies' (Anderson, 2001). This analysis was based on the same NMDS model described above.

3 Chapter 3: Macroinvertebrate responses to land use and sedimentation

3.1 Introduction

Human activities have caused significant ecological degradation in New Zealand's freshwater ecosystems through land use changes (Julian et al., 2017). Land use change significantly increases sedimentation, nutrient enrichment, and contaminant pollution, leading to hydrological alterations, removal of riparian vegetation, and loss of large woody debris (Allan, 2004). As described in Chapter One, these impacts have numerous adverse effects on freshwater ecosystems globally and within New Zealand. However, knowing which of these proximate drivers contribute the most to declines in stream ecosystem health remains uncertain, especially in the Bay of Plenty, where kiwifruit orcharding is a significant and relatively novel land use.

In New Zealand, regional councils are responsible for managing the health of the freshwater ecosystem (Ministry For the Environment et al., 2020). This responsibility is bestowed upon the councils through the Resource Management Act 1991 (RMA), which provides them with the authority to manage anthropogenic impacts on freshwater ecosystems (Ministry For the Environment et al., 2020). The RMA provides the legislative foundation for the National Policy Statement for Freshwater Management 2020 (NPS-FM) (Ministry For the Environment et al., 2020). The NPS-FM provides management standards and biocriteria for freshwater ecosystems that set the targets to be achieved by councils regarding the ecological health of freshwater ecosystems in their region (Ministry For the Environment et al., 2020).

The councils in New Zealand are required to monitor a wide range of environmental and biological attributes to ensure that the NPS-FM standards for freshwater ecosystem health are met. In this thesis, I have used benthic macroinvertebrates, one of the biomonitoring tools used by regional councils for management purposes (Clapcott et al., 2017; Collier, 2008; Milne et al., 2022). The macroinvertebrate attributes used in the NPS-FM include the Macroinvertebrate Community Index (MCI) and its quantitative version (QMCI), and a multi-metric index, the Average Score Per Metric (ASPM), which is made up of the MCI and the relative abundance (%) and richness of EPT taxa (Burdon et al., 2013; Clapcott et al., 2017; Collier, 2008). I also included sediment-specific indices (the relative richness and abundance of sediment ‘decreasers’) that have been shown to be more effective indicators of the stress imposed by deposited fine sediment in the New Zealand context (Clapcott et al., 2017; Davis et al., 2022). These taxonomic-based indicators (e.g. MCI) are well-established as bioindicators of stream ecosystem health in New Zealand.

Macroinvertebrate-based biomonitoring tools are often employed alongside other indicators that measure changes in the stream environment (Clapcott et al., 2017; Harding et al., 2009; Milne et al., 2022). This dual use of biological and environmental indicators is essential in helping to establish cause-and-effect relationships, where environmental conditions can significantly influence changes in the biological communities of an ecosystem (Harding et al., 2009; Townsend et al., 2008). Although change can occur within biological communities independent of their environment, such as through altered biotic interactions, it is crucial to use environmental indicators to identify if and how anthropogenic activities affect ecosystem health (Harding et al., 2009; Milne et al., 2022; Townsend et al., 2008). Using biological indicators alone might identify human land use's ultimate effects on ecological communities, but not necessarily the underlying proximate drivers (Charvet et al., 1998). This thesis measured the following environmental parameters: sedimentation, water quality, hydrology, and riparian habitat conditions alongside local land use (Harding et al., 2009; NIWA, 2020; Milne et al., 2022). These environmental indicators are also attributes regional councils monitor for the NPS-FM (i.e., % deposited fine sediment cover, nitrate-nitrogen concentrations).

Sedimentation was assessed using the SAM2 method (Clapcott et al. 2011). This method estimates the percentage of the area of the streambed covered by deposited fine sediment (< 2mm in grain size). Water quality parameters measured included dissolved oxygen (mg/L, %), water temperature (°C), specific conductivity (μS/cm), and pH. Nutrient concentrations of nitrate-nitrogen (mg/L) were analysed from water samples. Hydrological parameters measured included flow velocities (m/s), and riparian habitat integrity was assessed using the Riparian Condition Index (Harding et al., 2009).

This chapter aims to investigate the impacts of local land use on stream ecosystem health within the Tauranga Moana catchment using taxonomic-based macroinvertebrate indicators. The first hypothesis of my thesis states that Deposited Fine Sediment (DFS) explains the most variation between stream sites in different land uses but that any remaining differences not explained by sediment would indicate the influence of other stressors such as contaminant pollution (pesticides), nutrient enrichment, flow alteration, and loss of riparian vegetation. I also hypothesized that the influence of these alternative stressors would be less apparent using macroinvertebrate indicators designed specifically to measure sediment impacts since they should be specific to this stressor.

3.2 Results

3.2.1 Environmental indicators

There were significant differences in stream-deposited fine sediment (% DFS) between land uses (*P*<0.01; Table 2, Figure 2a). Pastoral streams had the highest levels of DFS, followed by horticultural streams, and native forest streams had the lowest levels of DFS (Figure 2a). Post-hoc testing revealed that DFS was significantly higher in pastoral stream sites than native forest streams (*P*<0.01). However, there were no statistically significant differences between horticultural and native forest stream sites (*P*>0.05) or between horticultural and pastoral sites (*P*>0.05).

Stream concentrations of nitrate-nitrogen (mg/l) differed significantly between land uses (*P*<0.001; Table 2, Figure 2b). Pastoral streams had the highest concentrations of nitrate-nitrogen, followed by horticultural streams, and native forest streams had low levels close to the limit of detection (Figure 2b). The NO_3-N concentrations were significantly higher in pastoral (*P*<0.01) and horticultural (*P*<0.01) sites than in native forest streams, but there were no statistically significant differences between horticultural and pastoral sites (*P*>0.05).

Riparian habitat integrity indicated by the Riparian Condition Index (RCI) differed significantly between land uses (*P*<0.001; Table 2, Figure 2c). The RCI scores were significantly higher in native forest streams when compared to pastoral (*P*<0.001) and horticultural (*P*<0.001) sites, but there were no differences between horticultural and pastoral sites (*P*>0.05; Table 2, Figure 2). Pastoral and horticultural land uses showed similarities in degraded riparian conditions with less woody vegetation cover and increased presence of non-native plants, including introduced grasses (Figure 2) (Zermeño-Hernández et al., 2020).

Mean stream flow velocities (m/s) did not differ statistically between land-use types (*P*>0.05; Table 2, Figure 2d), which was indicative of the relatively steep catchments draining to the Tauranga Harbour. Stream velocities were highest in horticultural streams, followed by pastoral and native streams, but all three land uses had a relatively broad distribution range for this attribute (Figure 2d).

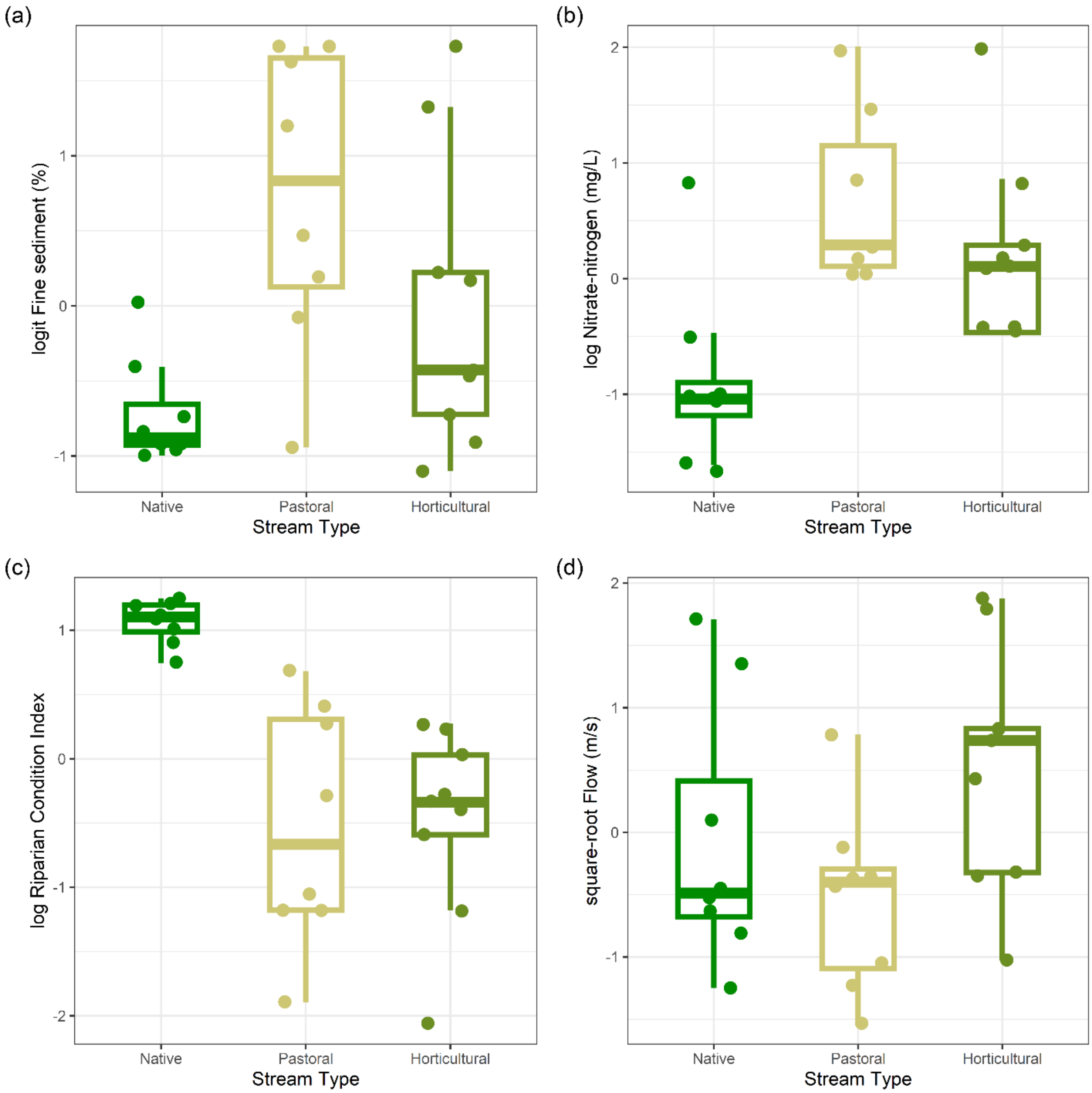

Figure 2. *Changes in key environmental indicators across streams in three local land-use types in the Te Awanui/Tauranga Harbour basin. a) logit-transformed deposited fine sediment (%) on the stream bed; b) log-transformed concentrations of nitrate nitrogen (mg/); c) riparian habitat integrity indicated by the Riparian Condition Inde; and d) square-root transformed flow velocities (m/s). All indicators are standardized (centred on the mean and scaled to unit variance).*

Table 2. *ANOVA results from mixed models testing changes in standardized environmental indicators across streams in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin.*

Response	***F*-value**	***df***	***df*.residual**	***P*-value**
Logit Fine sediment (%)	7.6	2	15.9	**0.005**
Log Nitrate-nitrogen (mg/L)	13.6	2	14.4	**<0.001**
Log Riparian Condition Index	14.1	2	17.5	**<0.001**
Square-root Flow (m/s)	3.1	2	17.2	0.069

The relative abundance of the substrate composition of streams in each land-use type (native, pastoral, horticultural, overall) was measured using the SAM3 method, also called the Wolman walk (Figure 3).

In native forest streams, cobble was the dominant substrate type (37.1%), followed by boulders (23.7%), gravel (13.1%), then bedrock (9.7%), then fine gravel (8.4%), then sand (8%) and silt which was not detected using the SAM 3 method (0%); (Figure 3a, Table 3).

In pastoral streams, silt was the dominant substrate type (34.2%), followed by cobble (22%), then sand (18.8%), then gravel (8.7%), then boulder (8.3%), then fine gravel (7.1%), then bedrock (0.9%); (Figure 3b, Table 3).

In horticultural streams, cobble was the most dominant substrate type (35.3%), followed by sand (14.7%), mud (13.3%), boulders (11.5%), gravel (10.7%), bedrock (8.4%) and fine gravel (6%); (Figure 3c, Table 3).

Overall, cobble was the dominant substrate type (32.3%), followed by boulders (15.3%), sand (14%), mud (13.3%), fine gravel (11%), gravel (7.2%) and bedrock (6.8%); (Figure 3d, Table 3).

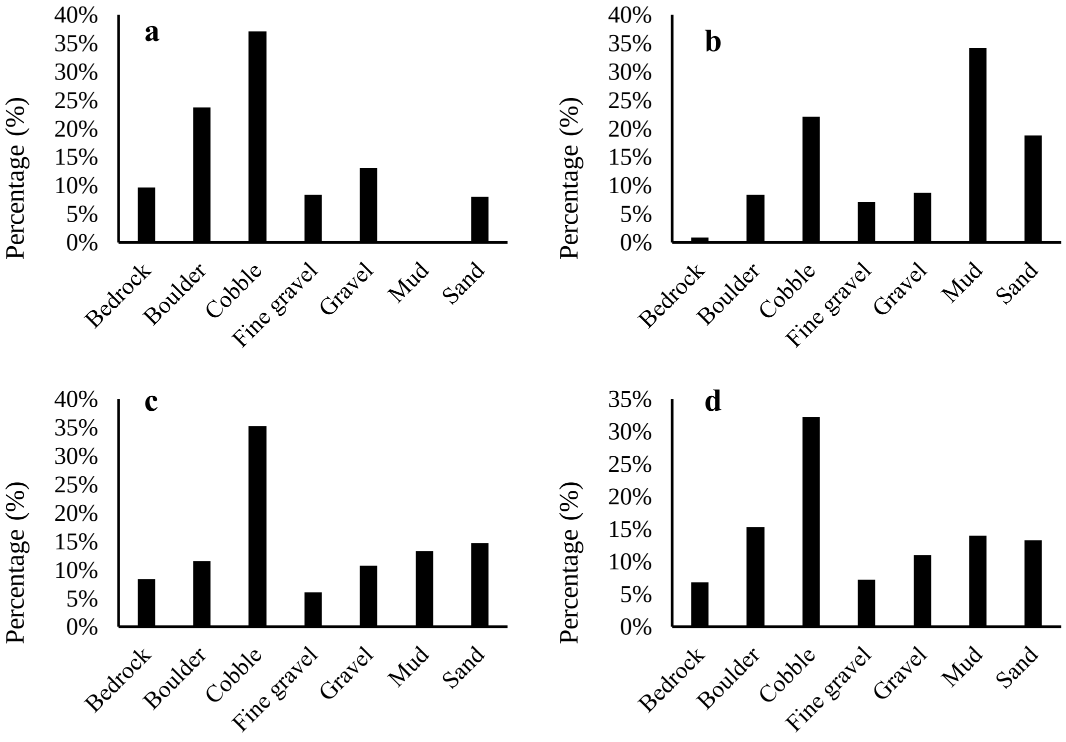

Figure 3. *The substrate composition of a) native forest streams, b) pastoral streams, c) horticultural streams and d) overall. Substrate composition was measured using the SAM3 method (Wolman walk).*

Table 3. *Mean substrate compositions of native forest, pastoral, and horticultural streams, along with all streams combined.*

Substrate Size Category	Native	Pastoral	Horticultural	Overall
Bedrock	9.7%	0.9%	8.4%	6.8%
Boulder (>256 mm)	23.7%	8.3%	11.5%	15.3%
Cobble (>64 -256mm)	37.1%	22%	35.3%	32.3%
Gravel (16 - 32 mm)	13.1%	8.7%	10.7%	7.2%
Fine gravel (>2 - 16mm)	8.4%	7.1%	6%	11%
Sand (>0.06 - 2mm)	8%	18.8%	14.7%	14%
Silt (<0.06mm)	0%	34.2%	13.3%	13.3%

Standard water quality parameters were measured across native, pastoral, and horticultural streams, with the mean value of each parameter for each land-use type being reported (Table 4). It should be noted that the reported values may not be truly representative of streams in each land use, as sampling was a singular occurrence at random times throughout the day (Harding et al., 2009).

Mean air temperature was highest in native forest streams, followed by horticultural and pastoral streams (Table 4). Ambient conductivity was highest in horticultural streams, followed closely by horticultural and pastoral streams. Dissolved oxygen (mg/L) concentrations were highest in pastoral streams, followed closely by horticultural and native forest streams. However, dissolved oxygen saturations were highest in native forest streams, followed by pastoral and horticultural streams. The pH of native and pastoral streams was the same, with horticultural streams having a slightly lower pH. Specific conductivity was highest in horticultural streams, followed by pastoral streams and then native forest streams. The water temperature was highest in native forest streams, followed by pastoral and horticultural streams (Table 4).

Table 4. *Mean water quality parameters of native forest, pastoral and horticultural streams*

Variable	Native	Pastoral	Horticultural
Air temperature	17.6	14.8	17.9
Ambient conductivity	41.5	28.9	42.1
Dissolved O2 (mg/L)	9.7	10.4	9.8
Dissolved oxygen (%)	98.2	98.1	97.5
pH	7.4	7.4	7.3
Specific conductivity	50.3	29.8	51.2
Water temperature	15.7	12.7	15.4

3.2.2 Macroinvertebrate community composition

The non-metric multidimensional scaling (NMDS) plot of macroinvertebrate community composition revealed significant changes across local land-use types (Figure 4). These changes were highly significant, as indicated by Permutational Multivariate Analysis of Variance (PERMANOVA). The changes in community composition across all local land-use types ($F_{2,21}$=3.01, $P<0.001$, R^2=22.3%) were matched by differences between native forest streams and pastoral sites ($F_{1,14}$=4.10, $P<0.01$, R^2=22.6%), and native forest and horticultural sites ($F_{1,14}$=4.24, $P<0.001$, R^2=23.2%). There was no difference between pastoral and horticultural sites ($F_{1,14}$=0.95, $P>0.05$, R^2=6.4%).

Indicator analysis showed that five taxa were significantly associated with native forest streams: the mayfly *Coloburiscus humeralis*, the caddisflies *Helicopsyche* sp. and *Beraeoptera roria*, and the stoneflies *Austroperla cyrene* and *Zelandoperla* sp. (Figure 4). In contrast, only one taxon was exclusively an indicator for pastoral sites: a beetle species in the family Hydrophilidae. Chironomid midge larvae (Diptera) were an indicator taxon for pastoral and horticultural sites (Figure 4).

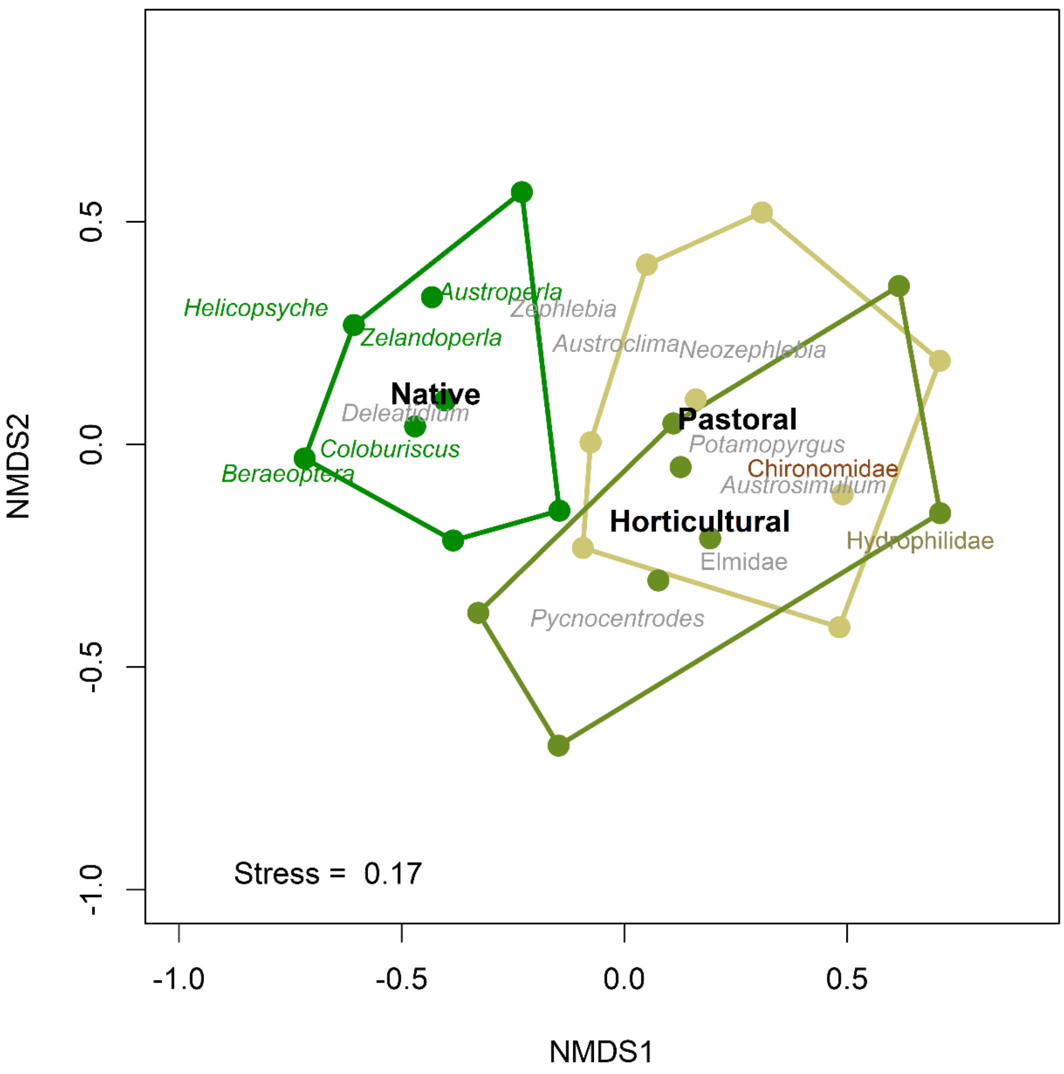

Figure 4. *Non-metric multidimensional scaling (NMDS) plot of macroinvertebrate communities across streams in three local land-use types (green, native forest; khaki, pastoral; olive, horticultural) in the Te Awanui/Tauranga Harbour basin. Indicator taxa are shown by coloured text: green, native forest; khaki, pastoral; brown, pastoral and horticultural. Taxa in the grey text are the ten most abundant found across all sites (including two indicator taxa). Lines indicate convex hulls for each land use type.*

3.2.3 Macroinvertebrate diversity indices

The total abundance of macroinvertebrates recorded from kicknet samples (after accounting for sub-sampling) did not differ significantly between land-use types (native forest, pastoral, horticultural) but did when deposited fine sediment (DFS) was included as a covariate (*P*>0.05; Table 5). However, post-hoc testing of this model confirmed no statistically significant differences in abundance between land-use types (*P*>0.05; Appendix 1, Table 6).

Native forest streams generally had the highest total abundances, followed by lower abundances in pastoral and then horticultural streams, with the latter particularly reflecting the higher levels of DFS found at these sites (Figure 5a). Site 3 (Native forest) had the highest abundance, while Site 18 (Pastoral) had the lowest (Figure 6a). Since the sampling method was semi-quantitative, caution should be used when interpreting these analyses of total abundances (Stark & Maxted, 2007).

Taxa richness differed significantly across land use types, even when DFS was included as a covariate ($P<0.05$, Table 5). Native forest streams generally had the highest taxa richness, followed by pastoral and horticultural streams (Figure 5b). Site 23 (Native forest) had the highest species richness, while Site 18 (Pastoral) had the lowest (Figure 6b). However, post-hoc testing revealed that the only statistically significant difference was between native forest and horticultural streams ($P<0.05$; Appendix 1, Table 6).

Simpson's Diversity Index did not differ significantly across the three site types, with DFS added as a covariate ($P>0.05$) (Table 5). Post-hoc testing further confirmed this finding ($P>0.05$) (Appendix 6). The Simpson's Diversity Index was highest in pastoral streams, followed closely by native forest streams and then horticultural streams (Figure 5c), although Site 2 (horticultural) had the highest Simpson's Diversity Index score but also had the lowest, as shown by site 8 (horticultural); (Figure 6c).

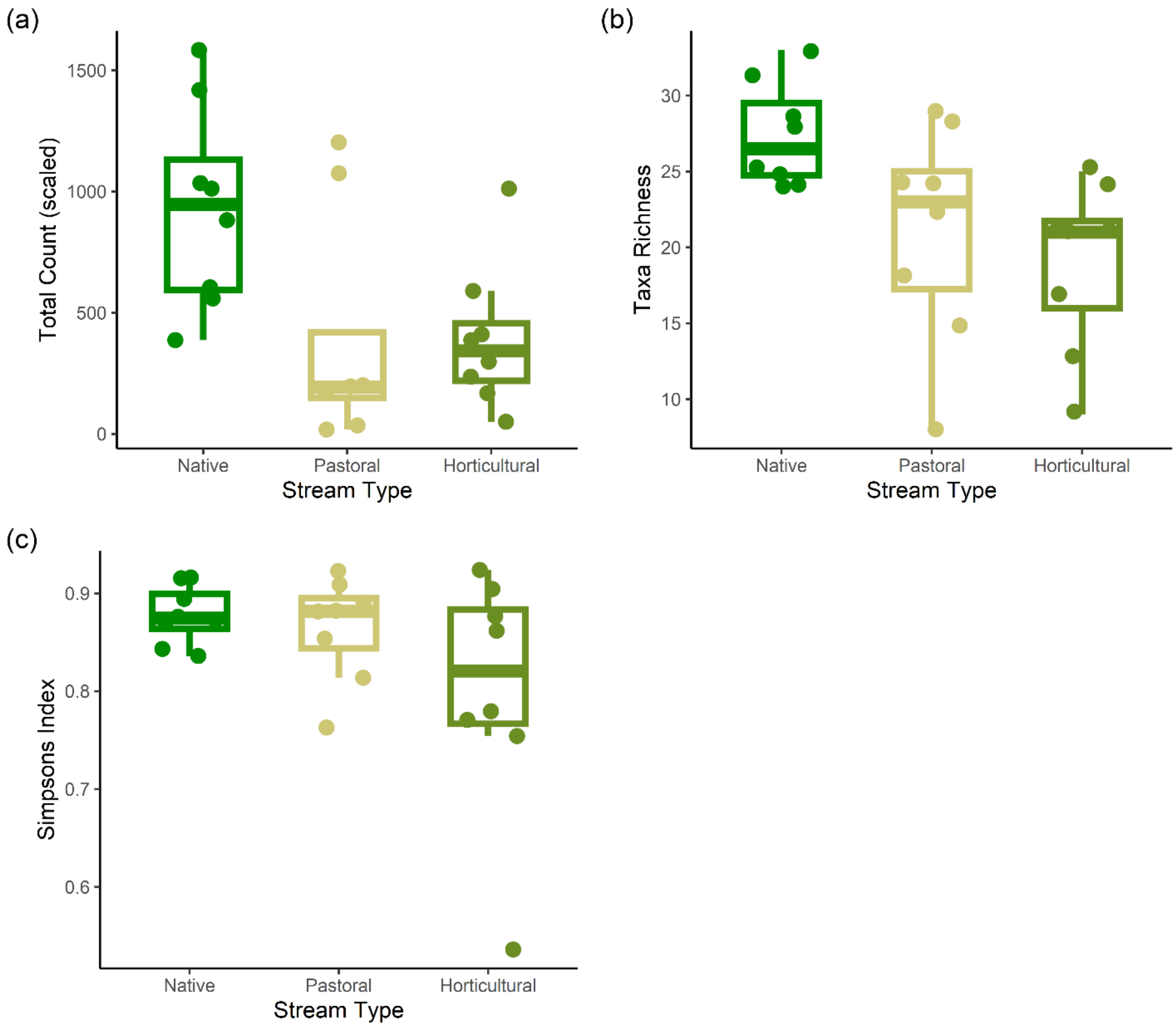

Figure 5. *Boxplots showing the (a) total abundance, (b) taxa richness, and (c) Simpson's Diversity Index of macroinvertebrates sampled across 24 sites in the Te Awanui catchment (Bay of Plenty). Local land-use type is indicated (Native forest, Pastoral, Horticultural)*

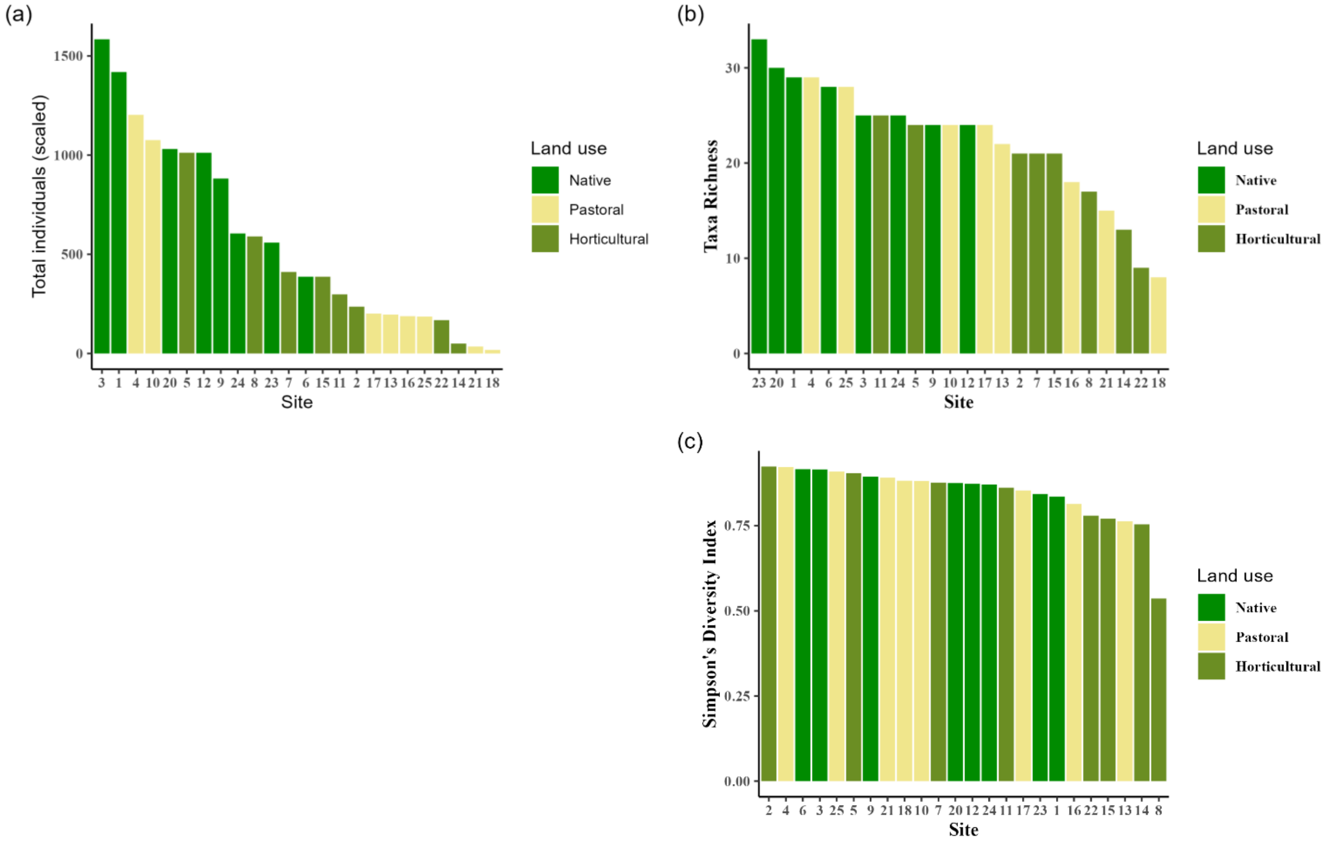

Figure 6. *Barchart shows the (a) total abundance, (b) taxa richness, and (c) Simpson's Diversity Index of macroinvertebrates sampled across 24 sites in the Te Awanui catchment (Bay of Plenty). Local land-use type is indicated (Native forest, Pastoral, Horticultural).*

Table 5. *ANOVA results from mixed models testing the response of macroinvertebrate abundance and diversity indices across three local land-use types (native forest, pastoral and horticultural) with deposited fine sediment (DFS) as a covariate. These analyses are from data collected at the 24 sites sampled in the Te Awanui/Tauranga Harbour catchment.*

Responses	**Predictor**	***F*-value**	***df***	***df*.residual**	**P-value**
(log) Total count	Site type	1.16	2	17.1	0.34
	Sediment	7.97	1	16.8	**0.01**
Taxa richness	Site type	6.03	2	20.0	**0.02**
	Sediment	7.79	1	20.0	**0.01**
(log) Simpsons Index	Site type	1.70	2	16.9	0.21
	Sediment	0.05	1	17.1	0.82

3.2.4 *Macroinvertebrate ranked abundances*

The top ten most abundant macroinvertebrates taxa in native forest streams (Figure 7a) in order from highest to lowest mean abundances were *Deleatidium* (384 individuals), *Coloburiscus* (322), *Helicopsyche* (213), *Neozephlebia* (161), *Pycnocentrodes* (135), *Zephlebia* (133), *Austroclima* (129), *Pycnocentria* (108), *Potamopyrgus* (107), *Beraeoptera* (101); (Table 7).

The top ten most abundant taxa in pastoral streams (Figure 7b) in order from highest to lowest were *Elmidae* (182 individuals), *Austroclima* (149), *Chironomidae* (132), *Neozephlebia* (120), *Pycnocentrodes* (112), *Deleatidium* (108), *Zephlebia* (103), *Austrosimulium* (101), *Potamopyrgu*s (77) and *Aoteapsyche* (56); (Table 7).

The top ten most abundant taxa in horticultural streams (Figure 7c) were *Pycnocentrodes* (535), *Potamopyrgus* (266 individuals), *Neozephlebia* (165), *Aoteapsyche* (137), *Zephlebia* (126), *Elmidae* (85), *Olinga* (78), *Chironomidae* (75), *Austrosimulium* (75) and *Triplectides* (62); (Table 7)

The top ten most abundant taxa overall (Figure 7d) in order from highest to lowest were, *Pycnocentrodes* (782 individuals), *Deleatidiu*m (506), *Potamopyrgus* (450), *Neozepheblia* (446), *Zephlebia* (362), *Coloburiscus* (352), *Austroclima* (330), *Elmidae* (291), *Austrosimulium* (245) and *Chrionomidae* (233); (Table 7).

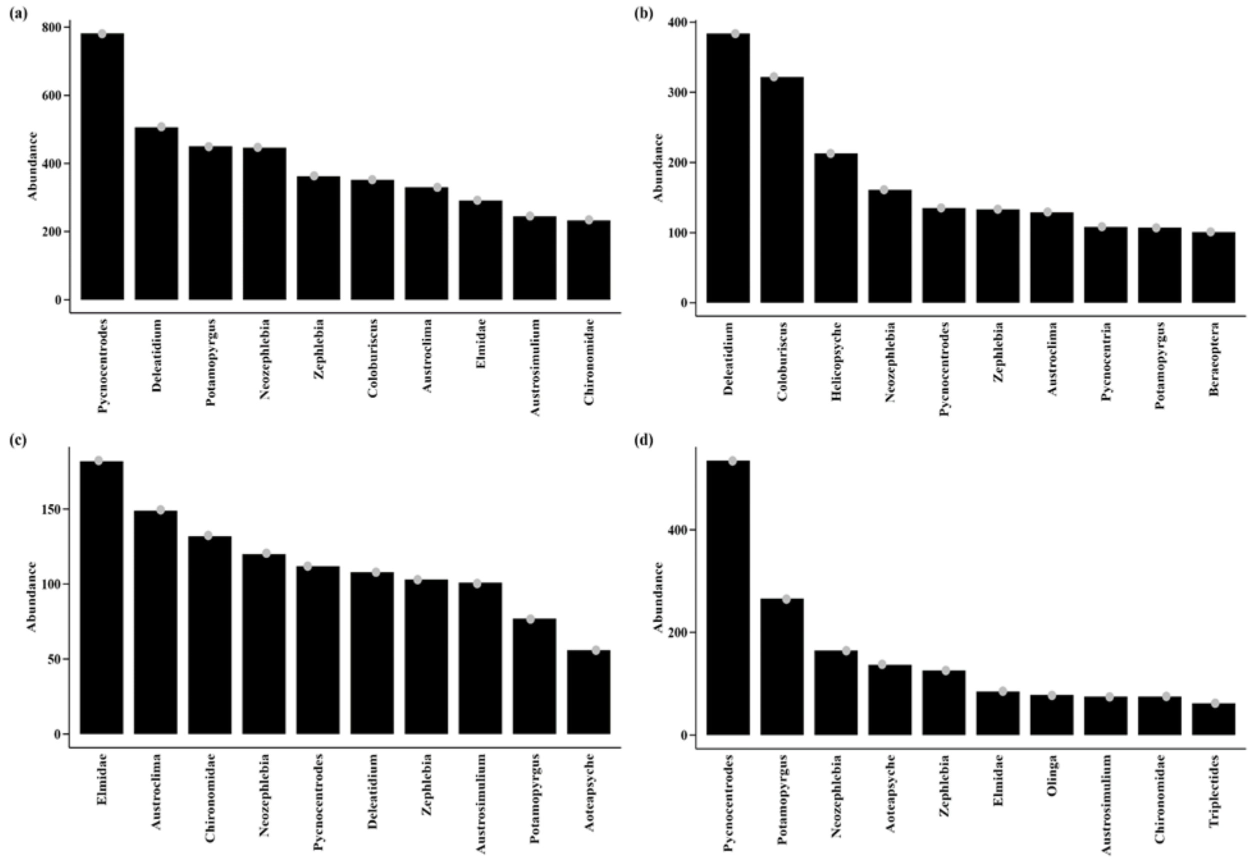

Figure 7. *The top ten most abundant stream macroinvertebrate taxa at sites sampled in the Te Awanui catchment: a) overall, b) in native forest streams, c) pastoral streams and d) horticultural streams.*

Table 7. *Ranked abundance tables displaying the top ten most abundant taxa within native forest streams (top left), pastoral streams (top middle), horticultural streams (top right,) and all sites (bottom left).*

Taxa	**Total**
Deleatidium	384
Coloburiscus	322
Helicopsyche	213
Neozephlebia	161
Pycnocentrodes	135
Zephlebia	133
Austroclima	129
Pycnocentria	108
Potamopyrgus	107
Beraeoptera	101

Taxa	**Total**
Elmidae	182
Austroclima	149
Chironomidae	132
Neozephlebia	120
Pycnocentrodes	112
Deleatidium	108
Zephlebia	103
Austrosimulium	101
Potamopyrgus	77
Aoteapsyche	56

Taxa	**Total**
Pycnocentrodes	535
Potamopyrgus	266
Neozephlebia	165
Aoteapsyche	137
Zephlebia	126
Elmidae	85
Olinga	78
Chironomidae	75
Austrosimulium	75
Triplectides	62

Taxa	**Total**
Pycnocentrodes	782
Deleatidium	506
Potamopyrgus	450
Neozephlebia	446
Zephlebia	362
Coloburiscus	352
Austroclima	330
Elmidae	291
Austrosimulium	245
Chironomidae	233

3.2.5 National Objective Framework Bands

Using the macroinvertebrate indicators and biocriteria stated in the NPS-FM (2020) indicated that native forest streams were generally in the 'A' band, with pastoral and horticultural sites more likely to be in the 'B' and 'C' bands (Figure 8). There were a few instances where a site fell below the national bottom line in the 'D' band (Figure 8).

The MCI indicated similar patterns (Figure 8a). The seven sites in the 'A' band were all native forest streams. Twelve sites were in the 'B' band, with six having horticulture as their local land-use type. Five sites were in the 'C' band, with three having pasture as their local land-use type. No sites were in the 'D' band for the MCI (Figure 8a).

Using the QMCI indicated that nine sites were in the 'A' band, with six of these being native forest streams (Figure 8b). Seven sites were in the 'B' band. Five of the seven sites in the 'C' band had horticulture as their local land-use type. One pastoral site was in the 'D' band for the QMCI (Figure 8b).

The ASPM showed patterns consistent with the MCI and its quantitative equivalent (Figure 8c). Eight of the eleven sites in the 'A' band were native forest streams. The eight sites in the 'B' band were evenly split between having horticulture and pasture as their local land-use type. Four sites were in the 'C' band, with three having pasture as their local land-use type. A single site horticultural site was in the 'D' band for the ASPM (Figure 8c). EPT richness (Figure 8d) and relative abundances (Figure 8e) indicated that native forest streams generally had the best values for these indicators, with pastoral and horticultural streams showing a decline. While these results indicate trends in ecosystem health, caution should be given since the data presented was derived from a singular set of samples and not the minimum of 3 years of continuous annual sampling required by the NPS-FM to accurately calculate NOF bands (Collier et al., 2014).

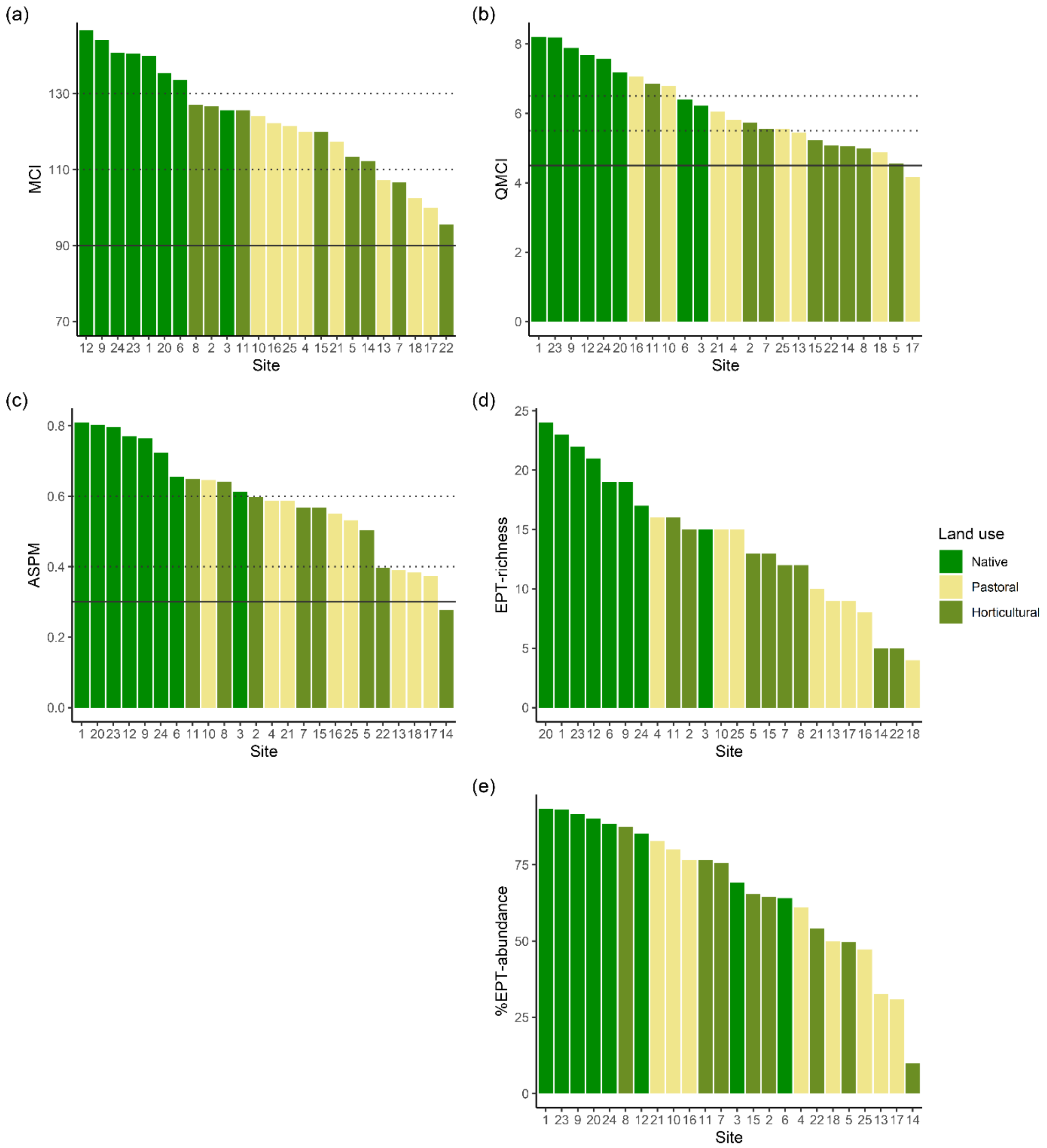

Figure 8. *Macroinvertebrate indices from stream sites in the Te Awanui/Tauranga Harbour basin. a) the Macroinvertebrate Community Index, and b) its quantitative equivalent, the QMCI; c) the Average Score Per Metric (APSM) index, including its constituent indices d) EPT taxa richness, and e) EPT relative abundance. Indices are ranked from highest to lowest, and the local land-use type is indicated. The horizontal lines indicate the biocriteria stated in the NPS-FM; the dotted lines indicate bands A-C; the solid line indicates a D status or below the national bottom line. Care is needed to interpret this data since it reflects only one sampling date and not the median values of five years of sampling required in the NPS-FM.*

3.2.6 NPS-FM metrics regarding land-use type and sediment

Macroinvertebrate Community Index (MCI) scores were higher in native forest streams and relatively similar between pastoral and horticultural streams (Figure 9a). The difference in MCI scores between the three land-use types was statistically significant (*P*<0.001; Table 8) with deposited fine sediment (DFS) included as a significant covariate (*P*<0.001; Table 8). Post-hoc testing indicated that the only statistically significant contrast was between native forest streams and horticultural streams (*P*<0.01); Appendix 1; Table 9). There were no significant differences in the MCI scores between native streams and pastoral streams or between horticultural streams and pastoral streams (Appendix 1, Table 9).

The quantitative version of the MCI, the QMCI, showed scores significantly higher in native forest streams, followed by pastoral and then horticultural streams, which had relatively similar QMCI scores (Figure 9b). The difference in QMCI scores between the three land-use types was statistically significant (*P* <0.001; Table 8) with sediment included as a significant covariate (*P*<0.01; Table 8). Post-hoc testing indicated that the only statistically significant difference was between native forest and horticultural streams (*P*<0.001; Appendix 1, Table 9). There was no statistically significant relationship between native forest and pastoral streams or between pastoral and horticultural streams (Appendix 1, Table 9).

The relative abundance of EPT taxa (%EPT abundance) was highest in native forest streams, followed by horticultural and then pastoral streams, which were relatively similar (Figure 9c). These differences were statistically non-significant (*P*>0.05) (Table 8) with sediment included as a significant covariate (*P*<0.05) (Table 8). Post-hoc analysis testing confirmed no statistically significant differences between land-use types (Appendix 1, Table 9).

The taxa richness of EPT taxa was highest in native forest streams, followed by horticultural and then pastoral streams, which were relatively similar (Figure 9d). The difference in the richness of EPT taxa between the three land-use types was statistically significant (*P*<0.05; Table 8) with sediment included as a signficant covariate (*P*<0.05; Table 8). Post-hoc testing revealed that the only significant difference was between native forest and pastoral streams (*P* <0.01; Appendix 1, Table 9). There were no significant differences between native forest and horticultural streams or between pastoral and horticultural streams (Appendix 1, Table 9).

ASPM scores were highest in native forest streams, followed by horticultural and pastoral streams (Figure 9e). Native and horticultural streams had relatively similar ASPM scores (Figure 9e). The difference in ASPM scores between the three land-use types was statistically significant (*P* <0.05); Table 7) with sediment included as a significant covariate (*P*<0.001; Table 7). Post-hoc testing indicated that the only statistically significant difference was between native forest streams and horticultural streams (*P*<0.01; Appendix 1, Table 9). There were no significant differences in the ASPM scores between native and pastoral streams or between horticultural and pastoral streams (*P*>0.05; Appendix 1, Table 9).

The relative abundance of 'decreaser' taxa was higher in native forest streams, followed by horticultural streams and then pastoral streams, which were relatively similar (Figures 10a & 11a). This result was not significant across the three different land-use types (native forest, pastoral, and horticultural) (*P*>0.05; Table 8) with DFS included as a significant covariate (*P*<0.05; Table 8). Post-hoc testing confirmed no significant differences between streams of any land-use type (*P* >0.05; Appendix 1, Table 9).

The richness of 'decreaser' taxa in native forest streams was higher in native forest streams, followed by horticultural and pastoral streams, which were relatively similar (Figures 10b & 11b). This result was statistically significant across the three different land-use types (Native forest, pastoral, horticultural) (<0.05; Table 8) with sediment as a significant covariate (*P* <0.01; Table 8). Post hoc testing indicated that the only significant difference was between native forest and horticultural streams (*P*<0.05; Appendix 1, Table 9).

The relative abundance of 'increaser' taxa was highest in horticultural streams, followed by pastoral and native forest streams (Figures 12a & 13a). 'Increaser abundance' was relatively similar across pastoral & and horticultural sites (Figure 12a). However, this result was not statistically significant across the three different land-use types (native forest, pastoral, and horticultural) (*P*>0.05; Table 8) with sediment added as a covariate (*P*>0.05; Table 8). Post-hoc testing revealed no statistically significant result between any land-use type (*P* >0.05; Appendix 1, Table 9).

The richness of 'increaser' taxa was very similar in both pastoral and horticultural streams, with native forest streams having the lowest 'increaser' richness (Figures 12b & 13b). The richness of 'increaser taxa was relatively similar across pastoral and horticultural land-uses (Figure 12b). However, this result was not statistically significant across the three different land-use types (native forest, pastoral, and horticultural) (*P*>0.05; Table 8) with sediment added as a covariate

(*P*>0.05; Table 9). Furthermore, post-hoc testing revealed no statistically significant result between any land-use type (*P*>0.05; Appendix 1, Table 9).

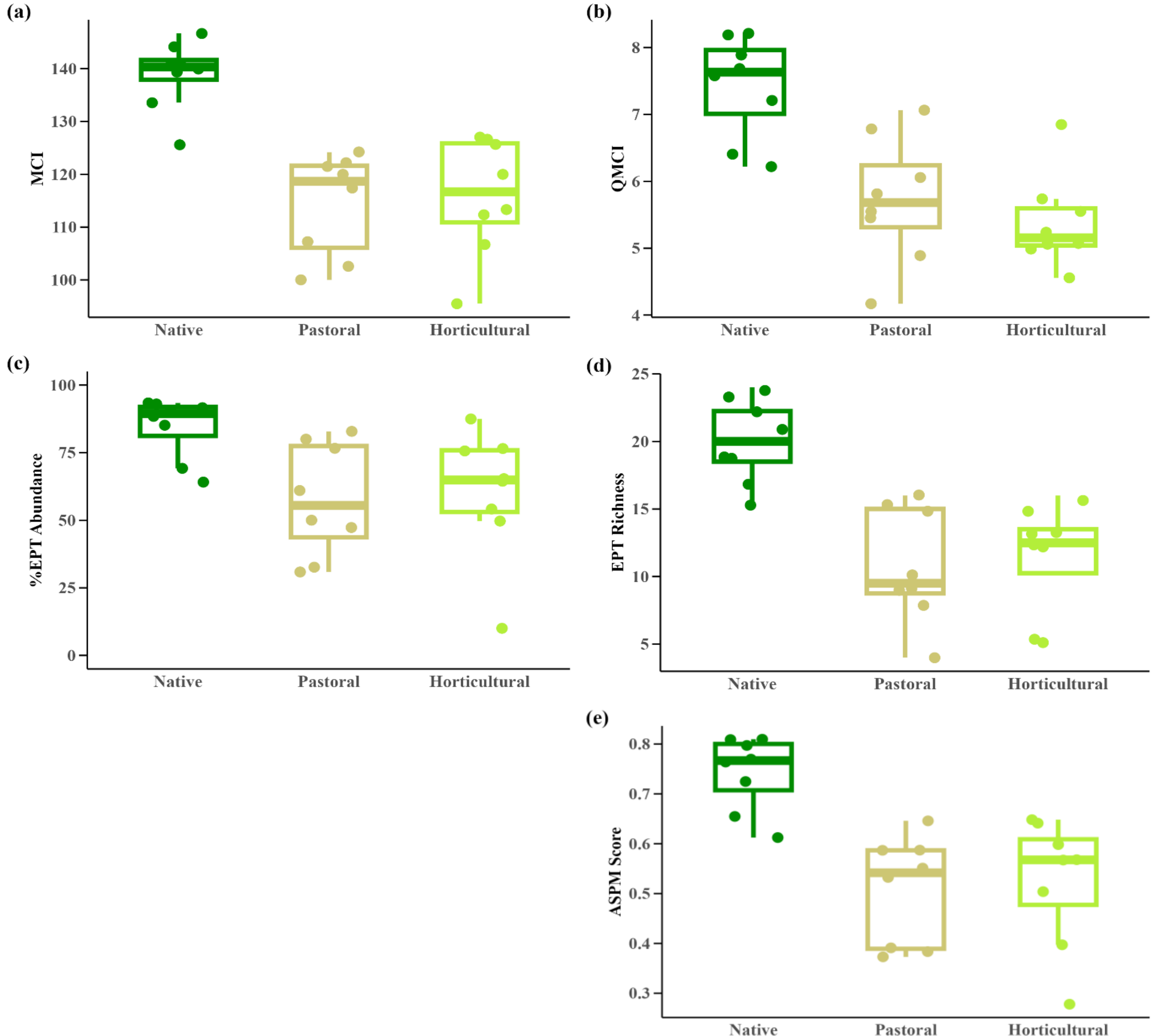

Figure 9. *Changes in key macroinvertebrate indicators across streams in three local land-use types in the Te Awanui/Tauranga Harbour basin. a) Macroinvertebrate Community Index (QMCI), b) Quantitative macroinvertebrate Community Index (MCI), c) EPT relative abundance (%EPT Abundance), d) EPT taxa richness (EPT richness), e) Average Score Per Metric (ASPM) index.*

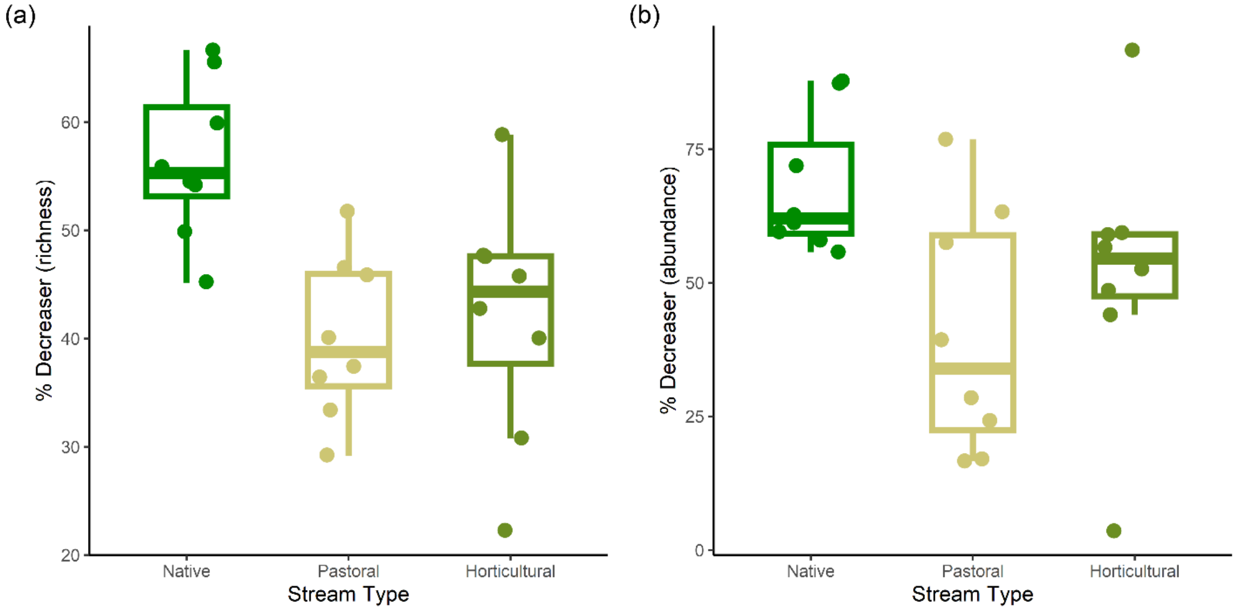

Figure 10. *Boxplots of the changes in sediment-specific macroinvertebrate indicators across streams in three local land-use types in the Te Awanui/Tauranga Harbour basin. a) % sediment 'decreasers' taxa richness, and b) % sediment 'decreasers' abundances.*

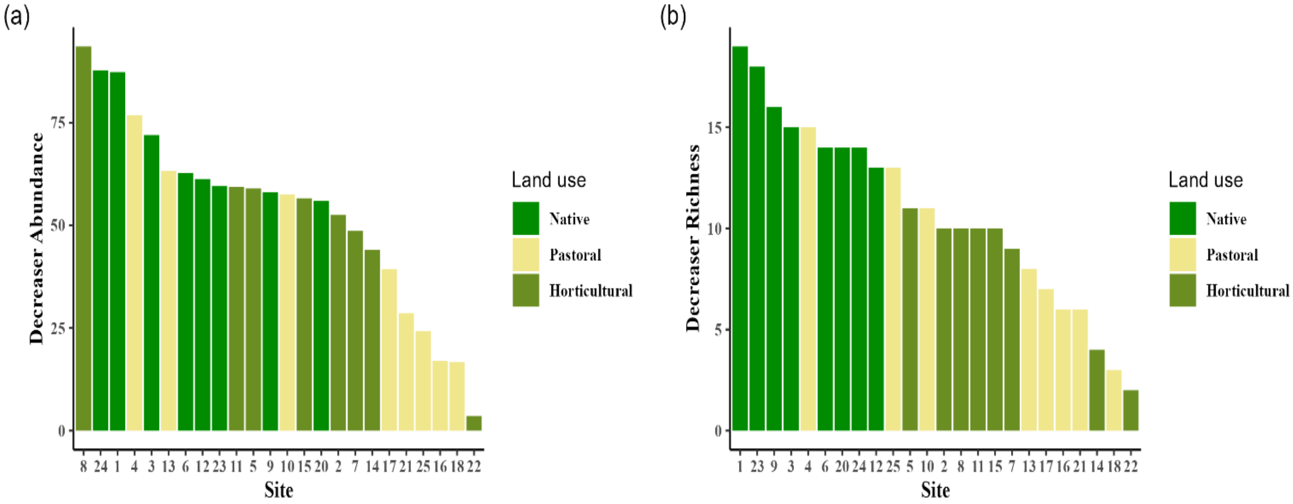

Figure 11. *Barcharts of the changes in sediment-specific macroinvertebrate indicators across streams in three local land-use types in the Te Awanui/Tauranga Harbour basin. a) % sediment 'decreasers' abundances and b) % sediment 'decreasers' taxa richness.*

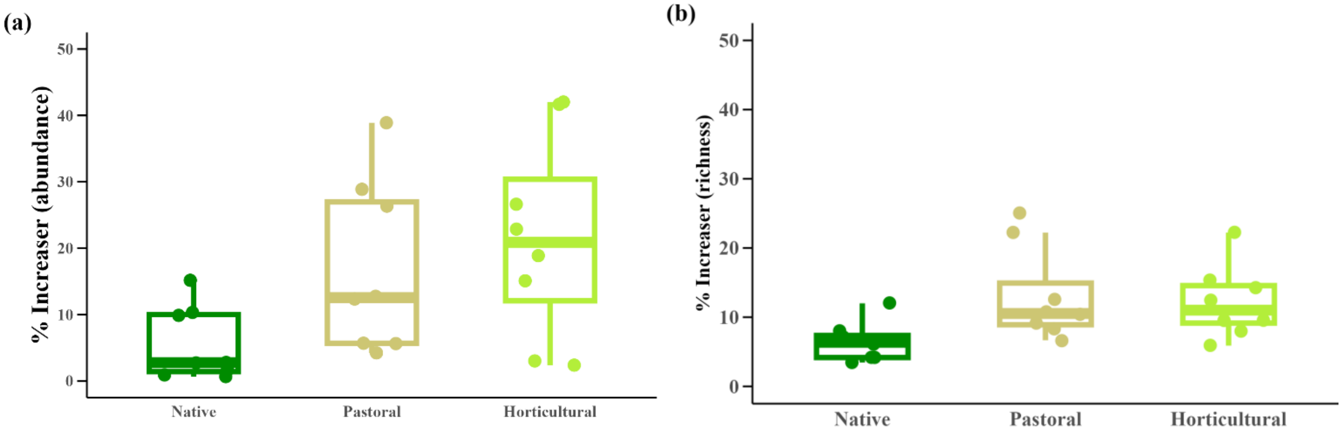

Figure 12. *Boxplots of the changes in sediment-specific macroinvertebrate indicators across streams in three local land-use types in the Te Awanui/Tauranga Harbour basin. a) % sediment 'increaser' taxa richness, and b) % sediment 'increaser' abundances.*

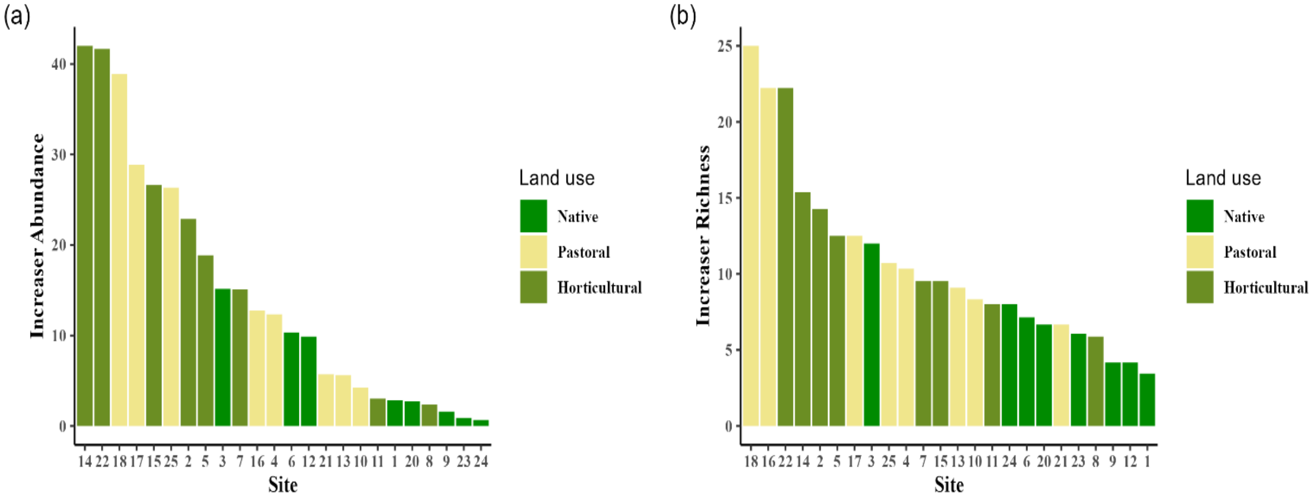

Figure 13. *Barcharts of the changes in sediment-specific macroinvertebrate indicators across streams in three local land-use types in the Te Awanui/Tauranga Harbour basin. a) % sediment 'increaser' taxa richness, and b) % sediment 'increaser' abundances.*

Table 8. *ANOVA results from mixed-models testing changes in standardized macroinvertebrate indices across streams in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	Predictor	F-value	df	df.res	P-value

MCI	Site type	11.9	2	15.6	**<0.001**
	Sediment	20.3	1	20	**<0.001**
QMCI	Site type	12.8	2	14.7	**<0.001**
	Sediment	7.28	1	18.9	**0.01**
ASPM	Site type	8.12	2	14.7	**0.004**
	Sediment	16	1	18.9	**<0.001**
EPT Richness	Site type	12.9	2	20	**0.01**
	Sediment	7.22	1	20	**0.01**
% EPT Abundance	Site type	2.6	2	16	0.11
	Sediment	4.61	1	19.9	**0.04**
% Decreaser Richness	Site type	5.53	2	16.5	**0.02**
	Sediment	11.8	1	19.4	**0.01**
% Decreaser Abundance	Site type	0.51	2	17.1	0.61
	Sediment	6.46	1	18.1	**0.02**
% Increaser Abundance	Site type	2.67	2	17.2	0.10
	Sediment	2.25	1	17.8	0.15
% Increaser Richness	Site type	2.12	2	16.8	0.15
	Sediment	1.80	1	18.8	0.20

3.2.7 NPS-FM macroinvertebrate metrics and sediment

A series of linear regression analyses were performed to investigate the relationship between key NPS-FM 2020 macroinvertebrate metrics and Deposited Fine Sediment (DFS). The data for these analyses were derived from all 24 sites in the Te Awanui/Tauranga Harbor catchment.

The first analysis revealed a statistically significant but weak correlation between QMCI scores and DFS ($P<0.05$; Figure 14a). This relationship was negative as QMCI scores decreased as DFS levels increased (Figure 14a).

The second analysis revealed a statistically significant, moderate correlation between MCI scores and DFS ($P<0.001$; Figure 14b). This relationship was negative as MCI scores decreased as DFS levels increased (Figure 14b).

The third analysis revealed a weak correlation between EPT abundance and DFS but was statistically significant ($P<0.05$); Figure 14c). This relationship was negative as EPT abundance decreased as DFS levels increased (Figure 14c).

The fourth analysis revealed a statistically significant but weak correlation between EPT taxa richness and DFS ($P<0.01$; Figure 14d). This relationship was negative as EPT richness scores decreased as DFS levels increased (Figure 14d).

The fifth analysis revealed a moderate correlation between ASPM scores and DFS that was statistically significant (*P*<0.001; Figure 14e). This relationship was negative as ASPM scores decreased as DFS levels increased (Figure 14e).

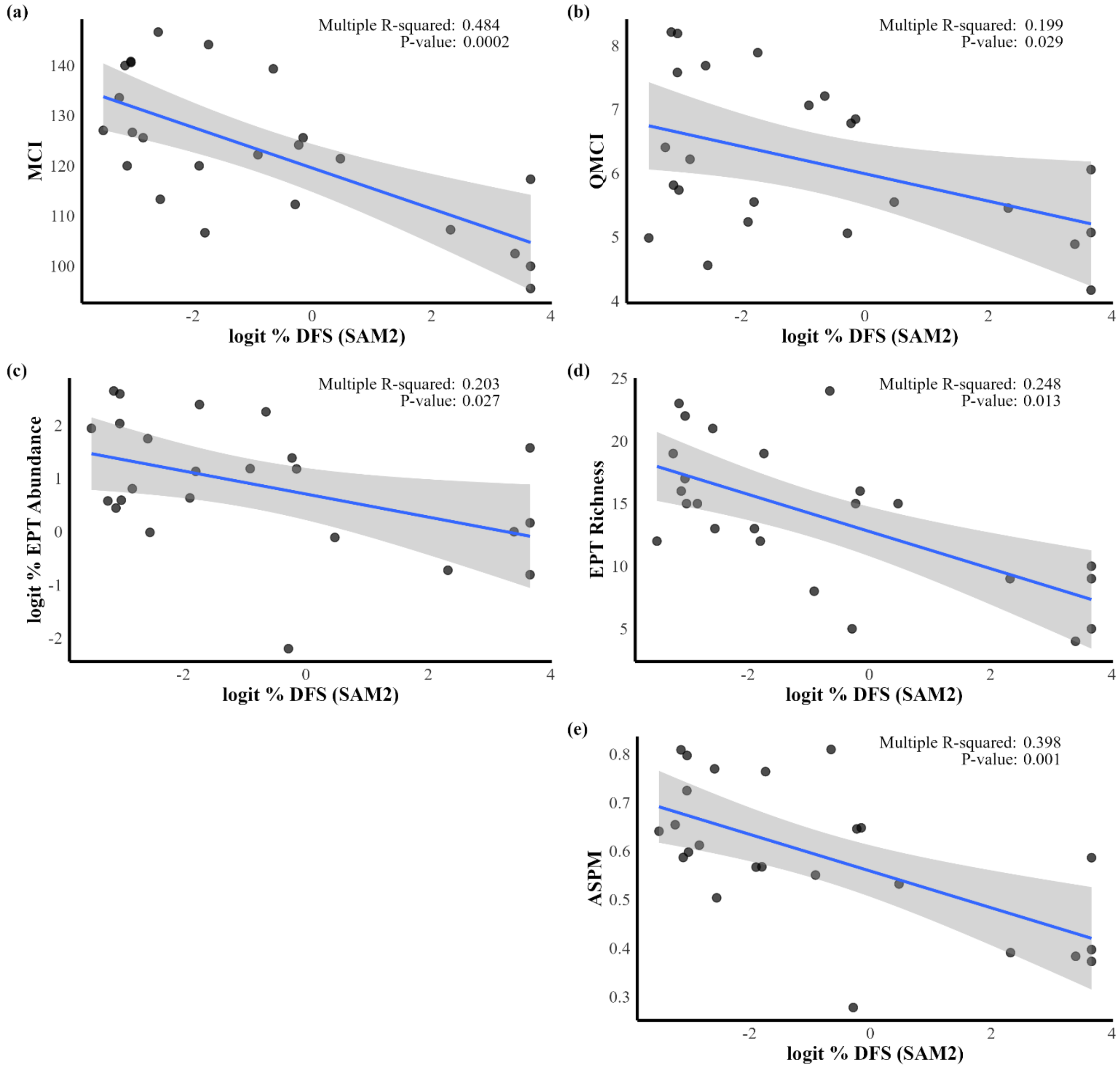

Figure 14. *Linear regressions showing the relationships between Deposited Fine Sediment (DFS) and a) QMCI, b) MCI, c) (logit) EPT abundance, d) EPT Richness and e) ASPM in the Te Awanui/Tauranga Harbour basin. The grey ribbon indicates the 95% confidence interval for the regression line.*

3.2.8 Sediment-specific macroinvertebrate metrics and sediment

A total of four linear regression analyses were performed to investigate the relationship between sediment-specific macroinvertebrate metrics and DFS. The data for these analyses were derived from all 24 sites in the Te Awanui/Tauranga Harbor catchment.

The first analysis revealed a statistically significant moderate correlation between ‘decreaser’ abundance and DFS (*P*<0.001; Figure 15a). This relationship was negative as the richness of ‘decreaser’ taxa decreased as DFS levels increased (Figure 15a).

The second analysis revealed a statistically significant moderate correlation between ‘decreaser’ richness and DFS (*P*<0.001; Figure 15b). This relationship was negative as the richness of ‘decreaser’ taxa decreased as DFS levels increased (Figure 15b).

The third analysis revealed a statistically but weak correlation between ‘increaser’ abundance and DFS that was marginally non-significant (*P*=0.051; Figure 16a). The relationship was positive as the abundances of ‘increaser’ taxa increased as DFS levels increased (Figure 16a).

The fourth analysis revealed a weak correlation between ‘increaser’ richness and DFS, which was statistically significant (*P*<0.50; Figure 16b). The relationship was positive as the richness of ‘increaser’ taxa increased as DFS levels increased (Figure 16b).

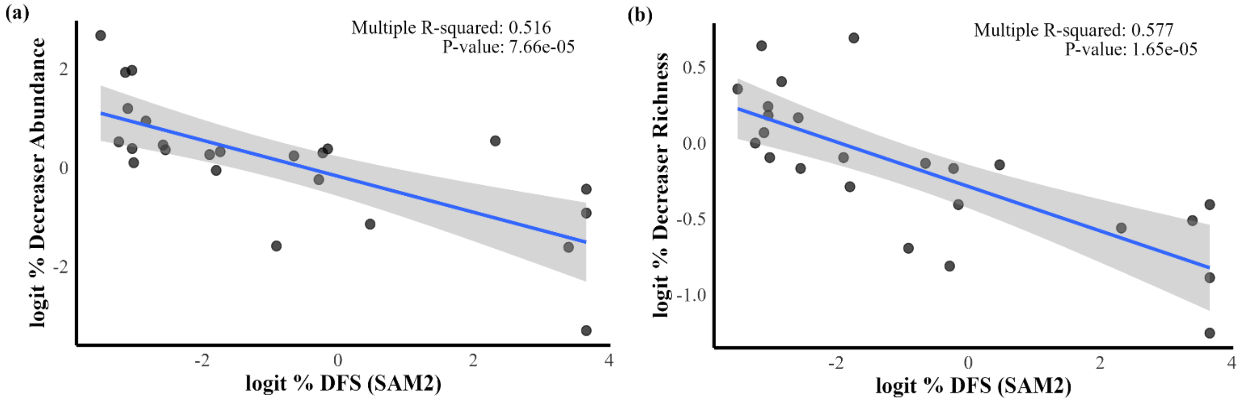

Figure 15. *The relationships between Deposited Fine Sediment (DFS) and a) decreaser abundance, b) decreaser richness.*

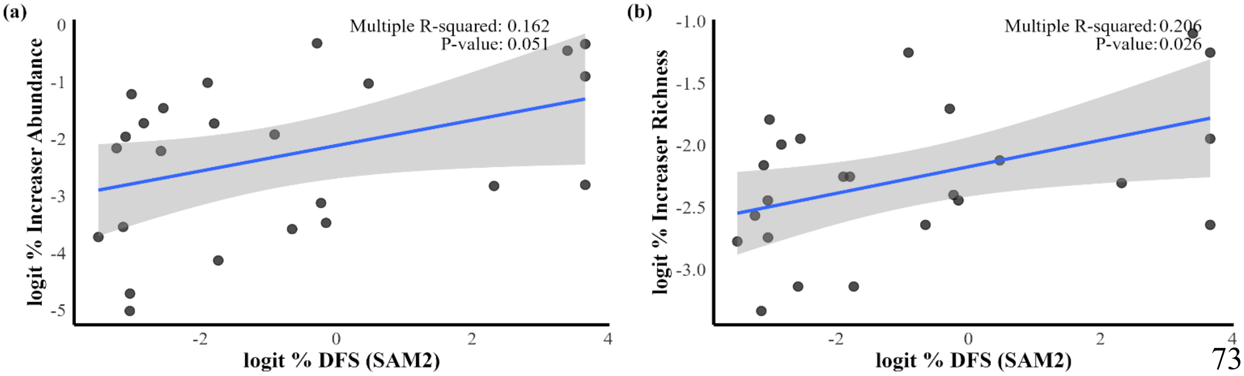

Figure 16. *Linear regressions showing the relationships between Deposited Fine Sediment (DFS) and a) increaser abundance, b) increaser richness.*

3.3 Discussion

3.3.1 Sediment

My results confirmed the negative influence of deposited fine sediment (DFS) on stream ecosystems, as indicated by macroinvertebrate indices analysed. For instance, all NPS-FM metrics (MCI, QMCI, and ASPM) decreased in response to heightened levels of DFS. Furthermore, these metrics responded to sediment despite the differences between land use types (native forest, pastoral & horticultural).

These results supported my first hypothesis predicting that DFS would be the primary factor driving variations between stream sites across the three different land uses sampled. However, there was also support for my alternative hypothesis that any remaining differences not explained by sedimentation would indicate the influence of other stressors, such as nutrient enrichment, flow alteration, and loss of riparian vegetation. This finding confirms that in addition to sedimentation, there are potentially other stressors associated with human land uses that impact stream macroinvertebrates and ecosystem health (Jackson et al., 2021; Judi Hewitt et al., 2014; Lange et al., 2014; Matthaei et al., 2010; Townsend et al., 2008; Villeneuve et al., 2018; Wagenhoff et al., 2011). Significant variations in DFS were observed among the different land-use types (native forest, pastoral, and horticultural). Specifically, DFS was notably higher in pastoral streams, followed by horticultural and then native forest streams. These findings align with previous studies highlighting increased DFS levels in streams adjacent to pastoral and horticultural land use, with streams near native forests exhibiting minimal DFS levels in comparison (Davis et al., 2022; Quinn & Stroud, 2002; Young et al., 2005).

The prevalence of DFS in pastoral and horticultural streams can be primarily attributed to the degradation of riparian habitats along stream margins, which was observed in this study (Allan, 2004; Burdon et al., 2013). Removing riparian vegetation leads to bank instability, escalating erosion rates and subsequent sedimentation (Allan, 2004). Additionally, activities such as livestock tramping and horticultural tilling contribute to sedimentation through runoff (Quinn, 2000).

In the context of this study, the influence of anthropogenic activities on sediment deposition rates has been extensively studied, as observed in the investigation by Hume et al. (2010b), which investigated the issue of sedimentation within the Tauranga Harbour catchment. The study by Hume et al. (2010b) revealed compelling insights into the differential contributions of land-use types to sediment load. Notably, pasture, covering 34% of the catchment area, accounts for the largest sediment load (62%). In comparison, native bush, scrub, and forest encompass a larger proportion of the catchment (44%) but only contribute 27% to the total sediment load (Hume et al., 2010b).

Although the contribution of horticultural land use towards sediment loads is minimal, its expansion, albeit less than 2% of the catchment, warrants attention (Bay of Plenty Regional Council, 2023a; Hume et al., 2010b). Despite the paucity of data regarding the rate and area of its expansion in the Bay of Plenty, national trends showed that the area of irrigated land doubled between 2002 and 2019, from 384,000 hectares to 735,000 hectares (91% increase), with 18% of this increase being attributed to horticulture (Ministry for the Environment, 2023). However, there is a lack of research measuring horticultural sediment contributions and related ecological changes in freshwater stream/riverine ecosystems (Larned et al., 2018)

However, the most relevant and recent finding was from a NIWA report, which found a significant correlation between horticultural land use and elevated sediment levels (Whitehead et al., 2022). Therefore, it would be wise to monitor this expansion, given that horticultural and pastoral activities share analogous pathways in elevating sediment loads (Allan, 2004).

Although anthropogenic land use has been heavily linked to increased sedimentation rates (Allan & Johnson, 1997), it is important to highlight the influence these sites' underlying geology can have on DFS levels (Allan, 2004). It is critical to differentiate between natural and anthropogenically influenced sources of DFS to prevent attributing solely human activity to elevated sediment loads in streams (Shearer & Young, 2011). Geological factors such as soil type, weathering processes and erosion susceptibility of different rock formations can significantly contribute to DFS levels in streams (Shearer & Young, 2011). Understanding the geological context of a region is crucial in assessing baseline sedimentation levels and distinguishing between background natural sedimentation and increased sedimentation due to human disturbances (Shearer & Young, 2011).

For instance, the Western Bay of Plenty's geological composition predominately consists of ignimbrite, a relatively weak rock characterized by high friction angles, rendering it susceptible to erosion (Moon, 1993). This fragility is underscored in the study by Glenjarman (2017), which emphasizes the propensity of ignimbrite material to form highly unstable stream beds, prone to erosion even under low water flow velocities. Furthermore, the friability of ignimbritic/igneous material has resulted in the two primary soil types of the Bay of Plenty being volcanic (Hume et al., 2010b; Rijkse & Guinto, 2010). Orthic allophanic soil, constituting 56.2% of the total soil area in the catchment, contributes an annual sediment load of 1.95 tonnes per hectare annually (Hume et al., 2010b). Similarly, orthic podzols, accounting for 30.1% of the total soil area, contribute 2.02 tonnes per hectare annually to the sediment load (Hume et al., 2010b). Although it is clear that the underlying geology of the Bay of Plenty plays a significant role in sedimentation, it is poor management of anthropogenic land-uses that greatly exacerbates this issue and the effects that it has on stream ecosystems (Allan, 2004; Larned et al., 2019).

As discussed previously, sedimentation has numerous adverse effects on macroinvertebrate communities and stream ecosystems. It alters benthic habitats by filling interstitial spaces, reducing suitable habitats for macroinvertebrates (Burdon et al., 2013). It also increases streambed instability, limiting habitat availability during high water flows (Shaw & Richardson, 2001). Moreover, elevated water turbidity obstructs essential activities of macroinvertebrates, such as foraging and mating (Rowe & Dean, 1998). Sedimentation can cause physical harm through abrasion (Francoeur & Biggs, 2006) and clogging of macroinvertebrate gills (Lemly, 1982), affecting their survival and impairing filter-feeding organisms (Forbes-Green & Cyr, 2023).

It is these adverse effects that likely contributed significantly to the consistent decline in key macroinvertebrate metrics in this research. For instance, MCI, QMCI, & ASPM scores, EPT abundance, and richness all decreased with rising levels of DFS. Moreover, sediment-specific metrics displayed a clear response, as 'decreaser' abundance and richness decreased while 'increaser' abundance and richness decreased with greater DFS cover. Although all of these results were statistically significant, except for 'increaser' abundance, which was marginally significant, the strongest relationship identified was between MCI and DFS. In contrast, all other metrics exhibited weaker correlations with DFS.

Furthermore, the lowest MCI, QMCI, and ASPM scores, along with EPT and 'decreaser' abundance and richness, were found in pastoral and horticultural sites where DFS levels were the highest, compared to native forest streams. The same pattern followed for additional metrics of macroinvertebrate communities, such as total abundance and overall taxa richness. Surprisingly, pastoral sites had the highest Simpsons Diversity Score; however, this was by a very slim margin compared to native forest streams and was not statistically significant. Greater diversity in pastoral streams could be due to the relative abundance of macroinvertebrates in pastoral streams being slightly more equally distributed across taxa, as native forest streams had higher taxa richness. Nevertheless, these observations further reinforce the adverse effects that anthropogenic land use has on stream ecosystems via sedimentation (Allan, 2004; Larned et al., 2018).

My findings are consistent with prior studies that have reported a decrease in these macroinvertebrate metrics with increasing DFS levels due to the adverse effects it imposes on macroinvertebrate communities (Clapcott et al., 2011; Davis et al., 2022; Niyogi et al., 2007; Oliveira et al., 2020). However, the surprisingly weak correlations between many macroinvertebrate metrics and DFS require further explanation. Such insights are provided by the study by Davis et al. (2022), which observed similar results. Specifically, in the study by Davis et al. (2022), such correlations only became prominent at moderate to high levels of land-use intensity, an aspect not measured in this study. Several explanations were offered that could shed some light on the results reported in this study.

One explanation is that in the study by Davis et al. (2022), it was reported that sites with low land-use intensity tend to have relatively stable DFS levels over time. Consequently, macroinvertebrate communities in such areas might exhibit a limited response to DFS, as they may have adapted to the existing DFS levels. This aligns with the proposition that, at lower levels, DFS might provide extra habitat for burrowing macroinvertebrates while preserving DFS-free habitats for species sensitive to sediment.

Another explanation is that higher-order streams are more susceptible to sediment impact than lower-order streams. (Davis et al., 2022). It was found in the study by Davis et al. (2022) that lower-order streams possess better sediment-dispersal abilities due to their steeper gradient and greater width, whereas sediment tends to accumulate more in higher-order streams due to reduced dispersal capacity. This explanation was supported by the study by Wagenhoff et al. (2011), which found a positive relationship between suspended sediment and % catchment runoff for streams orders 4-6 but not for 3rd-order streams. Furthermore, also referenced within the study by Davis et al. (2022) was the study by Naden et al. (2016b), which measured DFS from 230 agricultural streams across England and Wales and found stream power (measured by median annual flooding & channel slope) to be the most effective predictor of DFS (Naden et al., 2016b). While this explanation is not immediately applicable to the findings in this study, given that approximately 79% of our streams were third-order or less (Table 1), the study by Davis et al. (2022) provides further insight.

It was observed that DFS levels varied dramatically annually across its 46 sites, regardless of stream order over the 5 years this study was conducted Davis et al., (2022). This finding implies that DFS levels recorded in the 24 sampled streams during this research might have been comparably lower than the average annual levels. However, channel slope does not seem to clarify this variability due to the predominance of low-ordered streams in this study (Davis et al., 2022).

Another element linked to ‘stream power’ that could shed light on this observation is annual flooding, primarily impacted by rainfall. For instance, (Chappell, 2014) delves into the climatic patterns of the Bay of Plenty Region. It underscores the region’s notable seasonal variability in rainfall, attributed to the significant shelter provided by geological formations like the Kaimai ranges and the Mamaku Plateau. This dependence on northerly airstreams for a substantial portion of rainfall characterizes the region (Chappell, 2014).

Moreover, the study highlights heightened rainfall variability, particularly in the Western Bay of Plenty, where many study sites were located. It points out that the most substantial variability in rainfall occurs during the summer and autumn months (Chappell, 2014). Given that this study took place in October, it is plausible that seasonal rainfall patterns might have influenced the DFS levels in the streams prior to sampling, potentially reducing the correlation between DFS and recorded macroinvertebrate metrics (Chappell, 2014).

Despite the moderate correlation at its strongest, the results underscore the adverse impacts of sedimentation on stream ecosystem health and the supported macroinvertebrate communities. Numerous studies investigating sedimentation across multiple spatial scales have firmly established the severity of these adverse effects (Burdon et al., 2013; Davis et al., 2022; Matthaei & Piggott, 2019; Matthaei et al., 2010; Oliveira et al., 2020; Scarsbrook et al., 2016; Wagenhoff et al., 2011). For instance, the survey by Burdon et al. (2013) conducted across 30 streams in the Canterbury region (New Zealand) revealed that sedimentation significantly contributed to stream habitat degradation and its associated macroinvertebrate communities. Notably, this study observed a strong negative non-linear response of pollutant-sensitive macroinvertebrate taxa

(EPT) when streambed DFS coverage exceeded 20% at the reach scale. Additionally, the study by Pingram et al. (2019) emphasized the removal of DFS as the most effective action for enhancing stream ecosystems in the Waikato region. These studies collectively emphasize sedimentation as a 'Master stressor' with persistent and detrimental effects on stream ecosystems, as Blöcher et al. (2020) exemplified.

3.3.2 Land-use

In my study, the Macroinvertebrate Community Index (MCI) and its quantitative counterpart QMCI consistently demonstrated higher scores in native forest streams in comparison to pastoral and horticultural streams, both which exhibited relatively similar and lower values for these metrics. Furthermore, differences among land uses were significant even with including DFS (Deposited Fine Sediment) as a covariate. Intriguingly, post-hoc analysis indicated significant differences exclusively between native forest and horticultural streams in both assessments after controlling for the influence of sediment.

Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa abundance and taxa richness was higher in native forest streams compared to horticultural and pastoral streams. However, differences among land use types were not significant with the inclusion of DFS as a significant covariate, highlighting that DFS explained much of the land-use impact on pollutant-sensitive taxa.

The Average Score Per Metric (ASPM) score is derived from the previous the MCI and EPT metrics (abundance and richness), making it no surprise that the ASPM supported the trend that showed greater ecosystem health in native forest streams, in comparison to horticultural and pastoral streams, which were relatively similar. There was a significant influence of land use even with the inclusion of DFS as a covariate, reflecting the potential additional stressors acting on macroinvertebrate communities across land-use types.

Assessment of sediment-specific macroinvertebrate metrics, particularly sediment 'decreaser' richness and abundance, again displayed the same pattern, with higher values in native forest streams, and lower values in horticultural and pastoral streams. However, 'Decreaser' abundances did not differ between land-uses with the inclusion of sediment as a statistically significant covariate. Conversely, the abundance and richness of 'increaser' taxa displayed a different trend, favouring horticultural sites, followed by pastoral and then native forest streams. Although this finding is expected as 'increaser' taxa are pollutant-tolerant, it is noteworthy that these two 'increaser' metrics were higher in horticultural rather than pastoral streams, despite the pastoral streams having higher DFS levels in comparison. It could suggest that stressors other than sedimentation underlie this pattern, although it should be stressed that this difference was not statistically significant.

Upon evaluating multiple metrics reflective of stream ecosystem health, particularly emphasizing the pollution tolerance spectrum of various macroinvertebrate taxa, the collective data consistently elucidates the superior health status of native forest streams, followed sequentially by horticultural and pastoral streams. The pre-eminence of native forest streams is notably substantiated by their markedly heightened taxa richness and abundance. Conversely, pastoral streams demonstrated the highest Simpson's Diversity Index, closely followed by native forest and horticultural streams. However, this heightened score within pastoral streams appears predominately influenced by a more equitably distributed relative abundance across taxa.

Nevertheless, these results still confirmed that the ecosystems of native forest streams are the healthiest, followed by horticultural and then pastoral streams. The following section will explore the reasons underlying these patterns related to land use.

3.3.3 Riparian vegetation

Firstly, the degradation of stream ecosystem health due to pastoral and horticultural land use primarily occurs through the alteration or removal of riparian zone vegetation (Allan, 2004; Burdon et al., 2013). Riparian vegetation serves multifaceted roles; it acts as a buffer against runoff carrying sediment and excessive nutrients from fertilizers (Décamps et al., 2009). Additionally, it upholds streambank stability, averts erosion, provides crucial shading, regulates water temperatures ideal for aquatic life, and offers essential allochthonous food sources through leaf litter and woody debris (Allan, 2004).

The removal of alteration of riparian vegetation significantly diminishes its capacity to deliver these vital ecosystem services, consequently leading to adverse impacts on aquatic life within the streams (Allan, 2004). The results detailing the overall Riparian Condition Index (RCI) provide evidence that suggests that impaired riparian habitat conditions underlie the pattern that has been identified (Harding et al., 2009). Specifically, it was found that pastoral and horticultural streams exhibited the poorest RCI scores alongside the highest levels of Deposited Fine Sediment (DFS) and nitrate. In contrast, native forest streams showcased the highest RCI scores and the lowest quantities of DFS and nitrate. These outcomes underscore the critical role of riparian habitats in effectively reducing the influx of sediment and nitrate into streams, emphasizing their pivotal importance in maintaining stream health and quality (Harding et al., 2009).

These findings align with national trends observed in a study conducted by Clapcott et al. (2020), which examined datasets sourced from regional councils. This study focused on the Rapid Habitat Index (RHI) measured across 901 sites in nine regions over a minimum of three years and a maximum of six. It also focused on deposited fine sediment, recorded across 336 sites in six regions over a minimum of two years and a maximum of five years. Consistently, it was found that native forest streams demonstrated higher RHI scores in contrast to streams located in pastoral land-use areas. In direct opposition, pastoral streams consistently exhibited heightened levels of DFS, whereas streams within native forest environments consistently maintained the lowest levels of sediment deposition (Clapcott et al., 2020).

Notably, these results were obtained using methodologies similar to those employed in our study, complementing our findings (Harding et al., 2009). For instance, the Rapid Habitat Index (RHI) essentially assesses similar attributes to the Riparian Condition Index (RCI). Moreover, the measurement of deposited fine sediment using the SAM2 method aligns with the approach undertaken in our study.

The study conducted by Clapcott et al. (2020) holds considerable significance owing to the high-quality nature of its data. Firstly, the data source was of high quality as it was obtained from nine regional councils across New Zealand, which utilized recent methodologies for stream assessment aligning with the National Environmental Monitoring Standards (NEMS) (Milne et al., 2022). Also, it is worth noting that there are 11 regional councils in New Zealand, meaning that this study was largely representative of all regional councils nationally (LGNZ, 2023). Moreover, the dataset exhibits robust statistical power, collected from as many as 901 sites spread across a maximum of nine regions over nine years (Clapcott et al., 2020).

This wealth of data significantly enhances the study's credibility, ensuring a comprehensive representation of national trends (Clapcott et al., 2020). The extensive coverage across regions and the utilization of contemporary assessment methodologies strengthens the likelihood that the findings accurately reflect genuine national patterns, devoid of issues stemming from poor data quality (Clapcott et al., 2020).

3.3.4 Nitrate

Median nitrate levels were highest in pastoral streams, followed by horticultural and then native forest streams. This result is typical of what has been found in previous studies. For example, the study by Young et al. (2005) found that median NO_3-N (Nitrate-nitrogen) concentrations were the highest in pastoral sites, followed by horticultural sites, with native forest sites having the lowest concentrations.

Additionally, the most recent analysis of national trends in river/stream water quality by (Whitehead et al., 2022) provides the best benchmark to compare our results against. This study analyzed 828 sites in New Zealand over 30 years (beginning in 1990 and ending in 2020) and found a strong correlation shared across nitrate and MCI scores with sediment. Additionally, it was found that nitrate and sediment levels were highest, and MCI scores were lowest in pastoral and horticultural streams/rivers. In comparison, nitrate and sediment levels were lowest, and MCI scores were highest in native forest streams (Whitehead et al., 2022).

However, the collation of different land cover classes associated with agricultural land use may hinder its applicability in this study (Whitehead et al., 2022). Specifically, this study incorporated ‘high-producing exotic grasslands’, short-rotation crops’, ‘orchards’, and ‘vineyards’ into an ‘agricultural’ land cover class (Whitehead et al., 2022). Although the results from this study are still valid, it is important to differentiate between pastoral and horticultural land use to correctly identify the specific origin and drivers of ecological degradation (Larned et al., 2019). For instance, this observation is highlighted in the report by Larned et al. (2018), which discusses how research over the last 20 years has focused primarily on contaminant loss and management associated with pastoral land use, with studies on horticultural land-use lacking.

The assumption that nutrient losses from horticultural land use are insignificant compared to pastoral land use stems from the vast difference in land areas occupied by these two types in New Zealand (Larned et al., 2018). In 2019, dairy farming covered approximately 2,221,459 hectares, while horticultural land use only spanned 132,717 hectares (StatsNZ, 2021). Despite horticulture occupying a smaller area, recent reports have indicated nitrate leaching rates exceeding 100 kg per hectare annually, similar to rates observed in pastoral agriculture (Larned et al., 2018). This underscores the need for greater concern regarding the impact of horticultural practices on stream ecosystems, particularly considering the increasing intensification and expansion of this land use (Larned et al., 2018).

While it is well-known that effluent containing urea from livestock is the primary source of nitrate contamination in streams in pastoral areas, relatively limited research has been conducted on the impact of horticultural land use (Larned et al., 2018). However, the improper application of fertilizers in horticultural areas has been identified as a leading cause of stream nitrate contamination (Larned et al., 2018). However, there remains a necessity for more focused research on fertilizer application methods, storage practices, and their implications for stream ecosystems in horticultural settings (Larned et al., 2018). Despite the differences in the primary sources of nitrate pollution between pastoral and horticultural land use, the loss or depletion of riparian vegetation is a common issue in both, significantly exacerbating nitrate leaching rates into streams (Allan & Johnson, 1997; Larned et al., 2018; Snelder et al., 2022).

This is likely due to the assumption that nutrient losses from horticultural land use are negligible compared to pastoral land use due to the large difference in the area that the two land uses occupy in New Zealand (Larned et al., 2018). Specifically, recent reporting has shown that in New Zealand, dairy farming occupies approximately 2,221,459 hectares (as of 2019), whereas in the same year, horticultural land use occupied 132,717 hectares (StatsNZ, 2021). Although horticulture occupies a much lower area, there has been evidence that annual nitrate leaching rates exceed 100 kg per hectare annually, similar to rates reported from pastoral agriculture (Larned et al., 2018). This indicates that there should be more concern for the effects of horticultural land use on stream ecosystems, especially with the rise in land-use intensification and expansion of horticultural land use (Larned et al., 2018).

While it is well-known that effluent containing urea from livestock is the primary source of nitrate contamination in streams in pastoral areas, relatively limited research has been conducted on the impact of horticultural land use (Larned et al., 2018). However, the improper application of fertilizers in horticultural areas has been identified as a leading cause of stream nitrate contamination (Larned et al., 2018). However, there remains a necessity for more focused research on fertilizer application methods, storage practices, and their implications for stream ecosystems in horticultural settings (Larned et al., 2018). Despite the differences in the primary sources of nitrate pollution between pastoral and horticultural land use, the loss or depletion of riparian vegetation is a common issue in both, significantly exacerbating nitrate leaching rates into streams (Allan & Johnson, 1997; Larned et al., 2018; Snelder et al., 2022).

It is crucial that we have a comprehensive understanding of the pathways in which nitrate enters streams as a result of poorly managed anthropogenic land-use practices (Allan, 2004; Larned et al., 2018). This understanding is critical in preventing the detrimental impacts linked to excessive quantities of nitrate in waterways (Allan, 2004; Larned et al., 2018; Snelder et al., 2022; Snelder et al., 2020). Such impacts include eutrophication, which can lead to the proliferation of periphyton, which rapidly depletes dissolved oxygen levels and smothers streambed habitats (Francoeur & Biggs, 2006). However, not all stream types are susceptible to periphyton proliferation such as those subject to frequent and regular flood events, high levels of shade or comprised of fine and highly mobile substrate (Francoeur & Biggs, 2006) Also, when freshwater organisms absorb nitrate, it can be reduced to nitrite via gut bacteria, leading to toxic effects (Singh et al., 2022). Specifically, nitrite converts haemoglobin to methaemoglobin, which cannot bind oxygen and subsequently results in hypoxia (Singh et al., 2022).

The cumulative impact of these factors can lead to a alteration in the community structure of stream ecosystems, resulting in the loss of more pollutant-sensitive species, as noted in the study by Gutiérrez-Rial et al. (2023). This study observed this effect as MCI, QMCI & ASPM scores, and EPT richness and abundance were notably lower in pastoral and horticultural streams than in native forest streams. The likelihood that nitrate concentrations influenced these findings is relatively high, especially given that pastoral and horticultural streams showed significantly higher nitrate levels than native forest streams. Additionally, it is important to note that the Macroinvertebrate Community Index (MCI) and its quantitative variant, QMCI, were originally designed to assess the presence of organic pollution within stream ecosystems. This includes evaluating the impacts of nutrient enrichment within stream ecosystems, indicating that these indices can capture the effects of nitrate pollution (Stark & Maxted, 2007).

3.3.5 Pesticides

Pesticide usage represents a significant threat to the health of macroinvertebrates and other aquatic organisms in stream ecosystems (Bighiu et al., 2020; Knillmann & Liess, 2019b; Larned et al., 2018). Although not included in this study due to logistical constraints, it is possible that pesticides may account for some of the intriguing trends observed in this thesis. However, no data regarding pesticides were collected, with the following discussion points being purely speculative.

Specifically, further analysis of MCI and QMCI scores, along with EPT, 'Decreaser' and overall taxa richness, revealed significant differences solely between native forest and horticultural streams. This finding is noteworthy as it indicates underlying causes unique to horticultural land use that have influenced these findings.

This interpretation is supported by the fact that despite significant differences in other environmental indicators such as sediment, nitrate, and riparian condition across land-use types, there were no significant differences regarding macroinvertebrate metrics between native forest and pastoral streams or between pastoral and horticultural streams. Simply, it is reasonable to expect that when there are statistically significant differences in environmental conditions among the three land-use types sampled, it should result in significant differences in macroinvertebrate metrics across all land uses.

Given that other environmental factors such as sediment, nitrate, and riparian conditions did not account for the significant differences observed between native forest and horticultural streams, it is possible that pesticide usage might be the underlying cause. This suspicion stems from the extensive use of pesticides in horticultural land use and their known severe impacts on freshwater macroinvertebrates (Hageman et al., 2019; Knillmann & Liess, 2019b; Manktelow et al., 2005). In New Zealand, horticultural activities have been identified as the dominant user of pesticides, applying an average of 13 kg per hectare, significantly higher than the average of 3 kg per hectare applied in pastoral land use (Manktelow et al., 2005; Ministry for the Environment, 2002).

Fortunately, though, due to the implementation of updated Integrated Pest Management Plans, the most recent nationwide analysis has shown that the percentage of pesticides detected in groundwater wells has decreased from 24% in 2018 to 9% as of 2022 (Total of 184 wells tested across New Zealand); (Close & Banasiak, 2023). However, this survey did not detect pesticides in any of the 10 wells tested in the Bay of Plenty (Close & Banasiak, 2023). While this discovery may challenge my suspicions regarding pesticide usage as the cause behind notable differences in macroinvertebrate metrics between native forest and horticultural streams, previous studies have reported findings that suggest my suspicions might be still valid.

For instance, this nationwide survey primarily focused on quantifying pesticide concentrations in groundwater utilized for human consumption and did not account for ecological implications (Close & Banasiak, 2023). It is essential to distinguish between pesticide thresholds for humans and freshwater aquatic organisms, particularly macroinvertebrates. This is because the concentration in which mortality occurs in freshwater macroinvertebrates is significantly lower than the threshold detection limits used in groundwater analysis (Close & Banasiak, 2023; Werner & Moran, 2008).

The best example of this is the mass mortality incident of kōura (*Paranephrops planifrons*) that occurred in the Oraka stream in the Waikato region of New Zealand (Kirkeby, 2018; Ling, 2019). This case involved the application of the cypermethrin-containing insecticide AlphaScud to pine forests adjacent to the Okara stream, resulting in the death of hundreds of koura individuals. It is crucial to emphasize that initial examinations of water and sediment samples showed no presence of pesticides. However, upon further inspection of kōura tissue samples, cypermethrin was present below accepted detection threshold concentrations (0.01 mg/kg). Subsequently, it was strongly suggested that the spray drift or leaching of AlphaScud into the Oraka stream was responsible for the mass mortality of kōura (Ling, 2019).

Cypermethrin is a pyrethroid-based pesticide widely used in New Zealand, especially in the horticultural industry and is known to be highly toxic to freshwater macroinvertebrates. For example, in the study by Werner & Moran (2008), it was observed that acute toxicity to freshwater macroinvertebrates occurred at the incredibly low concentration of 1 pg/L (Werner & Moran, 2008). Furthermore, in the study by Werner & Moran (2008), significant decreases in the abundance and diversity of macroinvertebrates were observed up to several months after the direct aerial application of pyrethroid-based insecticides to streams, ranging from 0.28 to 140 g/ha (Werner & Moran, 2008). Similar results were also observed in the study by Antwi & Reddy (2015). In comparison, the concentration of cypermethrin applied to the forests adjacent to the Oraka stream ranged between 5 to 25 g per hectare (Ling, 2019).

It is worth mentioning that pyrethroid-based insecticides are frequently mixed with the synergist PBO (Piperonyl butoxide) to increase their efficiency in killing targeted pest insects. This is because PBO deactivates enzymes that aid in metabolizing pyrethroids within organisms. Thus, mixing these two chemicals dramatically increases pyrethroid insecticide toxicity to freshwater organisms, making detecting them significantly more challenging.

Although in the Oraka stream case, spray drift appeared to be the primary pathway in which pyrethroid-based pesticides entered the stream, resulting in the mass mortality of koura individuals, there is another pathway in which it can enter, which is sediment (Hladik & Kuivila, 2009; Liu et al., 2004).

It has been established in previous studies that pyrethroid-based pesticides have extremely hydrophobic properties, meaning that they are significantly more likely to be bound to sediment through sorption (Hageman et al., 2019; Liu et al., 2004). For example, in the study by Laskowski (2002), the log octanol-water partition coefficient (K_{ow}) of various pyrethroid-based pesticides ranged between 4.5 to 7 K_{ow}. For context, chemicals with a higher/positive K_{ow} value are hydrophobic (repelled by water), while those with a lower/negative K_{ow} value tend to be hydrophilic (attracted to water) (Moldoveanu & David, 2021). This is a metric of whether a chemical prefers to be attached to water or solids (like the fatty alcohol octanol).

The sorption of pyrethroid-based pesticides is determined by 6 factors. The first and most important factor is the physicochemical properties of the pyrethroid, such as its lipophilicity (represented by its K_{ow} value) and molecular structure (Laskowski, 2002; Lu et al., 2019). This factor is the most important because it strongly determines whether pyrethroid pesticides will be bound to sediment or freely available, allowing them to bind to aquatic organisms such as macroinvertebrates (Laskowski, 2002; Lu et al., 2019). The second factor is the size and surface area of sediment particles, as the sorption of pyrethroids decreases as the sediment size increases due to a subsequent decrease in surface area (Lu et al., 2019). The fourth factor is the quantity and quality of dissolved organic matter, as pyrethroids can bind to them, reducing the concentration of pyrethroids in a stream's water column. The fifth factor is the time in which pyrethroids and sediment are in contact, with sorption increasing as the length of time increases. The sixth factor is the salinity of the water, with the sorption of pyrethroids increasing as salinity increases (Lu et al., 2019).

These factors, particularly the physicochemical properties of pyrethroid pesticides, are very important as they determine their bioavailability (Lu et al., 2019). they determine if these toxins will remain bound to sediment upon entry or become released into the water column and attach to aquatic organisms, resulting in adverse effects, including mortality (Lu et al., 2019).

In summary, these studies have confirmed that the concentrations of pesticides (especially pyrethroid-based pesticides) leading to adverse effects and or mortality in macroinvertebrates are extremely low and challenging to identify. They strongly suggest the potential for pesticide application to be an underlying cause of the significant differences in macroinvertebrate metrics observed between native forest and horticultural streams in this study. However, the effects of pesticides on New Zealand stream ecosystems is an area of research that needs to be investigated.

This need has been highlighted by the report by Larned et al. (2018), along with the need for further research into the overall effects of horticultural land use on stream ecosystems and the macroinvertebrates inhabiting them.

3.3.6 Stream flow velocities

I found that median flow velocities were highest in horticultural streams, followed by pastoral and then native forest streams. However, this result was not statistically significant. Furthermore, the wide variability streams exhibit across all three land uses poses a challenge in determining if the variations in flow velocities can be solely attributed to anthropogenic land-use practices. Additionally, stream velocities are highly variable in space and time, meaning that the singular measurement taken in this study may not accurately represent the hydro-dynamics of these environments.

While it was noted that horticultural land use exhibited the highest median stream flow velocities, this particular finding might have been influenced more by the elevation and stream order of the horticultural sites sampled (Comiti et al., 2007; Flint, 1974).

For instance, horticultural streams had the lowest average elevation of 19 meters ASL (Above Sea Level) and were of a higher stream order. These geographical characteristics appeared to have a more substantial role in causing the observed higher flow velocities than the specific practices related to horticultural land use. Several factors contribute to this phenomenon (Comiti et al., 2007; Flint, 1974).

Firstly, higher-altitude and lower-order streams typically exhibit faster flow velocities due to a significant increase in gravitational kinetic energy (Comiti et al., 2007; Flint, 1974). Higher-order streams accumulate water from numerous lower-order tributaries with steeper gradients, augmenting their flow's magnitude and volume. This increased flow leads to accelerated erosion rates caused by larger volumes of water discharge, resulting in channels widening (Comiti et al., 2007; Flint, 1974).

The widening of these channels creates a synergistic effect as it allows for greater water volume accommodation, further amplifying the flow's magnitude (Comiti et al., 2007; Flint, 1974). Moreover, water velocity is notably higher in streams at higher elevations and lower-order categories, reducing friction. This reduction in friction is due to fewer obstructions from natural features like rocks and vegetation. The kinetic energy in these streams is sufficient to manoeuvre through or around obstacles or even displace them (Comiti et al., 2007; Flint, 1974).

It is also likely that the elevation and stream order can explain the flow velocities exhibited by native and pastoral streams (Comiti et al., 2007; Flint, 1974). For instance, pastoral streams had the second-highest median flow velocities, which makes sense given that they had the second-highest average elevation of 86 meters above sea level. In comparison, native forest streams had the lowest median flow velocities but the highest average elevations of 196 meters ASL.

While this study cannot determine the degree to which anthropogenic land-use practices have influenced stream flow velocities, numerous other studies have consistently established adverse changes associated with human-induced alterations in land-use. For example, agricultural land use has resulted in adverse changes in the natural flow regime of streams through the construction of weirs and levees, channelization and water abstraction (Allan, 2004; Bunn & Arthington, 2002). Specifically, weirs have been linked to hydrological changes such as a reduction in the magnitude and frequency of high flows, resulting in geomorphic responses such as increased deposition of fine sediment, channel narrowing and reduced formation of point bars, oxbows, secondary channels and changes in channel planforms (Chien, 1985; Copp, 1989; Johnson, 1994; Poff et al., 1997; Sear, 1995; Stevens et al., 1995; Williams & Wolman, 1984).

Channelization and levees have been linked to reduced water infiltration into the soil and reduced overbank flow, resulting in geomorphic responses such as reduced baseflows and channel restriction causing downcutting (erosion of streambed) (Kondolf & Curry, 1986; Leopold, 1968; Poff et al., 1997; Prestegaard, 1988). Also, water abstraction has been linked to lowered water table levels, resulting in a loss of riparian vegetation due to insufficient water supply, leading to streambank erosion and channel downcutting.

Although the degree to which the flow velocities of the streams sampled in this study have been influenced by anthropogenic land use is uncertain, it is still worthwhile to discuss the flow regime's role in shaping stream ecosystems and macroinvertebrate communities. This is because a stream flow regime has been labelled as a 'master' variable that profoundly affects water quality, ecosystem functioning, physical habitat and biodiversity, including biotic interactions that contribute to the ecological integrity of streams and rivers {Sofi, 2020 #466}. The significant influence that a stream flow regime has is best summarized in the study (Bunn & Arthington, 2002) in the form of four key principles.

The first one is that the magnitude, frequency, timing, and rate of change of a stream's flow regime are significant determinants of stream habitat availability (Bunn & Arthington, 2002; Szałkiewicz et al., 2022). Specifically, the characteristics of a stream flow regime interact with the geological features of a stream, such as a substrate, and thus determine stream geomorphology, including the formation of microhabitats such as riffles and pools (Bunn & Arthington, 2002; Szałkiewicz et al., 2022).

Also, the characteristics of a stream's flow regime influence the temperature of the water and its oxygen levels (Bunn & Arthington, 2002; Szałkiewicz et al., 2022). For example, slower-flowing streams are prone to having higher temperatures and lower oxygen levels in comparison to faster-flowing streams due to a lack of dispersion of energy in the form of solar radiation and oxygen, respectively (Bunn & Arthington, 2002; Szałkiewicz et al., 2022). This study shows this pattern as native forest streams had the lowest median flow velocities, dissolved oxygen levels (mg/L), and the highest temperatures. However, it is important to note that these differences in water quality parameters between sites were not statistically significant, and no sites displayed values that would be indicative of morality or severe adverse effects for macroinvertebrates. For instance, NIWA guidelines show that macroinvertebrates cannot inhabit water warmer than 22° Celsius, where the maximum water temperature was only 15.7° Celsius (Table 2).

However, the dissolved oxygen threshold values for New Zealand freshwater macroinvertebrates are unclear. For instance, the American study (Davis, 1975) reported mortality in macroinvertebrate taxa at dissolved oxygen concentrations between <1 mg/L and 8 mg/L.

Although the New Zealand Macroinvertebrate Community Index (MCI) is based on the differing sensitives of macroinvertebrate taxa to organic pollution, which in turn strongly affects dissolved oxygen levels (Stark & Maxted, 2007), there have been few studies focusing specifically on dissolved oxygen threshold values for New Zealand macroinvertebrates (Graham & Franklin, 2017).

Two studies tested the oxygen tolerance of *Paratya curvirostris* (Freshwater shrimp) and *Paranephrops planifrons* (Freshwater crayfish). The first study by Dean & Richardson (1999) reported 26.7% mean mortality rates for *Paratya curvirostris* after 48 hours at 1 mg/L dissolved oxygen (DO). In the study by Landman et al. (2005), mortality rates of 50% were reported for *Paratya curvirostris* and *Paranephrops planifrons* after 48 hours at 0.82 mg/L and 0.77 mg/L, DO, respectively.

Additionally, the Australian study by Connolly et al. (2004) reported macroinvertebrate mortality when dissolved oxygen levels became lower than 10% saturation, with sub-lethal effects exhibited by macroinvertebrates becoming more prominent the closer oxygen saturation levels became to 10%. Specifically, it was observed that chironomid midges and mayflies exhibited drift effects and reduced emergence. Fortunately, this highlighted that macroinvertebrate species will likely migrate away from low-oxygen environments before morality occurs. The authors also highlighted the importance of temperature, as the metabolic rate of macroinvertebrates increases as temperature increases. Additionally, the solubility of water decreases as its temperature increases, meaning it can hold less dissolved oxygen. Essentially, this means that the oxygen requirements of macroinvertebrates increase and oxygen saturation decreases as water temperatures increase, greatly exacerbating hypoxia within macroinvertebrates (Connolly et al., 2004).

Additionally, the characteristics of a stream's flow regime influence the distribution and availability of food, which can affect macroinvertebrate communities in different ways (Bunn & Arthington, 2002). For example, faster-flowing streams can provide a continuous supply of organic matter and detritus (Course Particulate Organic Matter, CPOM), which can favour macroinvertebrates that prefer this food type, such as shredders (Szałkiewicz et al., 2022). Conversely, slower-flowing streams can accumulate algal matter in the form of Fine Particulate Organic Matter (FPOM), potentially favouring macroinvertebrate species belonging to the 'grazer' functional group (Szałkiewicz et al., 2022). However, anthropogenic land use has been found to affect this food distribution significantly. For instance, the two European studies by Graeber et al. (2013) and (Pardo & García, 2016) found that in response to reduced water flows, the abundance of filter-feeding macroinvertebrate species decreased, linked to a reduction in the availability of FPOM.

The second principle is that maintaining natural patterns of longitudinal and lateral connectivity is essential to the viability of populations of many riverine species (Bunn & Arthington, 2002). This principle highlights the importance of maintaining the natural flow regime in order to ensure that macroinvertebrate metacommunities can disperse between different reaches or streams to maintain their population (Li et al., 2021; Szałkiewicz et al., 2022).

Disruptions to the natural flow regime can cause the dispersal ability of macroinvertebrates to be exceeded, preventing their ability to disperse across streams and maintain their population (Fahrig, 1992; Li et al., 2021). For instance, the study by Irvine & Henriques (1984) highlighted the susceptibility of macroinvertebrate species belonging to the order Trichoptera (cased-caddisflies specifically), *Oligochaeta* and *Chironomidae* to large fluctuations in the magnitude and frequency of water flow. Specifically, it was found that the drift density for these macroinvertebrates remained low during periods of stable flow but increased dramatically even with small increases in the magnitude of stream flows (Irvine & Henriques, 1984).

The third principle is that alterations to the natural flow regime can facilitate the invasion/succession of exotic and alien species. For example, the study by Mathers et al. (2020) investigated the effects that changes in the flow regimes of streams had in facilitating the establishment of the invasive crayfish species *Pacifastacus leniusculus*, also called the signal crayfish. It was found that during years of low-flow conditions, the success rate of *Pacifastacus lenisculus* establishment rose to 80%. Conversely, the success rate of this species establishment was 1.6% under high flow conditions. The results of this study were attributed to the fact that juvenile individuals are more prone to being displaced under high-flow conditions but proliferate under low-flow conditions.

The fourth principle is that aquatic species have evolved life history strategies in direct response to the natural flow regime (Szałkiewicz et al., 2022). However, although this principle is not necessarily applicable in New Zealand stream macroinvertebrate communities, it provides important and interesting insights.

It has been identified in the review by Winterbourn et al. (1981) that New Zealand macroinvertebrate species have adapted flexible life history patterns in direct response to highly unpredictable environmental conditions, such as a stream's flow velocity. For instance, it was found that the composition of macroinvertebrate communities and functional feeding groups exhibited little difference between upstream and downstream locations. Additionally, the composition of macroinvertebrate functional feeding groups and community composition did not display a distinct seasonal change (Winterbourn et al., 1981).

Further evidence of this phenomenon is provided in the study by Dewson & Death (2007), which found that macroinvertebrate community composition, density, abundance and percentage of EPT taxa were similar upstream and downstream of water abstraction sites despite decreases in stream flow velocity, depth and wetted width. Additionally, the study (Jowett & Richardson, 1990) identified that the mayfly *Deleatidium* did not have an upper limit of flow velocity tolerance. This study also noted that New Zealand macroinvertebrates can tolerate a broad range of flow velocities by seeking refuge in coarse substrate (Jowett & Richardson, 1990).

This observation could provide an additional explanation as to why it is unlikely that stream flow velocities would have had a significant influence on the macroinvertebrate indices reported in this study. For example, cobble was the most prevalent substrate type across all three land-use types (32.3%), which supports the idea that the macroinvertebrate communities in the sampled streams had an abundance of refugia to seek shelter from rises in flow velocities (Jowett & Richardson, 1990).

Although the degree to which anthropogenic land use has altered the flow velocities of the streams sampled is uncertain due to a combination of the large variability between sites and statistical insignificance, it is possible that anthropogenic land-use may have influenced stream flow velocities, and in turn the results of the macroinvertebrate indices reported in this study. For instance, horticultural streams had the highest flow velocities but the lowest median MCI and QMCI scores, the lowest taxa richness and Simpson's Diversity Index scores. In comparison, native forest streams had the lowest median flow velocities and highest median MCI, QMCI, and ASPM scores.

However, it is also likely that the elevation and order of streams have influenced these results, as native streams had the highest mean elevations, whereas horticultural streams had the lowest. Although stream flow velocities have likely affected these results, it should be noted that the diversity and availability of habitat and food tend to decrease from the headwaters of a stream to its lower reaches. This observation originates from the well-known study by Vannote et al. (1980) and is a core aspect of the River Continuum Concept (RCC). The RCC highlights that headwater streams are characterized by narrow streams with abundant dense riparian vegetation, which provides an ideal abundance and diversity of habitats and food sources. In comparison, lower reaches are characterized by wide streams that are poorly shaded, with a significant influx of FPOM and suspended solids resulting in turbid water (Vannote et al., 1980).

4 Chapter 4: Land use impacts on invertebrate traits and functional diversity

4.1 Introduction

In comparison to the traditional taxonomic metrics used under the National Policy Statement for Freshwater Management 2020, metrics that utilise functional diversity have been shown to provide a deeper level of insight into ecological problems faced by New Zealand streams (Clapcott et al., 2017; Dolédec et al., 2006; Dolédec et al., 2011; Townsend et al., 2008). Changes made to the environmental conditions of a macroinvertebrate habitat by anthropogenic activities such as land-use intensification can have disproportionate effects on macroinvertebrate taxa (Allan, 2004; Dolédec et al., 2011; Dolédec et al., 1999).

These disproportionate effects can be better understood through the concept of functional traits. A functional trait is defined as 'morpho-physio-phenological' traits which impact fitness indirectly via their effects on growth, reproduction and survival, the three components of individual performance' (Violle et al., 2007). Each macroinvertebrate species has its own set of functional traits that have arisen through natural selection that allow them to adapt to dealing with different ecological problems, such as the environmental conditions of their habitat (Laureto et al., 2015; Usseglio-Polatera et al., 2000a; Violle et al., 2007). These functional traits comprise functional diversity, defined as the variety and distribution of functional traits within a biological community (Coccia et al., 2021; Laureto et al., 2015; McGill et al., 2006).

Functional traits are commonly separated into two groups: biological traits and ecological traits. Biological traits affect an organism's 'life cycle, physiological and behavioural characteristics such as maximum body size, lifespan, feeding and reproductive strategies and mobility' (Usseglio-Polatera et al., 2000a). Ecological traits are 'related to habitat preferences, like pH and temperature tolerances, tolerance to organic pollution and biogeographic distribution' (Usseglio-Polatera et al., 2000a). In the context of biomonitoring, a macroinvertebrate taxon's sensitivity to pollution/disturbance depends on their traits. Simply, macroinvertebrates that are more tolerant of pollution have traits that allow them to perform better in comparison to highly sensitive taxa, which do not have traits compatible with living in such an environment (Clapcott et al., 2017; Judi Hewitt et al., 2014; Usseglio-Polatera et al., 2000a).

It is important to note that although traits are commonly split into two categories, separating them can be difficult as they are frequently interlinked (Menezes et al., 2010; Poff et al., 2006; Storey & Quinn, 2011). This interlinkage is caused by trait syndromes, which occur when an organism's biological traits (particularly life history) have been chosen through natural selection, helping it to adapt to different ecological problems. In other words, not all traits an organism possesses are in direct response to a specific stressor, as some are simply an evolutionary response to its natural environment. This means a species' ecological traits have been shaped by its biological traits and are not necessarily wholly separate entities, making ecological traits poor predictors of an individual's response to stressors. Also, some traits may not reflect an organism's preferred habitat preferences due to a concept called trait trade-offs. As organisms have limited resources, traits that guarantee the highest survival rate are selected through natural selection, meaning that biological traits can be favoured over ecological traits (Menezes et al., 2010; Poff et al., 2006; Storey & Quinn, 2011).

It is crucial to discuss trait tradeoffs and syndromes as they have implications that need to be considered when using macroinvertebrates for biomonitoring (Menezes et al., 2010; Poff et al., 2006). The first implication is that a species' traits may have been selected to solve an ecological problem unrelated to its habitat preferences. Also, pollution/disturbance can have a more significant effect on certain species. Although macroinvertebrate species can have the same or similar ecological traits/habitat preferences, they can have different biological traits due to having different adaptations for living in the same environment. This means that pollution/disturbance can significantly affect some biological traits more than others and, in turn, the species that carry them. This means that although ecological traits can be a good predictor of how a macroinvertebrate community will respond to environmental changes caused by pollution/disturbance, certain species will differ in their response based on their biological traits (Clapcott et al., 2017; Menezes et al., 2010; Poff et al., 2006).

Due to trait syndromes, trait trade-offs and confoundment, it can be helpful to employ the use of the response trait paradigm when assessing the effects of pollution/disturbance on macroinvertebrate communities (Menezes et al., 2010; Poff et al., 2006; Sterk et al., 2013). Response traits are essentially any traits associated with the response of an organism to its environment, including instances of pollution/disturbance (Delerue & Michalet, 2022; Schmera et al., 2022; Sterk et al., 2013). Effect traits describe how an organism affects ecosystem functioning and can also be useful in assessing the effects of pollution/disturbance. In instances where pollution/disturbance is significant enough to cause mortality or severe physiological stress, the functions that each macroinvertebrate species performs can be severely hindered by either reducing the abundance of each species, fulfilling that functional role or hindering their efficiency. In turn, this can negatively affect the functional diversity of a stream ecosystem (Delerue & Michalet, 2022; Schmera et al., 2022; Sterk et al., 2013).

4.1.1 Metrics of functional diversity

In this chapter, I tested five different metrics that describe different components of functional diversity. Functional diversity is defined as ‘the components of biodiversity that influence how an ecosystem operates’ (Schmera et al., 2017; Tilman et al., 1997).

Functional richness is defined as ‘the amount of functional trait space occupied by the species’ (Mason et al., 2005; Schmera et al., 2017). Functional richness essentially measures the overall distribution of traits in an ecological community (Villéger et al., 2008). As functional richness increases, so do the niches filled by different species (Ding et al., 2016).

Functional evenness is defined as ‘the distribution of abundance among functional characteristics’ (Mason et al., 2005; Schmera et al., 2017). Functional evenness essentially measures how equal the distribution of functional traits across an ecological community is (Ding et al., 2016). High functional evenness implies that all niches are adequately utilized due to an equal distribution of functional traits across an ecological community. Conversely, low functional evenness implies an unequal distribution of functional traits, meaning that some niches will be underutilized due to organisms within the ecological community lacking functional traits to fill that niche (Ding et al., 2016).

Functional divergence is defined as ‘the degree to which the abundance distribution maximizes differences in functional characters within the community’ (Mason et al., 2005; Schmera et al., 2017). Functional divergence essentially measures the divergence (dissimilarity) in the distribution of functional traits in an ecological community's trait space/niches (Ding et al., 2016). It implies that an ecological community with high functional divergence would display a high degree of niche differentiation and efficient resource use, leading to low resource competition (Ding et al., 2016).

Rao’s quadratic entropy (Rao’s Q hereafter) is defined as the pairwise distance of the functional traits of organisms, weighted by their relative abundance (Ding et al., 2016; Rao, 1982; Schmera et al., 2017). This metric essentially combines functional richness and divergence while accounting for the relative abundance of different species. It has been compared to the Simpson's Diversity Index but instead for functional diversity (Ding et al., 2016; Schmera et al., 2017).

Functional dispersion is defined as the 'distribution of traits in trait space while accounting for their relative abundance' (Ding et al., 2016). Essentially, functional dispersion measures the spread or variability of functional traits within the trait space of an ecological community while accounting for the relative abundances of these traits. For example, an ecological community with high functional dispersion will have an assemblage of organisms that display a wide variety of traits, while those with low functional dispersion will display the opposite (Ding et al., 2016). For example, a macroinvertebrate community with high functional dispersion can display a broad range of feeding strategies, while those with low functional dispersion will have a narrow range.

4.1.2 Objectives of this chapter

In this chapter, I aimed to investigate the impact of human land uses and sedimentation have on stream macroinvertebrate traits and functional diversity. I hypothesized that non-random species losses would lead to negative impacts of deposited fine sediment (DFS) and land-use on functional diversity (Jonsson et al., 2017). Underpinning these patterns, I hypothesised that certain macroinvertebrate trait modalities would respond to sediment stress and land use, respectively. I predicted that lift history traits (number of reproductive cycles per year <1; 'semivoltine'), morphology (gill respiration of aquatic stages; 'gill'), locomotion (crawling of aquatic stages; 'crawler'), and resource acquisition (feeding habits such as grazing; 'grazer', 'filter-feeding') would all be negatively impacted by DFS (Dolédec et al., 2011; Judi Hewitt et al., 2014; Wagenhoff et al., 2012). I also predicted that the following modalities would have negative responses to sediment: 'submerged', 'univoltine', 'size1', 'surface', and 'eggprotected' (Clapcott et al., 2017; Dolédec et al., 2011; Philips & Smith, 2012; Phillips & Reid, 2012b). Regarding land-use change (native forest contrasted to pasture, and native forest contrasted to horticulture), I predicted that the following modalities would respond: 'pluriv', 'single', 'submerged', 'scraper', 'algalp', 'eggprotected', 'plastron', 'lda2', 'lda3', 'surface' and 'lowflex' (Clapcott et al., 2017; Dolédec et al., 2011; Philips & Smith, 2012; Phillips & Reid, 2012b).

4.1.3 Functional diversity indicators

Functional richness of macroinvertebrate communities was highest in pastoral streams, albeit with the greatest variance, followed by native forest streams (Figs. 17a, 18a). Functional richness was lowest in the horticultural streams. These differences were not statistically significant at α=0.05 (*P*=0.11) when deposited fine sediment (DFS) was included as covariate (*P*=0.60; Table 18).

Functional evenness was highest in horticultural streams, followed closely by pastoral and then native forest streams, but values were relatively similar across all land-use types (Figs. 17b, 18b). These differences were not statistically significant (*P*=0.87) when DFS was included as covariate (*P*=0.48; Table 18).

Functional divergence was highest in pastoral streams, followed by native forest and then horticultural streams, but values were again relatively similar across all land-use types (Figs. 17c, 18c). These differences were not statistically significant (*P*=0.84) when DFS was included as covariate (*P*=0.10; Table 18).

Functional dispersion was highest in pastoral streams, followed by horticultural streams which was relatively similar, and lowest in native forest streams (Figs. 17d, 18d). These differences were not statistically significant (*P*=0.83) when DFS was included as covariate (*P*=0.30; Table 18).

Rao's Q, which is closely related to functional dispersion, was relatively similar across all three land use types (native forest, pastoral & horticultural); (Figs. 17e, 18e). These differences were not statistically significant (*P*>0.99) when DFS was included as covariate (*P*=0.60; Table 18).

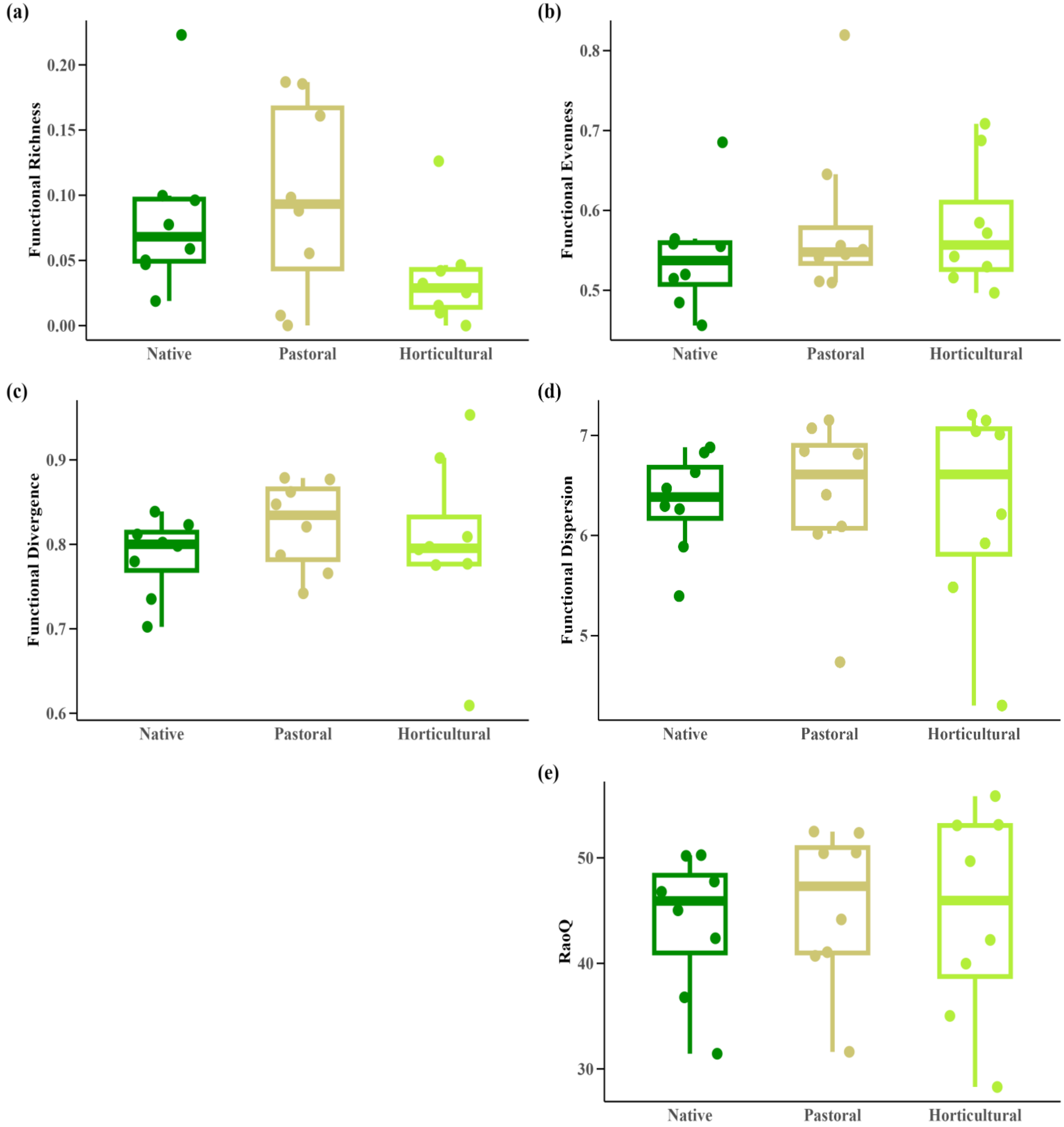

Figure 17. *Boxplots of the changes in functional diversity metrics across streams in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) Functional richness, b) Functional evenness, c) Functional Divergence, d) functional dispersion, e) RaoQ.*

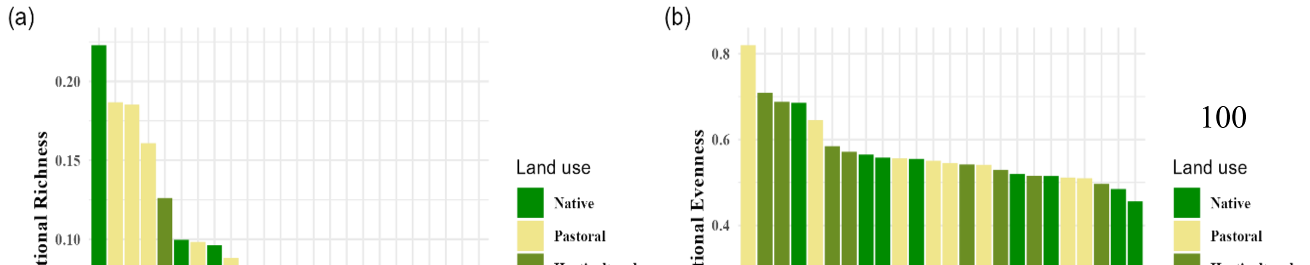

Figure 18. *Barchart of the changes in functional diversity metrics ranked from highest to lowest across streams in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) Functional richness, b) Functional evenness, c) Functional Divergence, d) functional dispersion & e) Rao's Quadratic Entropy.*

Table 18. *ANOVA results from mixed-models testing changes in metrics of macroinvertebrates' functional diversity across streams in three local land uses (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Predictor**	**F-value**	***df***	***df*.res**	***P*-value**

Functional Richness	Site type	2.50	2	16.4	0.11
	Sediment	0.29	1	19.6	0.60
Functional Evenness	Site type	0.15	2	17.4	0.87
	Sediment	0.51	1	17.5	0.48
Functional Dispersion	Site type	0.18	2	17.5	0.84
	Sediment	3.12	1	17.2	0.10
Function Divergence	Site type	0.19	2	17.5	0.83
	Sediment	1.14	1	17.2	0.30
Rao's Q	Site type	0.01	2	18.0	1.00
	Sediment	0.25	1	15.6	0.60

The correlation of each macroinvertebrate functional diversity metric (functional richness, evenness, divergence & dispersion) with DFS was also tested using linear regression (Figure 19). The results of these five analyses revealed no significant relationships with DFS (Figure 19), with the exception of functional divergence which increased with deposited fine sediment cover (Figure 19c; Appendix 2, Table 21).

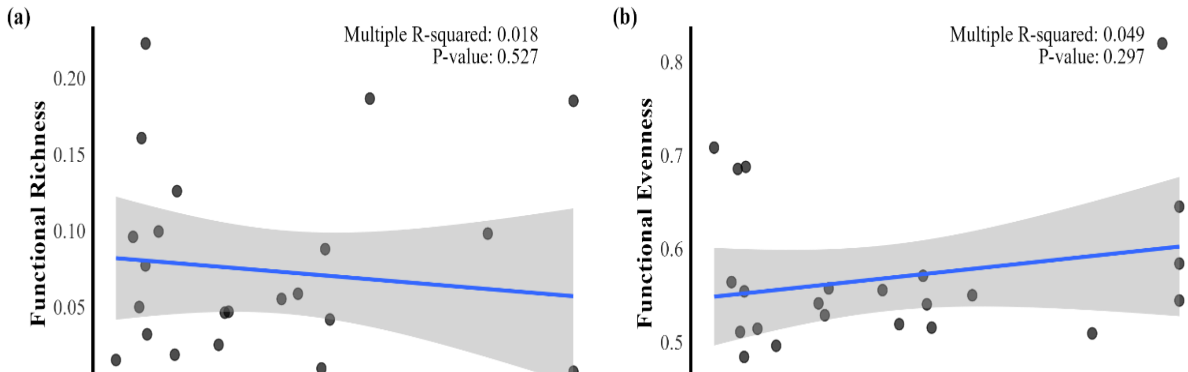

Figure 19. *Linear regression plots displaying the relationship between Deposited Fine Sediment (DFS) & a) Functional richness, b) Functional evenness, c) Functional divergence, d) Functional dispersion & e) Rao's Q.*

4.1.4 Macroinvertebrate trait abundances in relation to land use and sediment

4.1.4.1 Overview

This section aimed to assess the influence that land-use type (Native Forest, Pastoral, Horticultural) and Deposited Fine Sediment (DFS) have on the Community Weighted Mean (CWM) abundances of 58 macroinvertebrate trait modalities (Phillips & Smith, 2018). These 58 trait modalities belong to 16 biological traits and four trait categories (Morphology, Life history, Mobility, and Resource Acquisition) (Phillips & Smith, 2018).

Traits related to the 'Morphology' are 'maximum potential size', 'body flexibility', 'body form' and 'respiration of aquatic stages (not including eggs)'; traits related to the 'Life history' include 'maximum number of descendants per reproductive cycle', 'maximum number of reproductive cycles per year', 'number of reproductive cycles per individual', 'life duration of adults', 'reproductive techniques', 'oviposition sites', 'egg/egg mass' and 'aquatic stages (Phillips & Smith, 2018). Traits related to 'Mobility' include 'dissemination potential (all stages) and 'attachment to substrate of aquatic stages (excluding eggs)'. Traits related to the 'Resource Acquisition ' are 'feeding habits' and 'dietary preferences'. The results of the 58 modalities comprising these 16 biological traits are detailed below. The CWM was derived from the macroinvertebrate communities of the 24 streams sampled.

Due to the large number of trait modalities, a summary of the statistically significant results (α=0.05) is provided below. Results of mixed-model ANOVAs revealed that differences due to local land-use types in the CWM abundances of the following trait modalities were not significant with the inclusion of sediment as a signifcant covariate: 'highflex' ($P<0.01$; Table 21), 'univ' ($P<0.05$; Table 29), 'pluriv' ($P<0.05$; Table 28), 'surface' ($P<0.05$; Table 36), 'shredder' ($P<0.05$); Table 46), 'specialist' ($P<0.001$; Table 47), 'moderatespe, ($P<0.01$; Table 47).

The CWM abundances of the following trait modalities were only statistically significant across land-uses when sediment was included as a covariate: 'noflex' ($P<0.05$; Table 21), 'flattened' ($P<0.05$; Table 23), 'desc1' ($P<0.01$; Table 26), 'desc4' ($P<0.01$; Table 26), ($P<0.05$), 'cpi1' ($P<0.01$; Table 30), 'lda2' ($P<0.05$; Table 32), 'single' ($P<0.05$; Table 34), 'two' ($P<0.05$; Table 35), 'eggprotected' ($P<0.05$; Table 38).

Additionally, the modality 'submerged' was statistically significant different across land uses ($P<0.05$; Table 36) even with the inclusion of sediment as a significant covariate ($P<0.05$; Table 36).

Post-hoc analyses revealed significant differences in the CWM abundances across land uses for the following modalities. The following three modalities were significant only between native forest and horticultural streams: 'desc1' ($P<0.01$; Appendix 2, Table 27), 'desc2' ($P<0.01$; Appendix 2, Table 27), and 'submerged ($P<0.05$; Appendix 2, Table 37). However, the difference in the CWM abundance modality 'lda2' between native forest and horticultural streams was only marginally significant (*P*=0.05); Appendix 2, Table 33). The following two modalities were only statistically significant between pastoral and horticultural streams: ‘cpi1’ (*P*<0.05; Appendix 2, Table 31) and ‘two’ (*P*<0.05); Appendix 2, Table 35). The difference in the CWM abundance of the modality ‘single’ was only marginally significant (*P*=0.05; Appendix 2, Table 35).

4.1.4.2 Specific trait modalities

Regarding the trait **‘maximum potential size’**, there are five modalities called ‘size1’, size2’, ‘size3’, size4’ and ‘size5, which were analysed for differences across land-use types and DFS (Figure 20).

The CWM abundances of the modality ‘size1’ were higher in pastoral streams, followed closely by native streams and then horticultural streams (Figure 20a). ‘Size1’ was relatively similar across pastoral and horticultural land use (Figure 20a). ‘size1’ did not differ between the three land-use types (*P*=0.28) with sediment included as a covariate (*P*=0.92; Table 20).

The CWM abundances of ‘size2’ were higher in horticultural streams, followed closely by pastoral streams and then native streams; but overall, was relatively similar across all land uses (Figure 20b). The difference in ‘size2’ between the three land-use types was not statistically significant (*P*=0.28) with sediment included as a covariate (*P*=0.17; Table 20).

The CWM abundances of ‘size3’ were higher in horticultural streams, followed by pastoral streams and then native streams (Figures 20c). ‘size3’ did not differ between the three land-use types (*P*=0.26) with sediment included as a covariate (*P*=0.33; Table 20).

The CWM abundances of ‘size4’ were higher in native streams, followed by pastoral streams and then horticultural streams; however, ‘size4’ was relatively similar across pastoral and horticultural land uses (Figure 20d). ‘size4’ did not differ between the three land-use types (*P*=0.59; Table 20) with sediment included as a covariate (*P*=0.40; Table 20).

The CWM abundances of ‘size5’ were higher in horticultural, followed by pastoral streams and then native forest streams: however, they were relatively similar across pastoral and horticultural land uses (Figure 20e). ‘size5’ did not differ between the three land-use types (*P*=0.83) with sediment included as a covariate (*P*=0.30; Table 20).

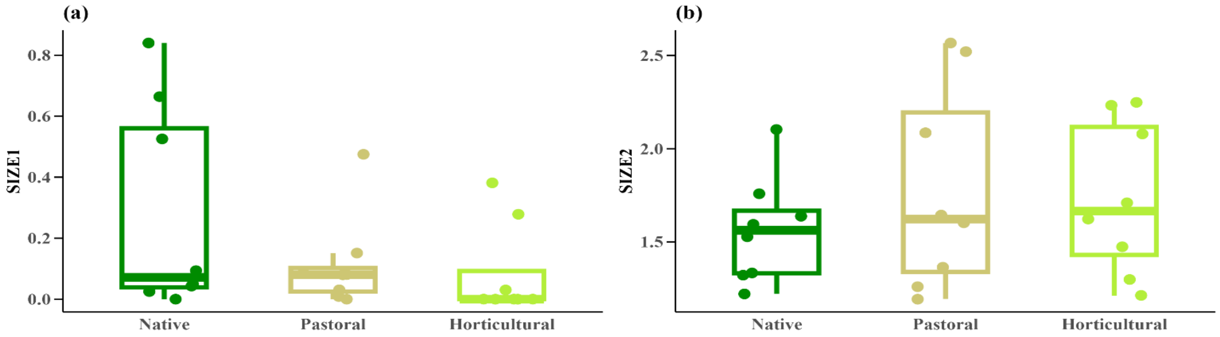

Figure 20. *Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'maximum potential size' across three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) size1, b) size2, c) size3, d) size4, e) size5.*

Table 20. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait maximum potential size' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	Predictor	F-value	df	df.res	P-value
size1	Site type	1.37E+00	2	15.5	0.28
	Sediment	9.79E-03	1	20.0	0.92
size2	Site type	1.38E+00	2	16.3	0.28
	Sediment	2.04E+00	1	18.9	0.17
size3	Site type	1.45E+00	2	17.2	0.26
	Sediment	1.02E+00	1	16.3	0.33
size4	Site type	5.51E-01	2	16.4	0.59
	Sediment	7.58E-01	1	18.7	0.40
size5	Site type	1.89E-01	2	15.6	0.83
	Sediment	1.14E+00	1	19.9	0.30

Regarding the trait **'body flexibility'**, there are three modalities called 'noflex', 'lowflex' and 'highflex', which were analysed against land-use type and DFS (Figure 21).

The CWM abundances of 'noflex' were higher in horticultural streams, followed by pastoral streams and then native streams; however, they were relatively similar across native and pastoral land uses (Figure 21a). 'noflex' showed a statistically significant difference between the three land-use types (*P*<0.05; Table 21) with sediment included as a covariate (*P*=0.74; Table 21). Post-hoc testing revealed a marginally significant difference between native and horticultural streams (*P*=0.05; Appendix 2, Table 22).

The CWM abundances of 'lowflex' were higher in native streams, followed by pastoral streams and then horticultural streams; however, they were relatively similar across native and pastoral land uses (Figure 21b). 'lowflex' did not differ between the three land-use types (*P*=0.14) with sediment included as a covariate (*P*=0.13; Table 21).

The CWM abundances of 'highflex' were higher in pastoral streams, followed by horticultural streams and then native streams, which were relatively similar (Figure 21c). 'highflex' did not differ between the three land-use types (*P*=0.95; Table 21), but sediment had a significant influence as a covariate (*P*<0.05; Table 21).

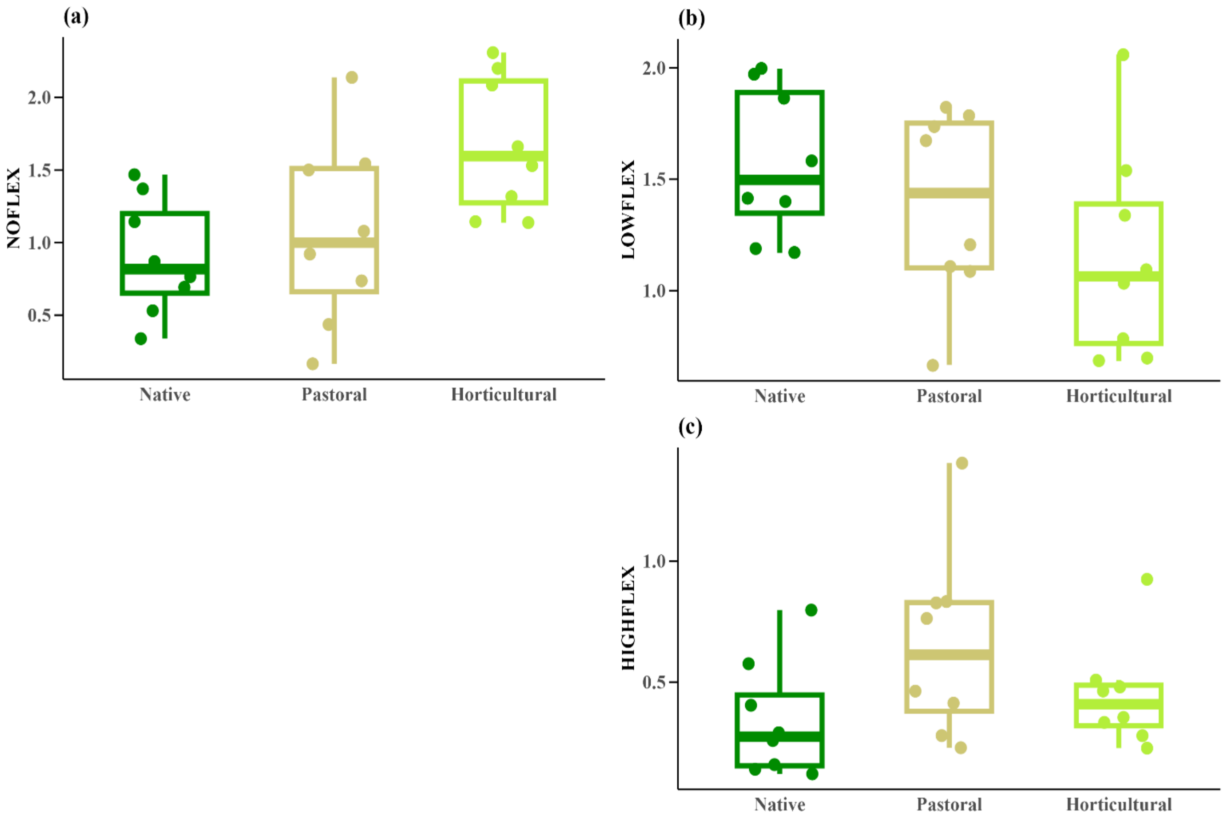

Figure 21. *Changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'body flexibility' across three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) noflex, b) lowflex, c) highflex.*

Table 21. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait' body flexibility' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Predictor**	**F-value**	**df**	**df.res**	**P-value**
noflex	Site type	4.29E+00	2	16.3	**0.03**
	Sediment	1.16E-01	1	18.9	0.74
lowflex	Site type	2.24E+00	2	17.2	0.14
	Sediment	2.56E+00	1	16.3	0.13
highflex	Site type	4.80E-02	2	14	0.95
	Sediment	5.33E+00	1	18	**0.03**

Regarding the trait **‘Body form’**, there are four modalities called ‘streamlined’, ‘flattened’, ‘cylindrical’, and ‘spherical’. These modalities were analysed against land-use type and DFS (Table 23).

The CWM abundances of ‘streamlined’ were higher in pastoral streams, followed by horticultural streams and then native forest streams (Figure 22a). ‘streamlined’ did not differ between the three land-use types (P=0.50; Table 23) with sediment included as a covariate (P=0.56; Table 23).

The CWM abundances of ‘flattened’ were higher in native forest streams, followed by pastoral streams and then horticultural streams (Figure 22b). ‘flattened’ showed a statistically significant difference between the three land-use types (P<0.05; Table 23). Sediment did not have an influence as a covariate (P=0.99; Table 23). Post-hoc testing showed a marginally significant difference (P=0.05) between native forest and horticultural streams (Appendix 2, Table 24).

The CWM abundances of ‘cylindrical’ were higher in horticultural streams, followed by pastoral streams and then horticultural streams (Figure 22c). ‘cylindrical’ did not differ between the three land-use types (P=0.13) with sediment included as a covariate (P=0.59; Table 23).

The CWM abundances of ‘spherical’ were higher in native forest streams, followed by pastoral and then horticultural streams, which were relatively similar (Figure 22d). ‘spherical’ did not differ between the three land-use types (P=0.83) with sediment included as a covariate (P=0.55; Table 23).

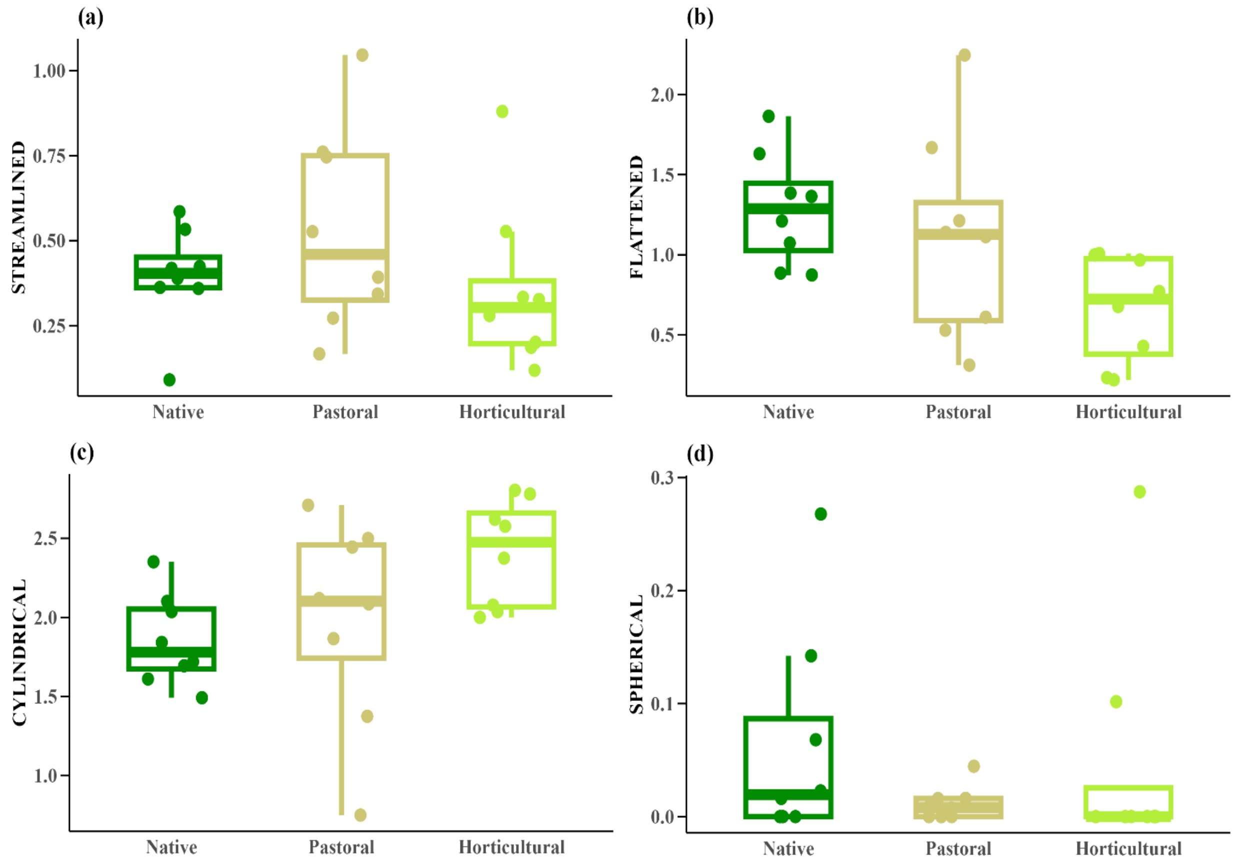

Figure 22. *Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'body form' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) streamlined, b) flattened, c) cylindrical, d) spherical.*

Table 23. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'body form' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Predictor**	**F-value**	**df**	**df.res**	**P-value**
streamlined	Site type	7.35E-01	2	14.9	0.50
	Sediment	3.56E-01	1	19.8	0.56
flattened	Site type	3.77E+00	2	16.3	**0.045**
	Sediment	7.24E-05	1	18.9	0.99
cylindrical	Site type	2.40E+00	2	13.8	0.13
	Sediment	2.96E-01	1	17.3	0.59
spherical	Site type	1.91E-01	2	17.2	0.83
	Sediment	3.64E-01	1	16.4	0.55

Regarding the trait **‘respiration of aquatic stages (not including eggs)’**, there are four modalities called ‘tegument’, ‘gill’, ‘plastron’, and ‘aerial’, which were analysed against land-use type and DFS (Table 25).

The CWM abundances of ‘tegument’ were higher in native forest streams, followed by horticultural streams and then pastoral streams, which were relatively similar (Figure 23a). ‘tegument’ did not differ between the three land-use types (*P*=0.10) with sediment included as a covariate (*P*=0.39; Table 25).

The CWM abundances of ‘gill’ were higher in horticultural streams, followed by pastoral streams and then native forest streams (Figure 23b). ‘gill’ did not differ between land-use types (*P*=0.24) with sediment included as a covariate (*P*=0.67; Table 25).

The CWM abundances of ‘plastron’ were higher in pastoral streams, followed by horticultural streams, which were relatively similar, and then native forest streams (Figure 23c). ‘plastron’ did not differ between land-use types (*P*=0.16) with sediment included as a covariate (*P*=0.76; Table 25).

The CWM abundances of ‘aerial’ were higher in pastoral streams, followed by horticultural streams and then native forest streams, which were relatively similar (Figures 23d). ‘aerial’ did not differ between land-use types (*P*=0.43) with sediment included as a covariate (*P*=0.92; Table 25).

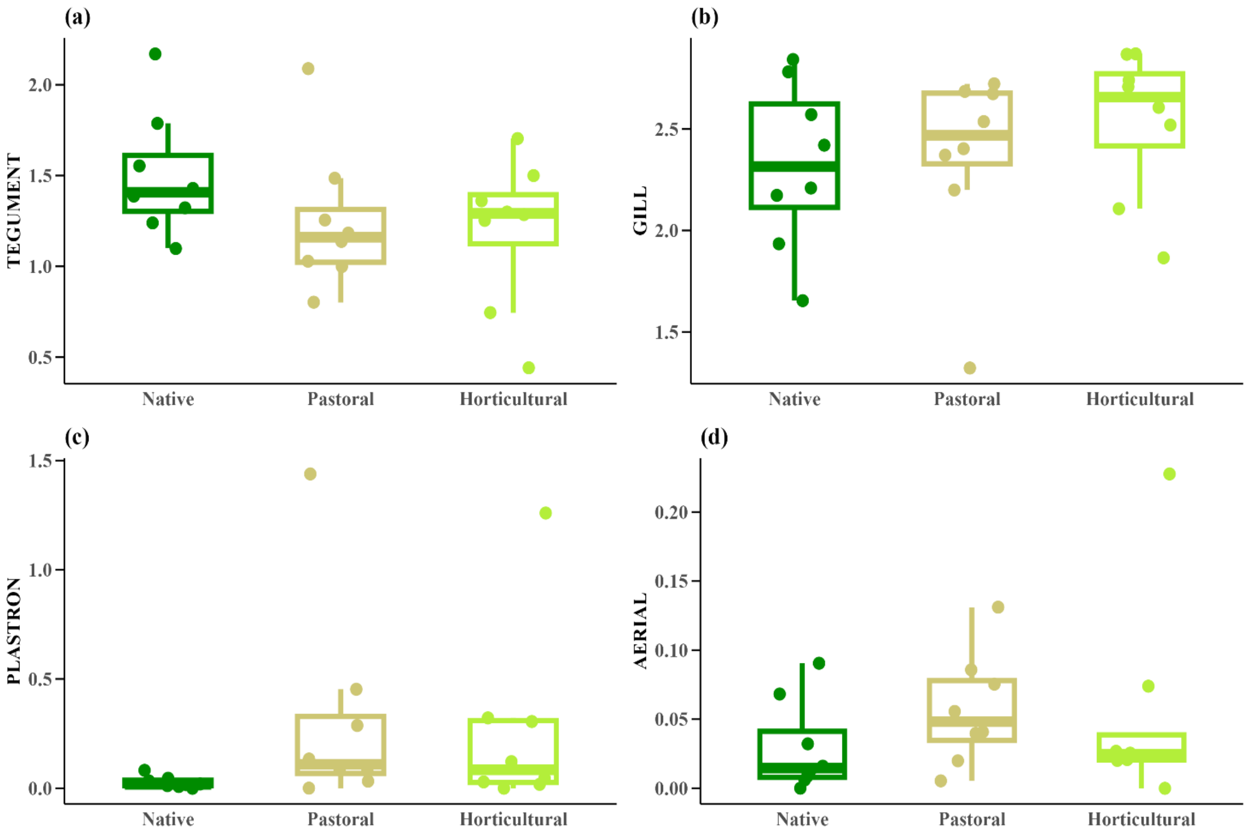

Figure 23. *Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'respiration of aquatic stages (not including eggs)' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) tegument, b) gill, c) plastron, d) aerial.*

Table 25. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'respiration of aquatic stages (not including eggs)' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Predictor**	**F-value**	**df**	**df.res**	**P-value**
tegument	Site type	2.73E+00	2	14.7	0.10
	Sediment	7.82E-01	1	19.6	0.39
gill	Site type	1.59E+00	2	15.4	0.24
	Sediment	1.93E-01	1	20	0.67
plastron	Site type	2.10E+00	2	15.1	0.16
	Sediment	9.88E-02	1	19.9	0.76
aerial	Site type	9.05E-01	2	14.9	0.43
	Sediment	9.37E-03	1	19.7	0.92

Regarding the trait **'Maximum number of descendants per reproductive cycle'**, there are four modalities called 'desc1', 'desc2', 'desc3', and desc4, which were analysed against land-use type and DFS (Table 26).

The CWM abundances of 'desc1' were highest in pastoral streams, followed by horticultural streams, which were relatively similar, and then native forest streams (Figure 24a). 'desc1' differed between the three land-use types (*P*<0.01; Table 26). Sediment did not have a significant influence at α=0.05 as a covariate (*P*=0.06; Table 26). Post-hoc testing revealed a statistically significant difference between native forest and horticultural streams (*P*<0.01; Appendix 2, Table 27).

The CWM abundances of 'desc2' were highest in native forest streams, followed by horticultural and then pastoral streams; albeit all relatively similar across land uses (Figure 24b). 'desc2' did not differ between the three land-use types (*P*=0.45) with sediment included as a covariate (*P*=0.36; Table 26).

The CWM abundances of 'desc3' were highest in native forest streams, followed by pastoral streams and then horticultural streams (Figure 24c). 'desc3' did not differ between the three land-use types (*P*=0.14; Table 26) with sediment included as a covariate (*P*=0.61; Table 26).

The CWM abundances of 'desc4' were highest in native forest streams, followed by pastoral and then horticultural streams, which were relatively similar (Figures 24d). 'desc4' differed significantly between the three land-use types (*P*<0.01) with sediment included as a covariate (*P*=0.26; Table 26). Post-hoc testing revealed that the only statistically significant difference was between native forest and horticultural streams (*P*<0.05; Appendix 2, Table 27).

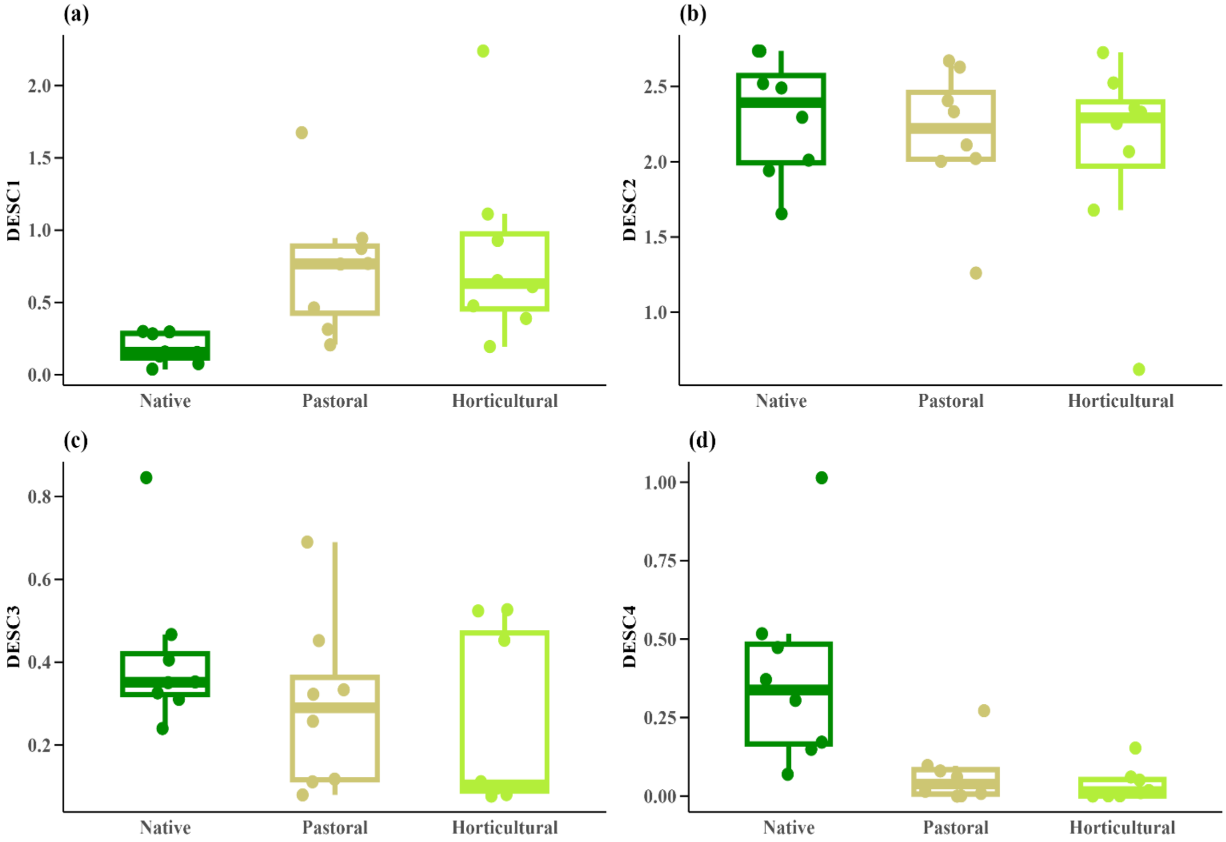

Figure 24. *Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'maximum number of descendants per reproductive cycle' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) desc1, b) desc2, c) desc3, d) desc4.*

Table 26. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'maximum number of descendants per reproductive cycle' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Predictor**	**F-value**	**df**	**df.res**	**P-value**
desc1	Site type	6.73E+00	2	14.6	**0.01**
	Sediment	4.13E+00	1	19.4	0.06
desc2	Site type	8.53E-01	2	16	0.45
	Sediment	9.40E-01	1	19.5	0.34
desc3	Site type	2.26E+00	2	15.1	0.14
	Sediment	2.75E-01	1	19.9	0.61
desc4	Site type	6.14E+00	2	17.5	**0.01**
	Sediment	1.35E+00	1	15.4	0.26

Regarding the trait **'maximum number of reproductive cycles per year'**, there are three modalities called 'semi', 'univ', and 'pluriv', which were analysed against land-use type and DFS (Figure 25).

The CWM abundances of 'semi' were highest in native forest streams, followed by horticultural and then pastoral streams, which were relatively similar (Figures 25a). 'semi' differed between the three land-use types (*P*<0.05; Table 28) controlling for sediment as a covariate (*P*=0.59; Table 28). Post-hoc testing revealed that the only statistically significant difference was between native forest and horticultural streams (*P*<0.05; Appendix 2, Table 29).

The CWM abundances of 'univ' were highest in native forest streams, followed by pastoral and then horticultural streams, which were relatively similar (Figure 25). 'univ' did not differ between the three land-use types (*P*=0.36), but sediment had a significant influence as a covariate (*P*<0.05; Table 28).

The CWM abundances of 'pluriv' were highest in horticultural streams, followed by pastoral streams, which were relatively similar, and then native forest streams (Figures 25c). 'pluriv' did not differ between the three land-use types (*P*=0.30), but sediment had a significant influence as a covariate (*P*<0.05; Table 28).

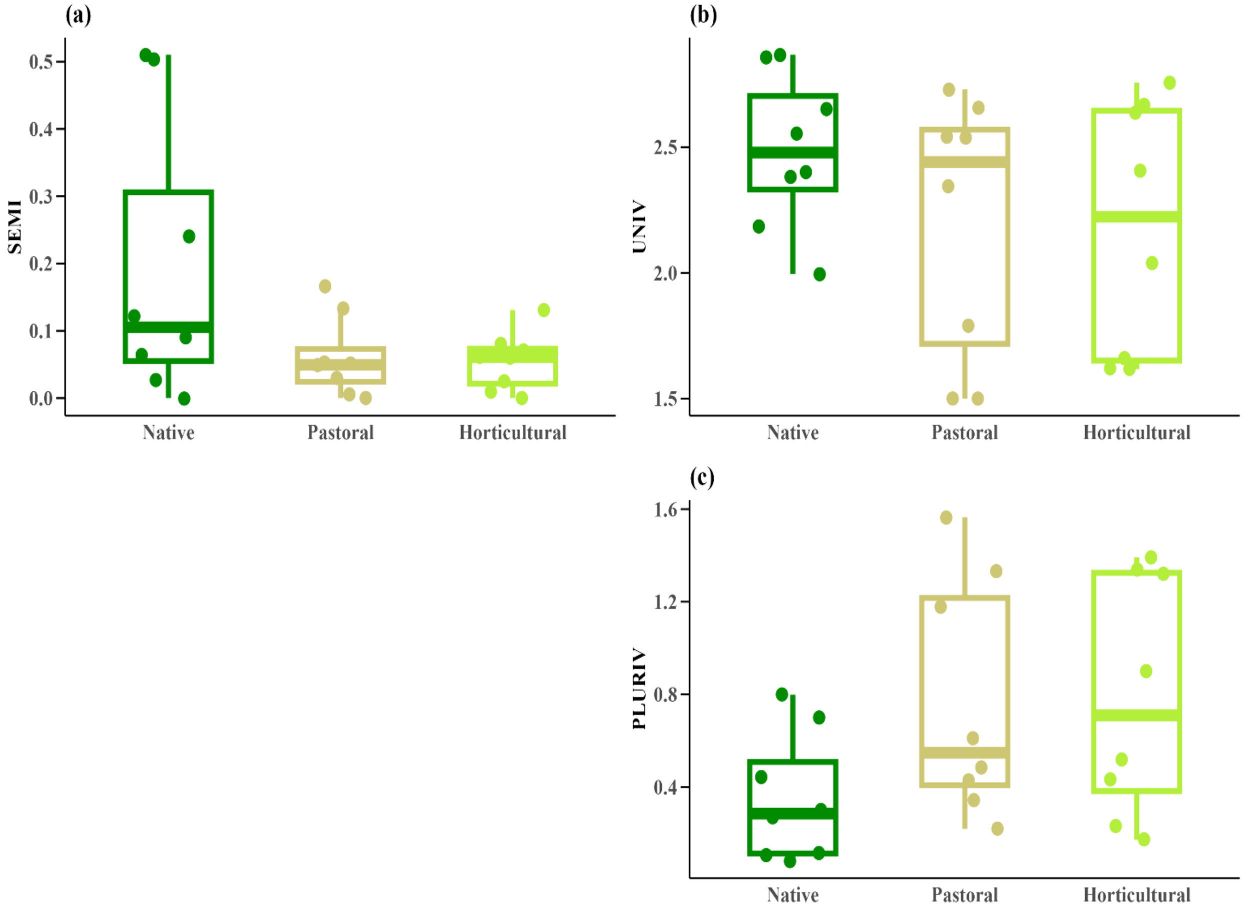

Figure 25. *Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'maximum number of reproductive cycles per year' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) semi, b) univ, c) pluriv.*

Table 28. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'maximum number of reproductive cycles per year' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Predictor**	**F-value**	**df**	**df.res**	**P-value**
semi	Site type	4.11E+00	2	16.2	**0.04**
	Sediment	3.03E-01	1	19	0.59
univ	Site type	1.09E+00	2	16.1	0.36
	Sediment	5.90E+00	1	19.2	**0.03**
pluriv	Site type	1.32E+00	2	15.6	0.30
	Sediment	5.16E+00	1	19.9	**0.03**

Regarding the trait **'number of reproductive cycles per individual'**, the two modalities 'cpi1' and 'cpi2' were analysed against land-use type and DFS (Figures 26).

The CWM abundances of 'cpi1' were highest in native forest streams and relatively similar in pastoral streams but lowest in horticultural streams (Figure 26a). 'cpi1' differed between the three land-use types (*P*<0.01) with sediment included as a covariate (*P*=0.08; Table 30). Post-hoc testing revealed that the only statistically significant difference was between pastoral and horticultural streams (*P*<0.05; Appendix 2, Table 31).

The CWM abundances of 'cpi2' were highest in horticultural streams, followed by pastoral and then native forest streams (Figure 26b). 'cpi2' did not differ between the three land-use types at α=0.05 (*P*=0.07) with sediment included as a covariate (*P*=0.11; Appendix 2, Table 30).

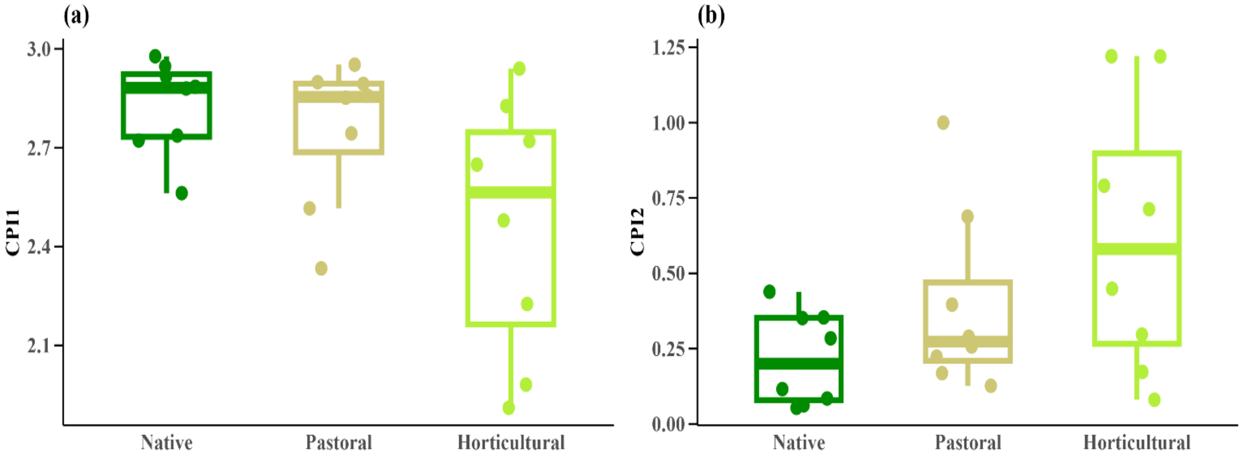

Figure 26. *Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'number of reproductive cycles per individual' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) CPI, b) cpi2.*

Table 30. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'Maximum number of reproductive cycles per individual' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	Predictor	F-value	df	df.res	P-value
cpi1	Site type	5.66E+00	2	17.3	**0.01**
	Sediment	3.59E+00	1	16.1	0.08
cpi2	Site type	3.15E+00	2	17.1	0.07
	Sediment	2.90E+00	1	16.7	0.11

Regarding the trait **‘Life duration of adults’**, the four modalities called ‘lda2’, ‘lda3’, ‘lda4’ and ‘lda5’ were analysed against land-use type and DFS (Figure 27). The modality ‘lda1’ was excluded from the analysis because it was not present in any of the taxa recorded.

The CWM abundances of ‘lda2’ was highest in native forest streams, followed by pastoral streams and then closely by horticultural streams, which was relatively similar (Figure 27a). ‘lda2’ differed between the three land-use types ($P<0.05$) with sediment included as a covariate ($P=0.26$; Table 33). Post-hoc testing revealed that the only statistically significant relationship was between native forest and horticultural land use ($P<0.05$; Appendix 2, Table 33).

The CWM abundances of ‘lda3’ were highest in native forest streams, followed by horticultural streams and then pastoral streams (Figure 27b). ‘lda3’ did not differ between the three land-use types ($P=0.97$) with sediment included as a covariate ($P=0.94$; Table 33).

The CWM abundances of ‘lda4’ were highest in pastoral streams, followed by horticultural streams and then native streams (Figures 27c). ‘lda4’ did not differ between the three land-use types at $\alpha=0.05$ ($P=0.06$) with sediment included as a covariate ($P=0.26$; Table 32).

The CWM abundance of ‘lda5’ was highest in horticultural streams, followed by pastoral streams and then native streams, which were relatively similar (Figures 27d). ‘lda5’ did not differ between the three land-use types ($P=0.09$) with sediment included as a covariate ($P=0.56$; Table 32).

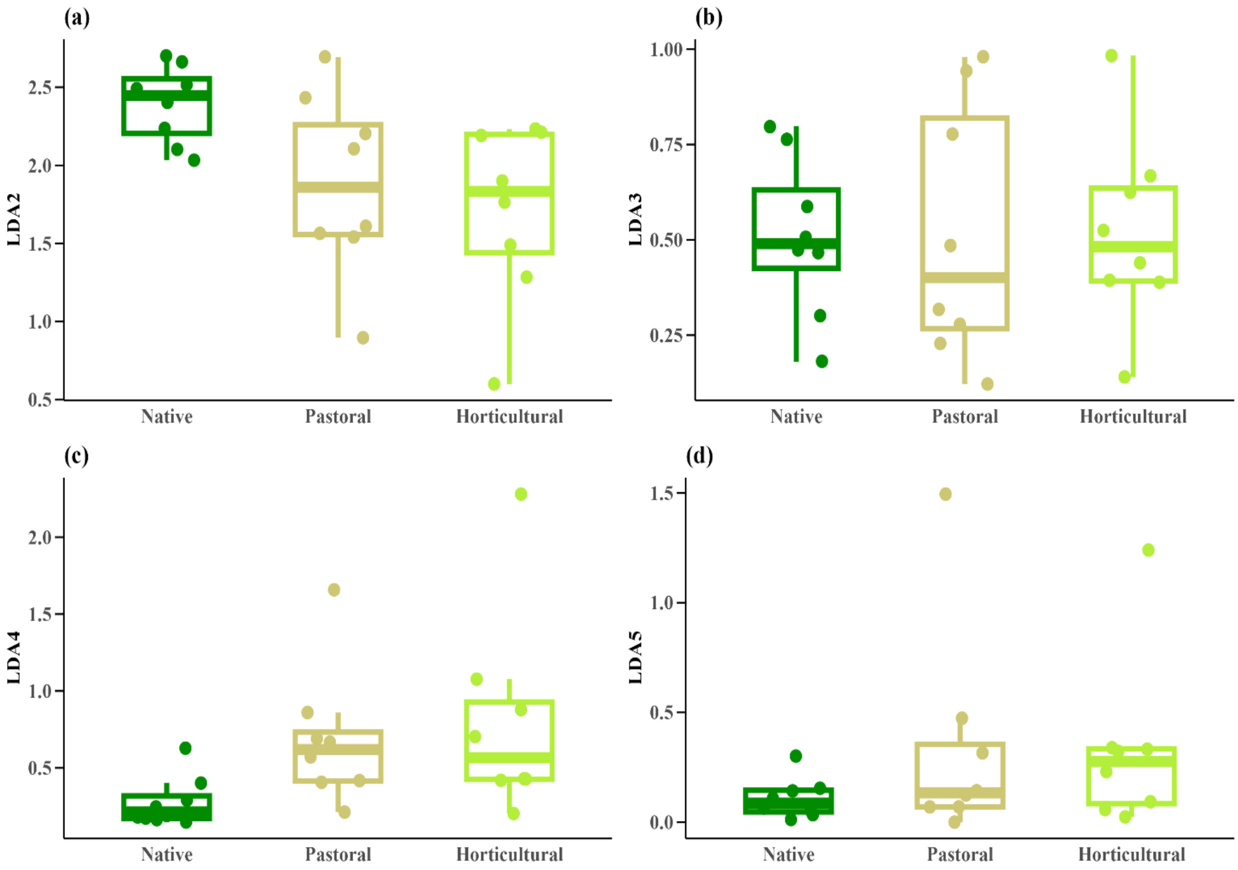

Figure 27. *Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'life duration of adults' across three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. lda1, b) lda2, c) lda3, d) lda4.*

Table 32. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'Life duration of adults' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	Predictor	F-value	df	df.res	P-value
lda2	Site type	4.62E+00	2	14.6	**0.03**
	Sediment	1.36E+00	1	19.3	0.26
lda3	Site type	3.46E-02	2	15.8	0.97
	Sediment	6.83E-03	1	19.7	0.94
lda4	Site type	3.47E+00	2	15.3	0.06
	Sediment	1.32E+00	1	20	0.26
lda5	Site type	2.81E+00	2	14.5	0.09
	Sediment	3.52E-01	1	19.1	0.56

Regarding the trait **‘Reproductive techniques’**, the three modalities called ‘single’, ‘herma’ and ‘two’ were analysed against land-use type and DFS (Figure 28).

The CWM abundances of ‘single’ were highest in horticultural streams, followed by pastoral and then native forest streams (Figure 28a). ‘single’ differed between the three land-use types (P<0.05) with sediment included as a covariate (P=0.07; Table 34). Post-hoc testing revealed a (marginally) statistically significant difference between pastoral and horticultural streams (P=0.05; Appendix 2, Table 35).

The CWM abundances of ‘herma’ were highest in native forest streams, followed by horticultural and then pastoral streams, which were relatively similar (Figure 28b). ‘herma’ did not differ between the three land-use types (P=0.69) with sediment included as a covariate (P=0.76; Table 34).

The CWM abundances of ‘two’ were relatively similar between pastoral and native forest streams and lowest in horticultural streams (Figure 28c). ‘two’ differed between the three land-use types (P<0.05; Table 34) with sediment included as a covariate (P=0.20; Table 34). Post-hoc testing revealed that the only statistically significant difference was between pastoral and horticultural streams (P<0.05; Appendix 2, Table 35).

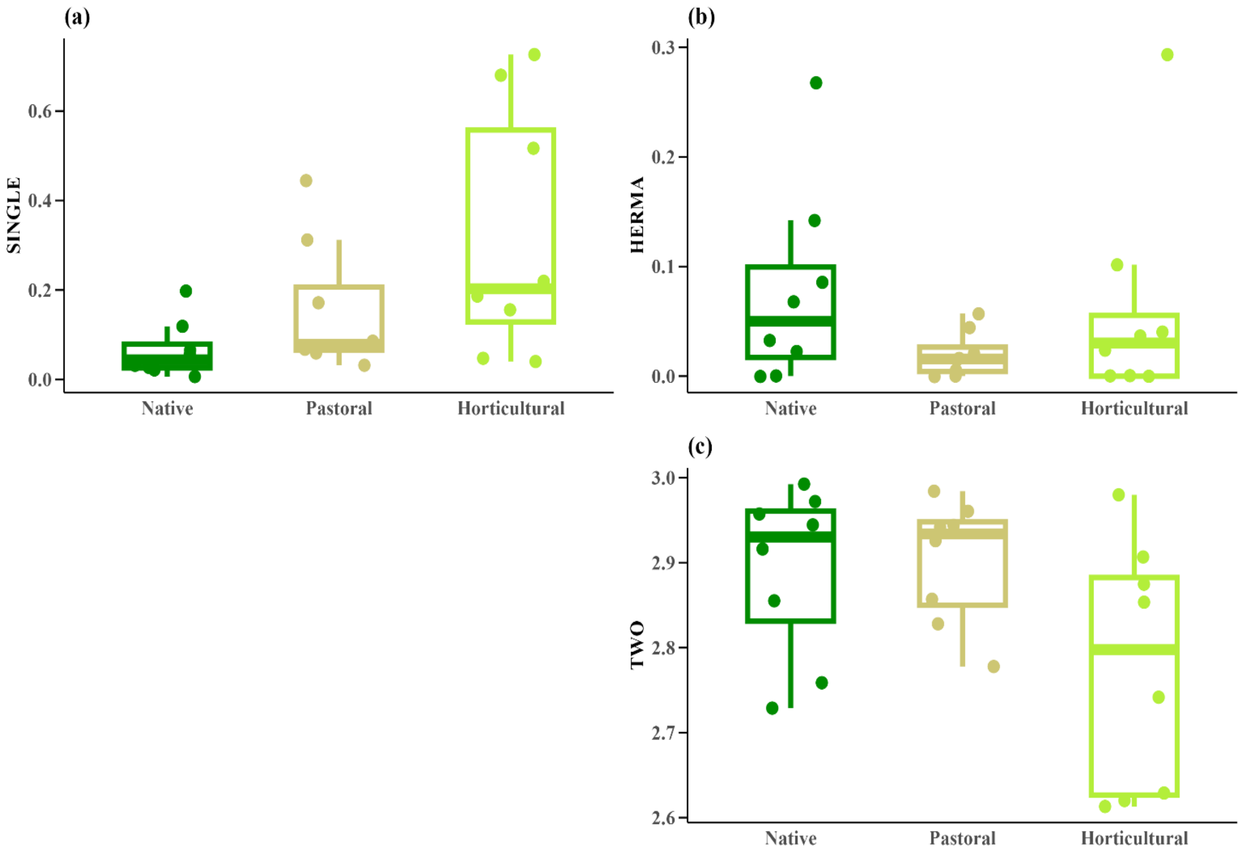

Figure 28. *Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'reproductive techniques' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) single, b) herma, c) two.*

Table 34. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'reproductive techniques' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	Predictor	F-value	df	df.res	P-value
single	Site type	4.64E+00	2	17.5	**0.02**
	Sediment	3.71E+00	1	15.4	0.07
herma	Site type	3.81E-01	2	17.1	0.69
	Sediment	9.59E-02	1	16.7	0.76
two	Site type	4.66E+00	2	16.8	**0.03**
	Sediment	1.76E+00	1	17.6	0.20

Regarding the trait **‘Oviposition site’**, the four modalities called ‘surface’, ‘submerged’, ‘terrestrial’, and ‘eggendo’ were analysed against land-use type and DFS (Figure 29).

The CWM abundances of ‘surface’ were highest in native forest streams, followed by horticultural and then pastoral streams, which were relatively similar (Figure 29a). ‘surface’ did not differ between the three land-use types (P=0.08), but sediment had a significant influence as a covariate (P<0.05; Table 36).

The CWM abundances of ‘submerged’ were highest in pastoral streams, followed by horticultural streams, which were relatively similar, and then native forest streams (Figure 29b). ‘submerged’ differed between the three land-use types (P<0.05; Table 36) and sediment had a significant influence as a covariate (P<0.05; Table 36). Post-hoc testing revealed that the only statistically significant difference was between native forest and horticultural streams (P<0.05; Appendix 2, Table 37).

The CWM abundances of ‘terrestrial’ were highest in pastoral streams, followed by horticultural streams, which were relatively similar, and then native forest streams (Figure 29c). ‘terrestrial’ did not differ between the three land-use types (P=0.18) with sediment included as a covariate (P=0.87; Table 36).

The CWM abundances of ‘eggendo’ were highest in pastoral streams, with native and horticultural streams having the same low CWM abundances of this modality (Figure 29c). ‘eggendo’ did not differ between the three land-use types (P=0.29; Table 36) with sediment included as a covariate (P=0.94; Table 36).

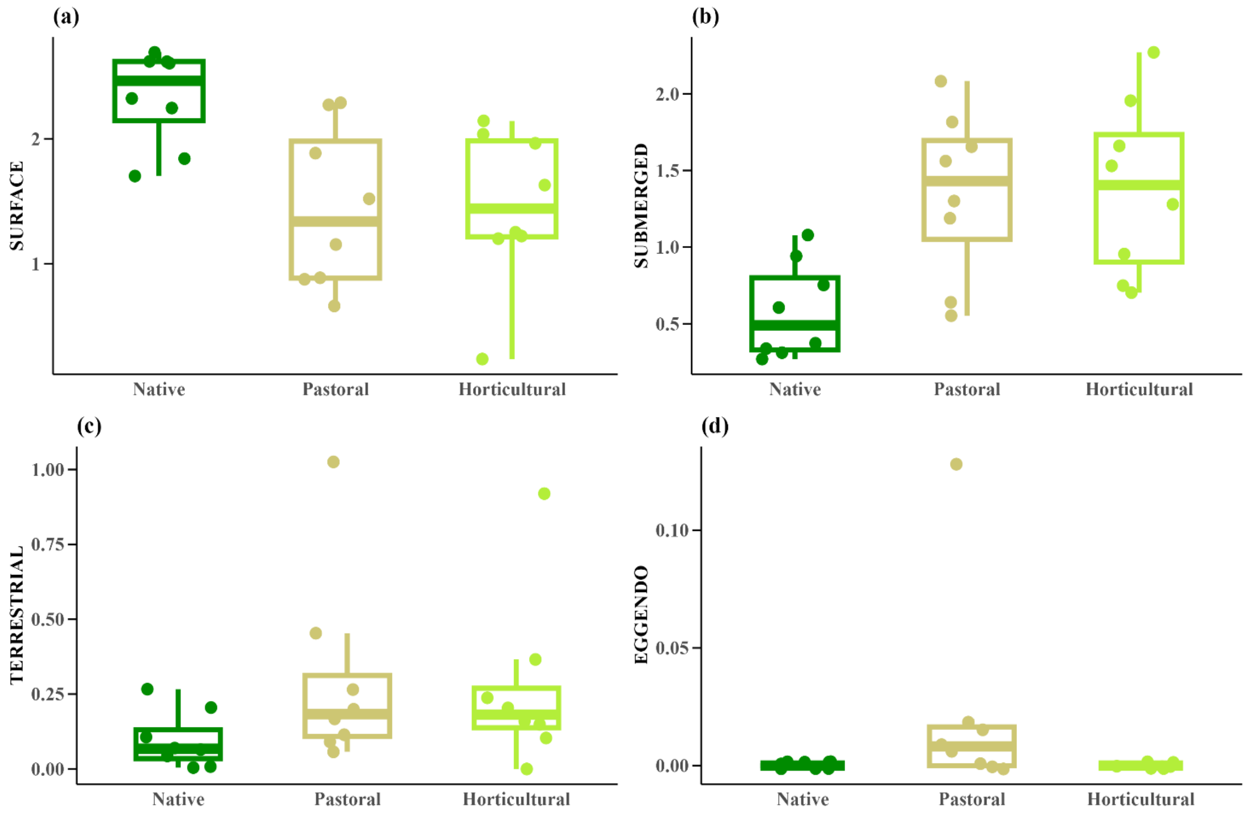

Figure 29. *Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the trait 'oviposition sites' modalities across three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) surface, b) submerged, c) terrestrial, d) eggendo.*

Table 36. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'oviposition sites' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	Predictor	F-value	df	df.res	P-value
surface	Site type	3.07E+00	2	14.7	0.08
	Sediment	4.91E+00	1	19.6	**0.04**
submerged	Site type	4.79E+00	2	15.3	**0.02**
	Sediment	4.76E+00	1	20	**0.04**
terrestrial	Site type	1.94E+00	2	15.2	0.18
	Sediment	2.98E-02	1	20	0.87
eggendo	Site type	1.34E+00	2	16.4	0.29
	Sediment	6.92E-03	1	18.7	0.94

Regarding the trait **‘Egg/egg mass’**, the three modalities ‘eggfree’, ‘eggcement’. and ‘eggprotected’ were analysed against land-use type and DFS (Figure 30).

The CWM abundances of ‘eggfree’ were highest in native forest streams, followed by pastoral and then horticultural streams, which were relatively similar (Figures 30a). ‘eggfree’ did not differ between the three land-use types at $\alpha=0.05$ ($P=0.06$) with sediment included as a covariate ($P=0.07$; Table 38).

The CWM abundances of ‘eggcement’ were highest in pastoral streams, followed by horticultural and then native forest streams (Figure 30b). ‘eggcement’ did not differ between the three land-use types ($P=0.08$; Table 38) with sediment included as a covariate ($P=0.23$; Table 38).

The CWM abundances of ‘eggprotected’ were highest in horticultural streams, followed by pastoral and then native forest streams (Figure 30c). ‘eggprotected’ differed between the three land-use types ($P<0.05$) with sediment included as a covariate ($P=0.08$; Table 31). Post-hoc testing revealed no statistically significant differences between land-use types at $\alpha=0.05$, although the contrasts between pastoral and horticultural streams ($P<0.06$) and native forest and horticultural streams ($P<0.09$) were noted (Appendix 2, Table 39).

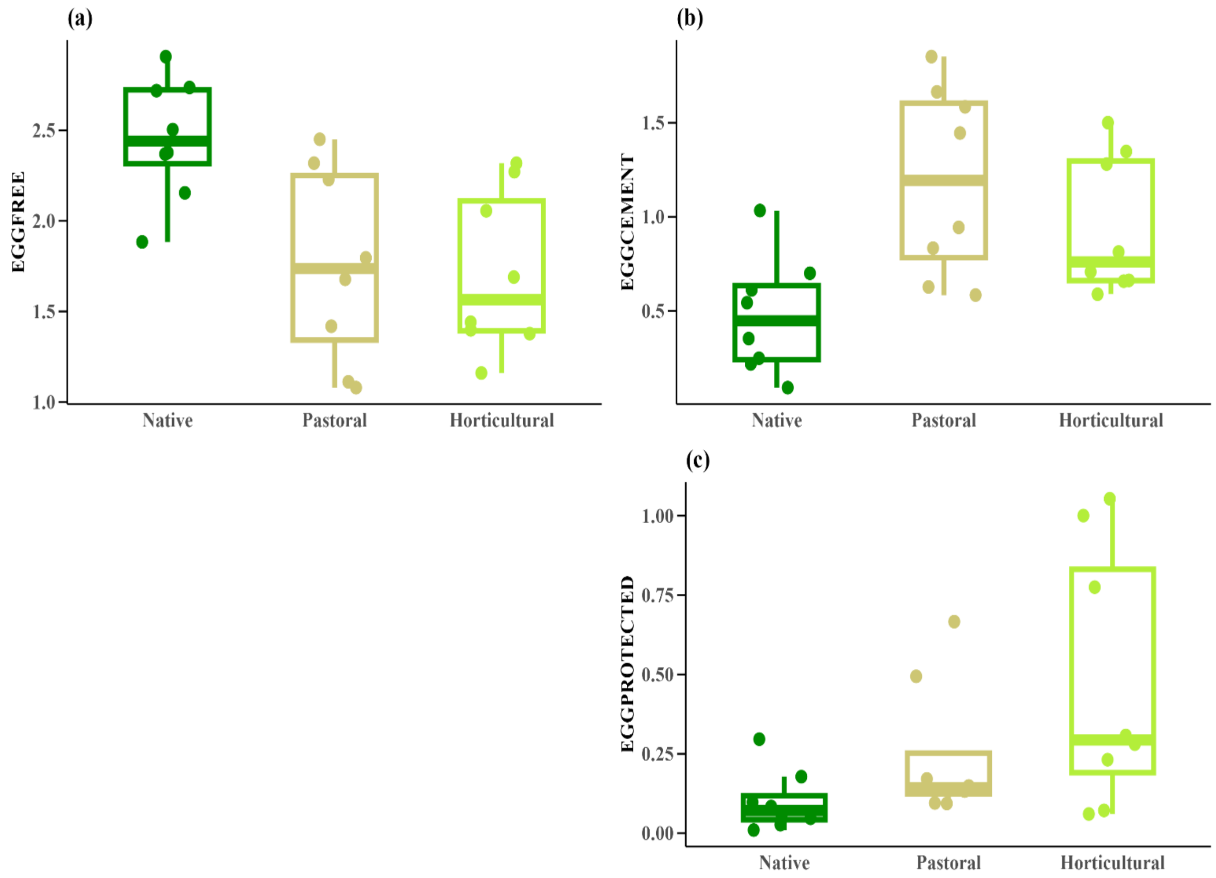

Figure 30. *Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the trait 'egg/egg mass' modalities across three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) eggfree, b) eggcement, c) eggprotected.*

Table 38. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'egg/egg mass' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	Predictor	F-value	df	df.res	P-value
eggfree	Site type	3.41E+00	2	15.3	0.06
	Sediment	3.70E+00	1	20	0.07
eggcement	Site type	2.97E+00	2	15.3	0.08
	Sediment	1.57E+00	1	20	0.23
eggprotected	Site type	4.29E+00	2	17.4	**0.03**
	Sediment	3.40E+00	1	15.6	0.08

Regarding the trait **'Aquatic stages'**, there are three modalities called 'aduandlar', 'aduorlar' and 'larandpup', which were analysed against land-use type and DFS (Figure 31).

The CWM abundances of 'aduandlar' were highest in horticultural streams, followed by pastoral and then native forest streams (Figure 31a). 'aduandlar' differed between the three land-use types (*P* <0.05) with sediment included as a covariate (*P* >0.05; Table 40). Post-hoc testing revealed a statistically significant difference between native forest and horticultural streams (*P*<0.001; Appendix 2, Table 41).

The CWM abundances of 'aduorlar' were highest in native forest streams, followed by pastoral and then horticultural streams (Figure 31b). 'aduorlar' The difference in the CWM abundance between the three land-use types was statistically significant (*P*<0.01; Table 40) with sediment included as a covariate (*P*=0.71; Table 40). Post-hoc testing revealed a statistically significant difference was between native forest and horticultural streams (*P*<0.05; Appendix 2, Table 41.

The CWM abundances of 'larandpup' was highest in horticultural streams, followed by pastoral and then native forest streams (Figure 31c). 'larandpup' did not differ between the three land-use types (*P*=0.47; Table 40) with sediment included as a covariate (*P*=0.84; Table 40).

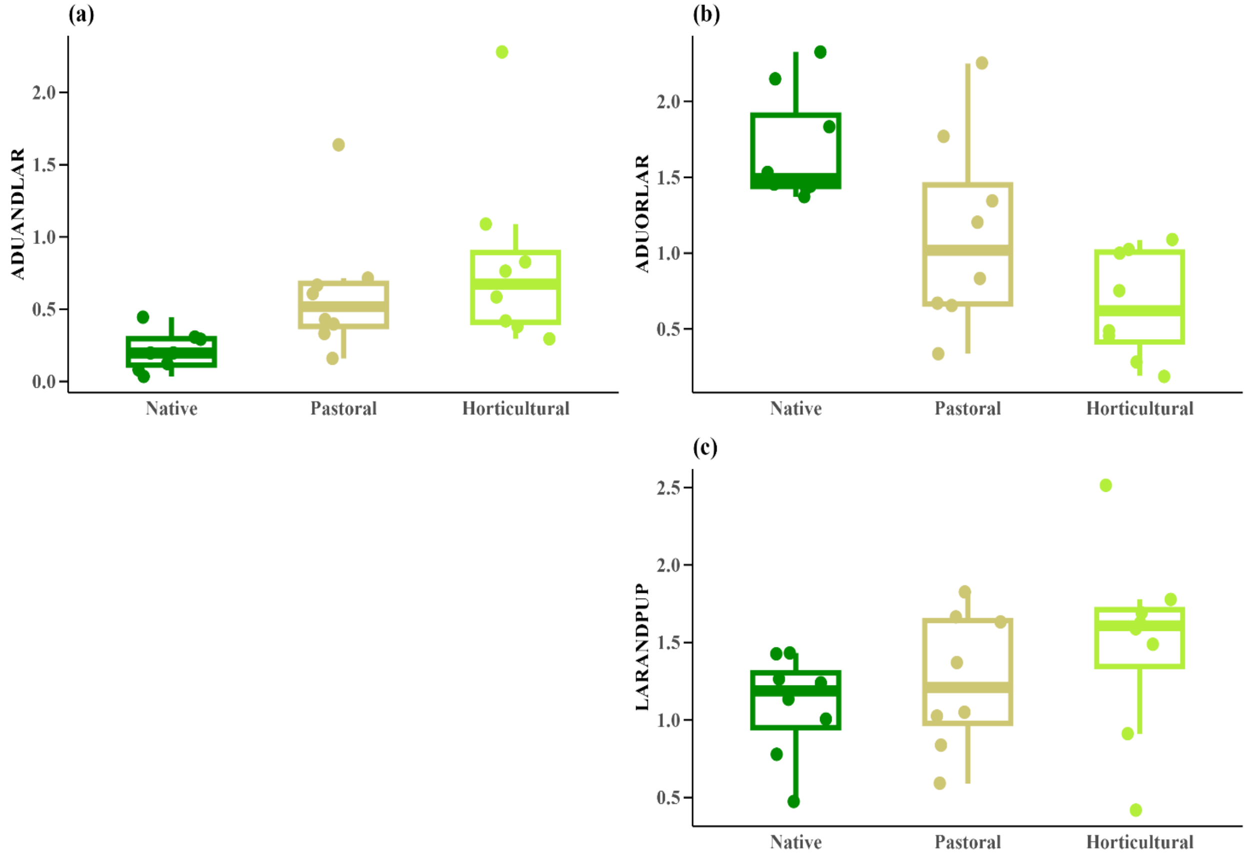

Figure 31. *Boxplots of the changes in the Community Weighted Mean (CWM) abundances of the trait 'aquatic stages' modalities across three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) aduandlar, b) andorlar, c) larandpup.*

Table 40. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'aquatic stages' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	Predictor	F-value	df	df.res	P-value
aduandlar	Site type	9.53	2	14.3	**0.002**
	Sediment	1.38	1	18.7	0.26
aduorlar	Site type	10.53	2	14.7	**0.001**
	Sediment	0.14	1	19.4	0.71
larandpup	Site type	0.79	2	15.0	0.47
	Sediment	0.04	1	19.8	0.84

Regarding the trait **‘Dissemination potential’**, the three modalities ‘disslow’, ‘dissmedium’ and ‘disshigh’ were analysed against land-use type and DFS (Figure 32).

The CWM abundances of ‘disslow’ were highest in horticultural streams, followed closely by native streams, which were relatively similar, and then pastoral streams (Figure 32a). ‘disslow’ did not differ between the three land-use types (*P*=0.13) with sediment included as a covariate (*P*=0.76; Table 42).

The CWM abundances of ‘dissmedium’ were highest in the native forest streams, followed by pastoral streams, which were relatively similar, and then horticultural streams (Figure 32). ‘dissmedium’ did not differ between the three land-use types (*P*=0.38) with sediment included as a covariate (*P*=0.29; Table 42).

The CWM abundances of ‘disshigh’ were highest in pastoral streams, followed by horticultural and then native forest streams (Figure 32c). ‘disshigh’ did not differ between the three land-use types (*P*=0.12) with sediment included as a covariate (*P*=0.96; Table 42).

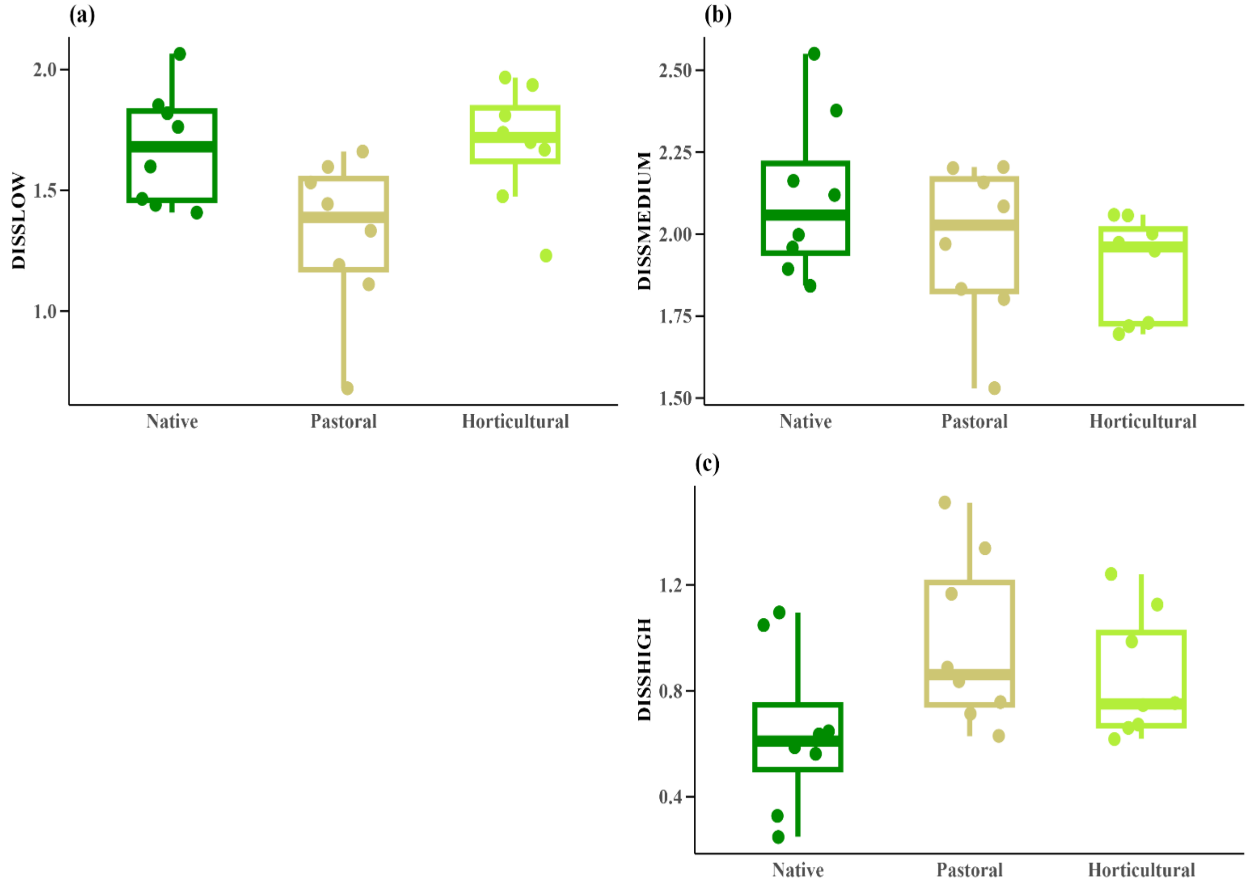

Figure 32. *Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'dissemination potential (all stages)' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) disslow, b) dissmedium, c) disshigh.*

Table 42. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'dissemination potential (all stages)' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Predictor**	**F-value**	**df**	**df.res**	**P-value**
disslow	Site type	2.31E+00	2	16.4	0.13
	Sediment	9.93E-02	1	18.7	0.76
dissmedium	Site type	1.03E+00	2	16.2	0.38
	Sediment	1.19E+00	1	19	0.29
disshigh	Site type	2.42E+00	2	15.6	0.12
	Sediment	2.06E-03	1	19.9	0.96

Regarding the trait **'Attachment to substrate of aquatic stages (excluding eggs)'**, the four modalities 'swimmer', 'crawler', 'burrower' and 'attached' were analysed against land-use type and DFS (Figure 33).

The CWM abundances of ‘swimmer’ were highest in native forest streams, followed by horticultural and then horticultural streams (Figure 33a). The difference in ‘swimmer’ between the three land-use types was marginally statistically significant (*P*=0.05) with sediment included as a covariate (*P*=0.57; Table 43). Post-hoc testing revealed that there were no statistically significant differences at α=0.05, although the contrast between native forest and horticultural streams was noted (*P*=0.07; Appendix 2, Table 44).

The CWM abundances of ‘crawler’ was highest in horticultural streams, followed by native forest streams, which were relatively similar, and then pastoral streams (Figure 33b). ‘crawler’ did not differ between the three land-use types (*P*=0.75) with sediment included as a covariate (*P*=0.31; Table 43).

The CWM abundances of ‘burrower’ were highest in pastoral streams, followed by native forest and then horticultural streams (Figures 33c). ‘burrower’ did not differ between the three land-use types (*P*=0.22; Table 43) with sediment included as a covariate (*P*=0.26; Table 43).

The CWM abundances of ‘attached’ were highest in native forest streams, followed by pastoral and then horticultural streams, which were relatively similar (Figure 33d). ‘attached’ did not differ between the three land-use types (*P*=0.21) with sediment included as a covariate (*P*=0.94; Table 43).

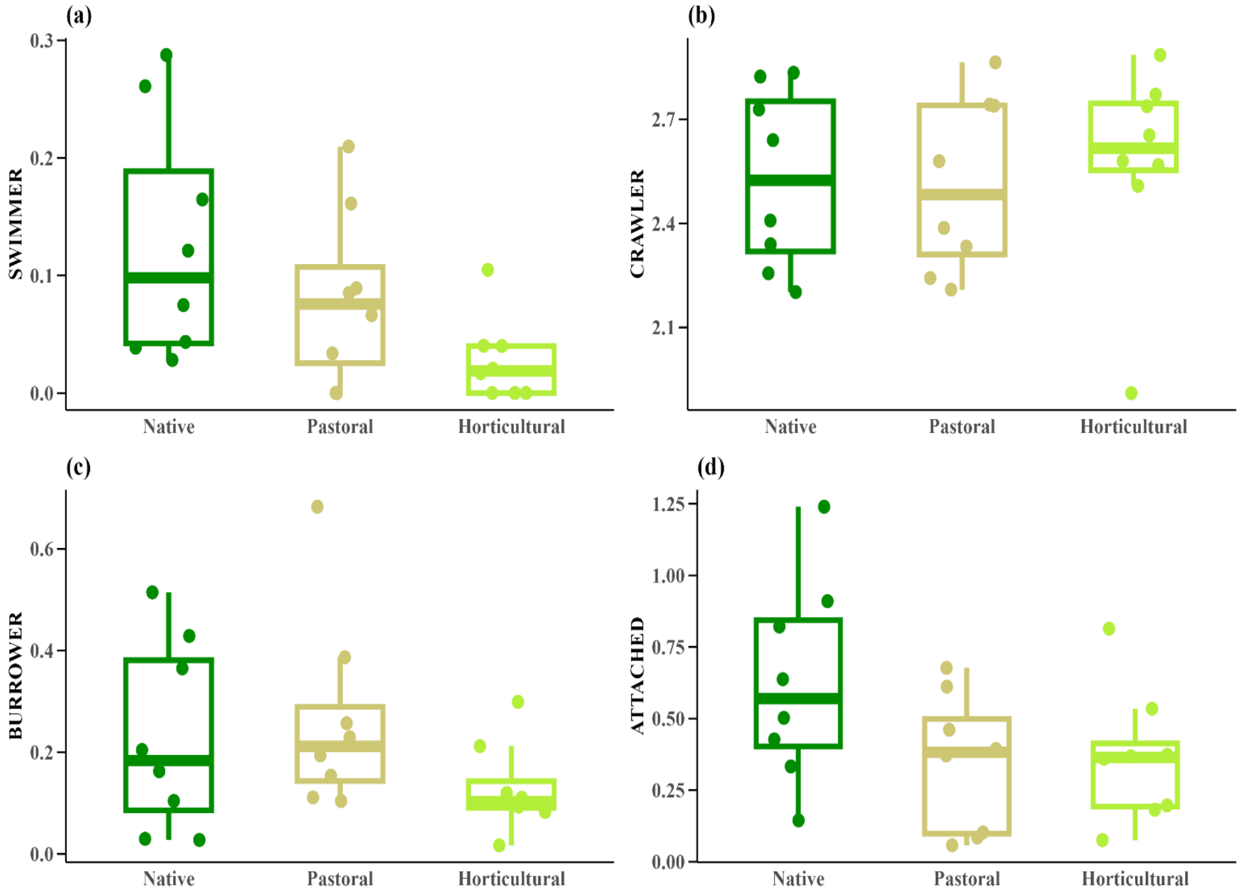

Figure 33. *Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'attachment to substrate of aquatic stages (excluding eggs)' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) swimmer, b) crawler, c) burrower, d) attached.*

Table 43. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'attachment to substrate of aquatic stages (excluding eggs)' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Predictor**	**F-value**	**df**	**df.res**	**P-value**
swimmer	Site type	3.65E+00	2	14.9	**0.05**
	Sediment	3.41E-01	1	19.7	0.57
crawler	Site type	2.90E-01	2	14.5	0.75
	Sediment	1.11E+00	1	19.2	0.31
burrower	Site type	1.69E+00	2	17	0.22
	Sediment	1.37E+00	1	17.1	0.26
attached	Site type	1.72E+00	2	15.5	0.21
	Sediment	6.38E-03	1	20	0.94

Regarding the trait 'Feeding habits', there are 6 modalities called 'shredder', 'scraper', 'deposit', 'filterfeed', 'predator' and 'algalp', which were analysed against land-use type (Native forest, pastoral & horticultural) and DFS/SAM2 (Figure 34).

The Community Weighted Mean (CWM) abundance of the modality 'shredder' was highest in pastoral streams, followed by horticultural and then native forest streams (Figure 34a). However, the modality 'shredder' median CWM abundance was relatively similar across all land uses (Figure 34a). The difference in the CWM abundance between the three land-use types was not statistically significant (*P* >0.05; Table 45) but became significant when sediment was included as a covariate (*P* <0.05; Table 45). Furthermore, post-hoc testing revealed that there were no statistically significant differences between any land-use type (Native forest, pastoral & horticultural land-use) (*P* >0.05; Appendix 2, Table 46).

The Community Weighted Mean (CWM) abundance of the modality 'scraper' was highest in horticultural streams, followed by pastoral and then native forest streams, which were relatively similar (Figure 34b). The difference in the CWM abundance between the three land-use types was not statistically significant (*P* >0.05) (Table 45) but was when sediment was included as a covariate (*P* >0.05; Table 45). Furthermore, post-hoc testing revealed that there were no statistically significant differences between any land-use type (Native forest, pastoral & horticultural land-use) (*P* >0.05; Appendix 2, Table 46).

The Community Weighted Mean (CWM) abundance of the modality 'deposit' was highest in pastoral streams, followed by native forest and then horticultural streams (Figure 34c). However, the modality 'deposit' median CWM abundance was relatively similar across all land uses (Figure 34c). The difference in the CWM abundance between the three land-use types was not statistically significant (*P* >0.05; Table 45) but was when sediment was included as a covariate (*P* >0.05; Table 45).

The Community Weighted Mean (CWM) abundance of the modality 'filterfeed' was highest in native forest streams, followed by pastoral and then horticultural streams, which were relatively similar (Figure 34d). The difference in the CWM abundance between the three land-use types was not statistically significant (*P* >0.05; Table 45) but was when sediment was included as a covariate (*P* >0.05; Table 45).

The Community Weighted Mean (CWM) abundance of the modality 'predator' was highest in horticultural streams, followed by native and then pastoral streams (Figure 34e). However, the modality 'predator' median CWM abundance was relatively similar across all land uses (Figure 34e). The difference in the CWM abundance between the three land-use types was not statistically significant (*P* >0.05; Table 45) but was when sediment was included as a covariate (*P* >0.05; Table 45).

The Community Weighted Mean (CWM) abundance of the modality 'algalp ' was highest in pastoral streams, followed by horticultural and then native forest streams (Figure 34f). However, the modality 'algalp' median CWM abundance was low and relatively similar across all land uses (Figure 34f). The difference in the CWM abundance between the three land-use types was not statistically significant (*P* >0.05; Table 45) but was when sediment was included as a covariate (*P* >0.05; Table 45).

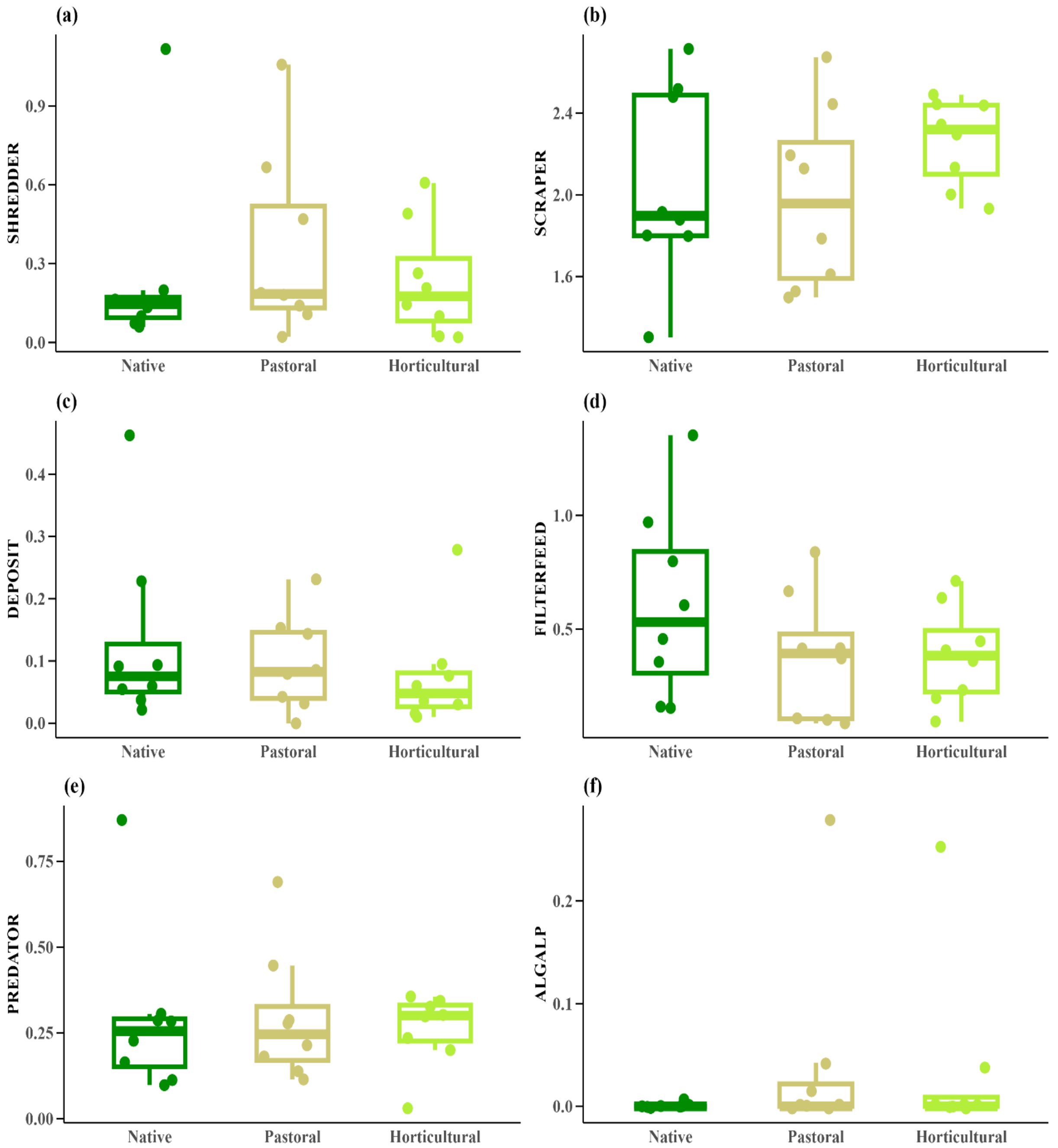

Figure 34. *Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'feeding habits' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) shredder, b) scraper, c) deposit, d) filterfeed, e) predator, f) algalp.*

Table 45. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'feeding habits' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Predictor**	**F-value**	**df**	**df.res**	**P-value**
Shredder	Site type	3.80E-01	2	15.1	0.7
	Sediment	4.73E+00	1	19.9	**0.04**
Scraper	Site type	1.16E+00	2	16.8	0.3
	Sediment	1.08E+00	1	17.6	0.3
Deposit	Site type	3.41E-01	2	17.4	0.7
	Sediment	2.95E-03	1	15.8	1.0
Filterfeed	Site type	4.36E-01	2	16.9	0.7
	Sediment	6.85E-01	1	17.4	0.4
Predator	Site type	1.64E+00	2	16.2	0.2
	Sediment	3.41E+00	1	19	0.1
Algalp	Site type	1.93E-01	2	15.7	0.8
	Sediment	2.06E-01	1	19.8	0.7

Regarding the trait **'Dietary preferences'**, the three modalities 'specialist', 'moderatespe' and 'generalist' were analysed against land-use type and DFS (Figures 35).

The CWM abundances of 'specialist' were highest in pastoral streams, followed by horticultural and then native forest streams, which were relatively similar (Figure 35a). 'specialist' did not differ between the three land-use types (*P*=0.55), but sediment had a significant influence as a covariate (*P*<0.01; Table 47).

The CWM abundances of 'moderatespe' were highest in native forest streams, followed by horticultural and then pastoral streams (Figure 35b). 'moderatespe' did not differ between the three land-use types (*P*=0.11), but sediment had a significant influence as a covariate (*P*<0.001; Table 47).

The CWM abundances of 'generalist' was highest in pastoral streams, followed very closely by horticultural streams and then native forest streams (Figure 35c). However, the median CWM abundances of 'generalist' were relatively similar across all land uses (Figure 35c). 'generalist' did not differ between the three land-use types (*P*=0.29) with sediment included as a covariate (*P*=0.42; Table 47).

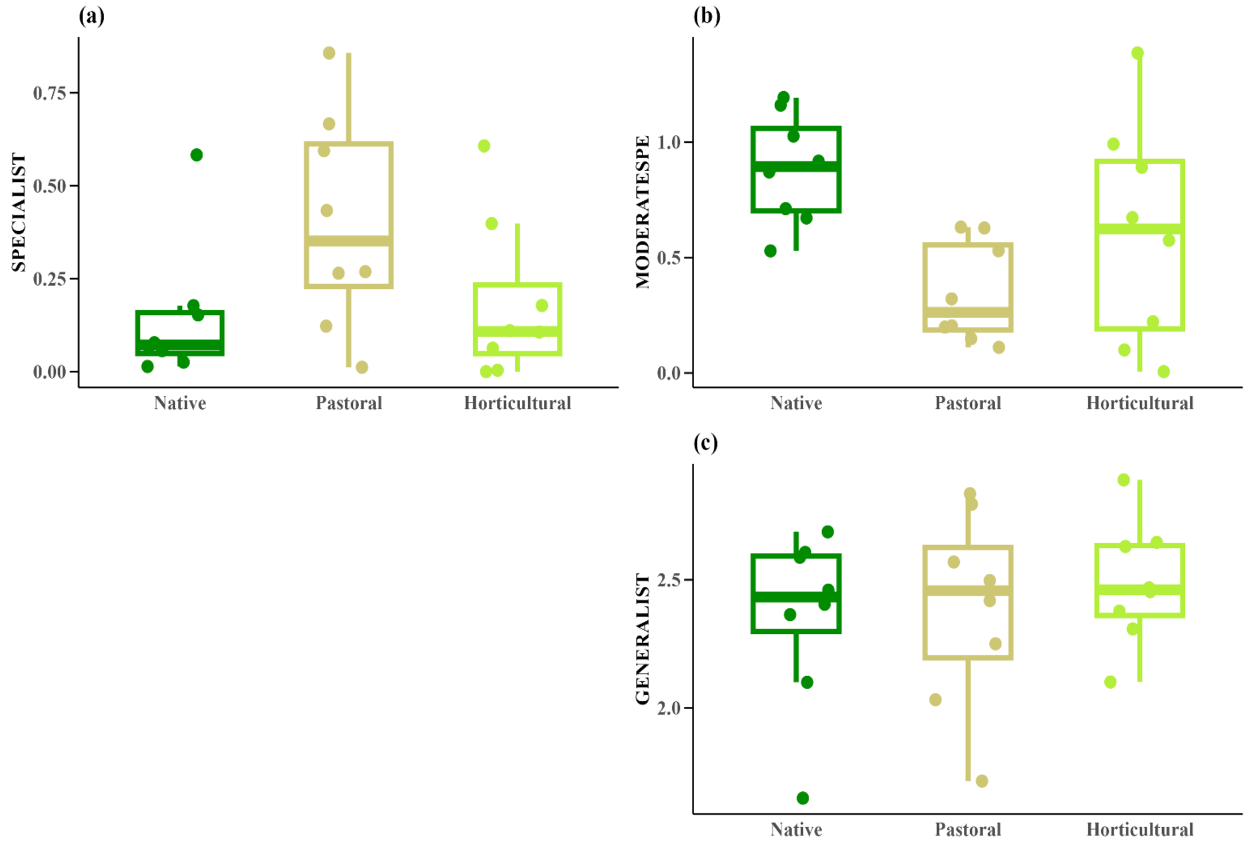

Figure 35. *Boxplots of the changes in the Community Weighted Mean (CWM) abundance of the modalities of the trait 'dietary preferences' across three different local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. a) specialist, b) moderatespe, c) generalist.*

Table 47. *ANOVA results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'dietary preferences' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Predictor**	**F-value**	**df**	**df.res**	**P-value**
specialist	Site type	6.23E-01	2	15.4	0.55
	Sediment	1.04E+01	1	20	**0.004**
moderatespe	Site type	2.54E+00	2	14.3	0.11
	Sediment	8.91E+00	1	18.8	**0.008**
generalist	Site type	1.34E+00	2	15.1	0.29
	Sediment	6.80E-01	1	19.9	0.42

4.1.5 The relationship between macroinvertebrate traits and sediment

4.1.5.1 Overview

Firstly, I have provided a summary of the key results for the trait modalities. Linear regression testing was conducted for the 58 trait modalities to analyze their relationship with Deposited Fine Sediment (DFS). The modality 'lda1' was excluded due to its absence in the macroinvertebrates recorded. Unless stated otherwise, the data for these analyses were derived from all 24 sites in the Te Awanui/Tauranga Harbor catchment.

The following trait modalities displayed a statistically significant relationship with DFS: 'desc1' ($P<0.01$; Figure 40a), 'desc4' ($P<0.01$; Figure 40d), 'univ' ($P<0.05$; Figure 41b), 'pluriv' ($P<0.05$; Figure 41c), 'lda2' ($P<0.05$; Figure 43a) 'lda4' ($P<0.05$; Figure 43c), 'two' ($P<0.05$; Figure 44c), 'submerged' ($P<0.01$; Figure 45b), 'eggfree' ($P<0.05$; Figure 46a), 'eggcement' ($P<0.05$; Figure 46b), 'attached' ($P<0.05$; Figure 49d), 'shredder' ($P<0.01$; Figure 50a), 'filterfeed' ($P<0.05$; Figure 50d), 'specialist' ($P<0.01$; Figure 51a), and 'moderatespe ($P<0.001$; Figure 51b).

The strength of the correlations (R^2) ranged from weak ('eggcement', $R^2=0.17$; Figure 46b) to moderate-strong ('specialist, $R^2=0.58$; Figure 51a). Other modalities with notable degrees of correlation include 'shredder' ($R^2=0.37$; Figure 50a) and 'moderatespe' ($R^2=0.43$; Figure 51b).

4.1.5.2 Specific trait modalities

Regarding the trait **'Maximum potential size'**, a total of five linear regression analyses were performed to investigate the relationship between the CWM abundances of its five modalities ('size1', 'size2', 'size3', 'size4' and 'size5') and DFS (Figure 36).

The first analysis revealed a weak negative correlation between the modality 'size1' and DFS that was not statistically significant ($P=0.22$; Figure 36a). 'size2' had a weak negative correlation with DFS that was not statistically significant ($P=0.75$; Figure 36b). The modality 'size3' had a neutral relationship with DFS that was not statistically significant ($P=0.91$; Figure 36c). 'size4' had a negative correlation with DFS that was not statistically significant ($P=0.01$; Figure 36d). 'size5' had a weakly positive relationship with DFS that was not statistically significant ($P=0.63$; Figure 36e).

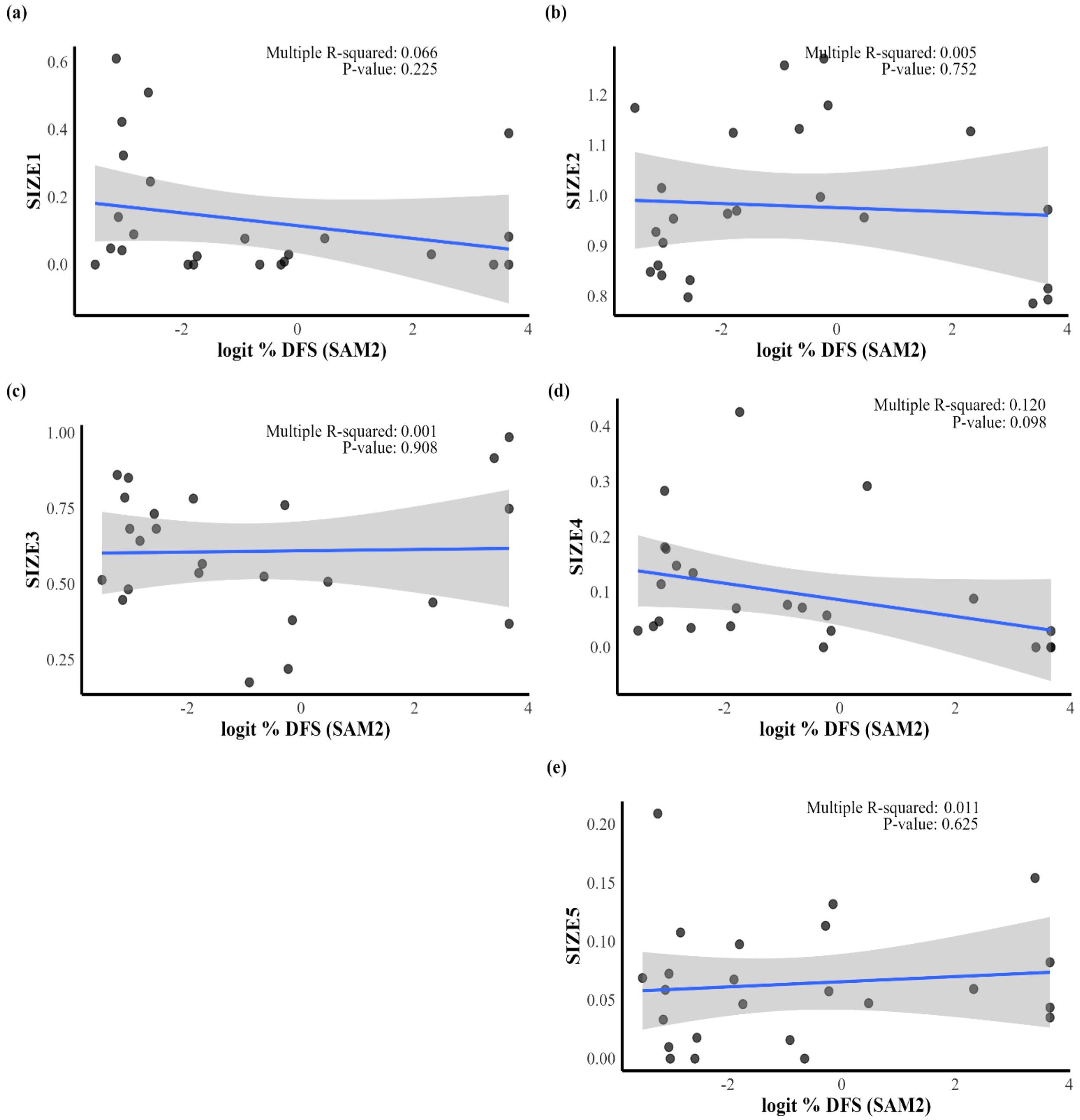

Figure 36. *The relationship between DFS and the modalities of the trait 'maximum potential size' in the Te Awanui/Tauranga Harbour basin. a) size1, b) size2, c) size3, d) size4, e) size4.*

Regarding the trait **‘Body flexibility’**, linear regression analyses were performed to investigate the relationship between the CWM abundances of three modalities (‘noflex’, ‘lowflex’, ‘highflex’) and DFS (Figure 37).

The first analysis revealed a weak postive correlation between the modality ‘noflex’ and DFS that was not statistically significant (*P*=0.56; Figure 37a). The weak negative correlation between ‘lowflex’ and DFS was not statistically significant (*P*=0.10) (Table 38b). The weak positive correlation between the modality ‘highflex’ and DFS was not statistically significant (*P*=0.24) (Table 38c).

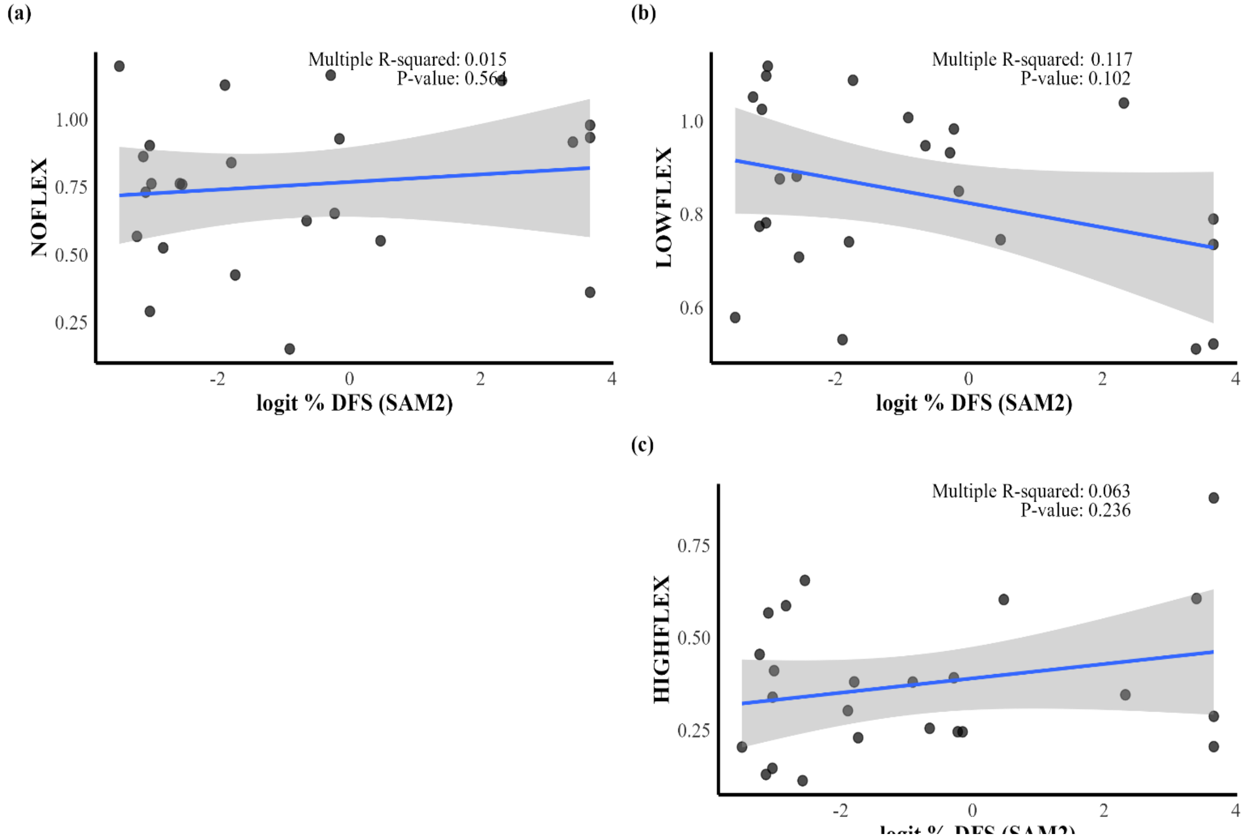

Figure 37. *The relationship between DFS and the modalities of the trait ‘body flexibility’ in the Te Awanui/Tauranga Harbour basin. a) a) noflex, b) lowflex, c) highflex.*

Regarding the trait **‘Body form’**, linear regression analyses were performed to investigate the relationship between the CWM abundances of its four modalities (‘flattened’, ‘cylindrical’, ‘spherical’, ‘streamlined’) and DFS (Figure 38).

The first analysis revealed a weak negative correlation between the modality ‘flattened’ and DFS that was not statistically significant (*P*=0.91; Figure 38a). The weak positive correlation between the ‘cylindrical’ and DFS was not statistically significant (*P*=0.84; Figure 38b). The weak negative correlation between ‘spherical’ and DFS was not statistically significant (*P*=0.12; Figure 38c). There was a weak positive correlation between the modality ‘streamlined’ and DFS that was not statistically significant (*P*>0.05; Figure 38d).

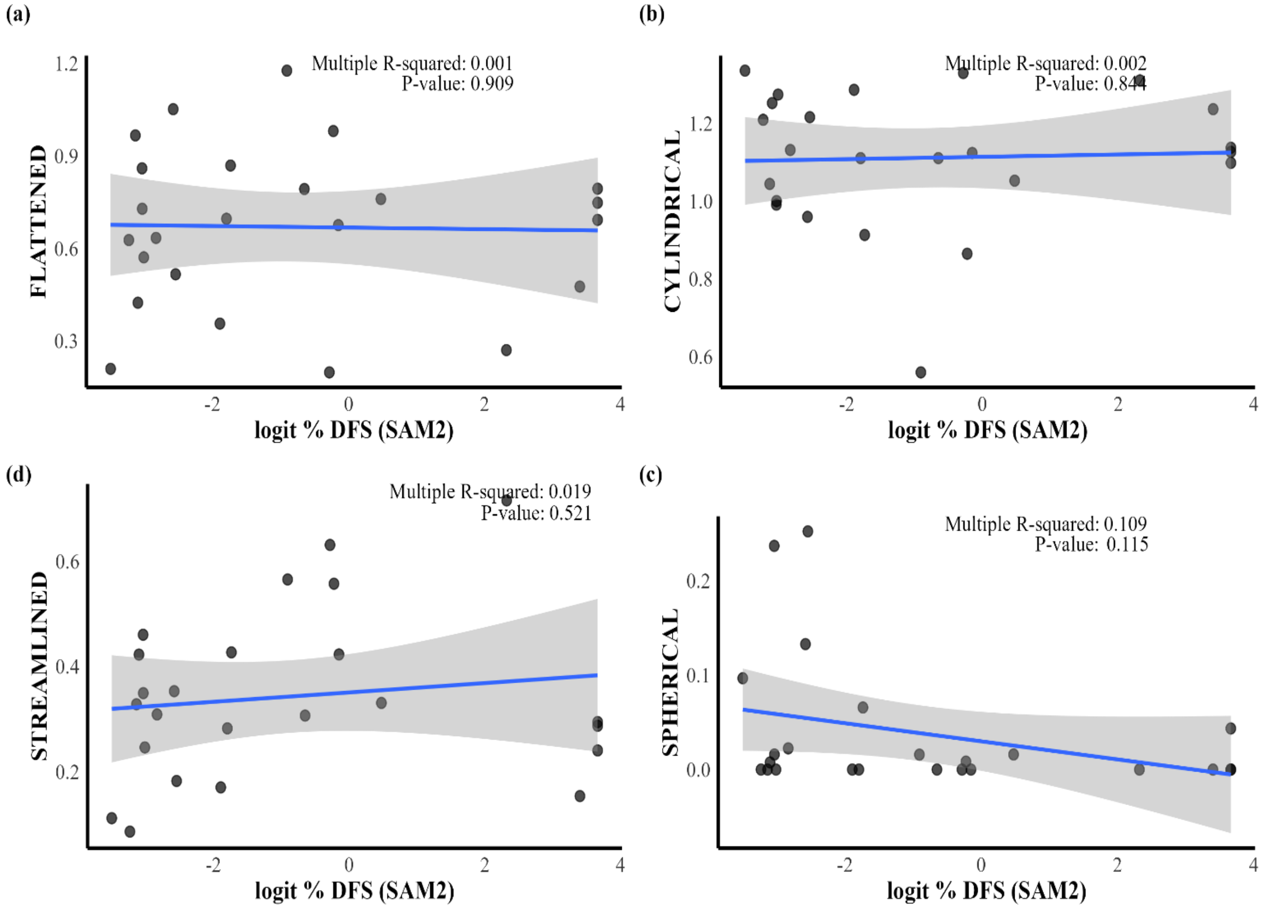

Figure 38. *The relationship between DFS and the modalities of the trait ‘body form’ in the Te Awanui/Tauranga Harbour basin. a) flattened, b) cylindrical, c) spherical, d) streamlined.*

Regarding the trait **‘Respiration of aquatic stages (not including eggs)’**, linear regression analyses were performed to investigate the relationship between the CWM abundances of its four modalities (‘tegument’, ‘gill’, ‘plastron’, ‘aerial’) and DFS (Figure 39).

The first analysis revealed a weak negative correlation between the modality ‘tegument’ and DFS that was not statistically significant (*P*=0.27; Figure 39a). There was a weak negative correlation between the modality ‘gill’ and DFS that was not statistically significant (*P*=0.71; Figure 39b). The modality ‘plastron’ had a weak positive correlation with DFS that was not statistically significant (*P*=0.40; Figure 39c). The weak negative correlation between the modality ‘aerial’ and DFS was not statistically significant (*P*=0.74; Figure 39d).

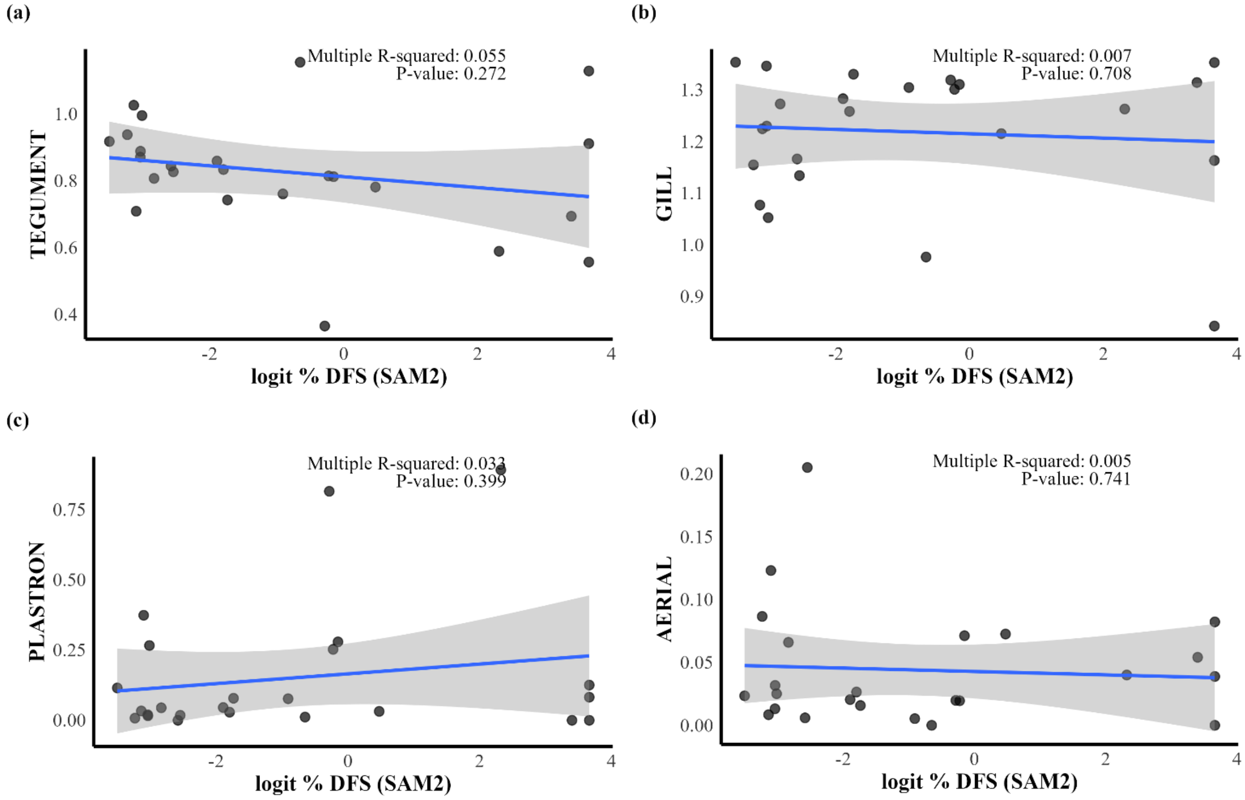

Figure 39. *The relationship between DFS and the trait ‘respiration of aquatic stages (not including eggs)’ modalities in the Te Awanui/Tauranga Harbour basin. a) tegument, b) gill, c) plastron, d) aerial.*

Regarding the trait **'Maximum number of descendants per reproductive cycle'**, linear regression analyses were performed to investigate the relationship between the CWM abundance of its four modalities ('desc1', 'desc2', 'desc3', 'desc4') and DFS (Figure 40).

The first analysis revealed a moderate-weak positive correlation (R^2=0.26) between the modality 'desc1' and DFS that was statistically significant (P<0.05; Figure 40a). The second analysis revealed a weak negative correlation between the modality 'desc2' and DFS that was not statistically significant (P=0.34; Figure 40b). There was a neutral relationship between the modality 'desc3' and DFS that was not statistically significant (P=0.96; Figure 40c). The modality 'desc4' had a moderate-weak negative correlation (R^2=0.26) with DFS that was statistically significant (P<0.05; Figure 40d).

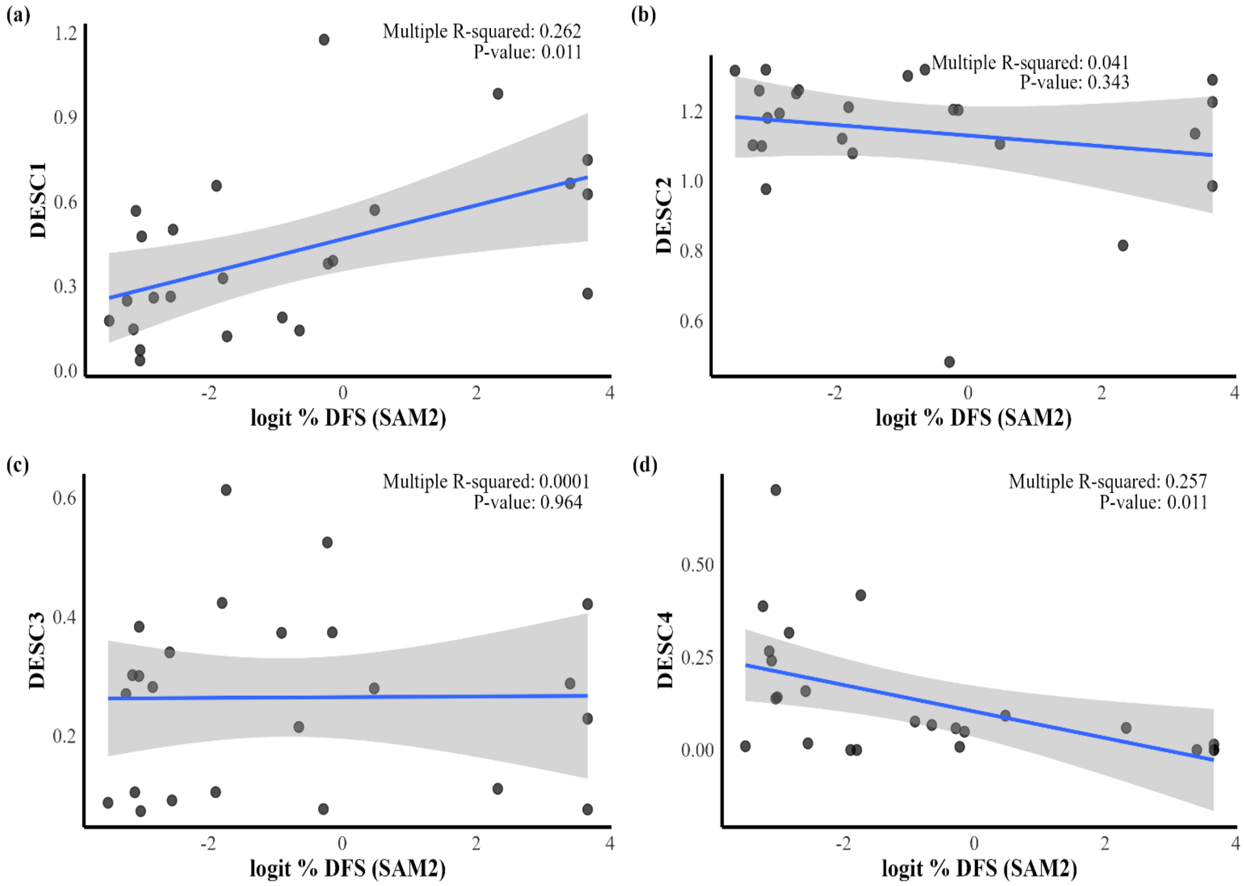

Figure 40*. The relationship between DFS and the modalities of the trait 'maximum number of descendants per reproductive cycle' in the Te Awanui/Tauranga Harbour basin. a) desc1, b) desc2, c) desc3, d) desc4.*

Regarding the trait **'Maximum number of reproductive cycles per year'**, linear regression analyses were performed to investigate the relationship between the CWM abundances of its three modalities ('semi', 'univ', 'pluriv') and DFS (Figure 41).

The first analysis revealed a weak negative correlation between the modality 'semi' and DFS that was not statistically significant (*P*=0.79; Figure 41). There was a moderate-weak negative correlation (R^2=0.21) between the modality 'univ' and DFS that was statistically significant (*P*<0.05; Figure 41b). The moderate-weak positive correlation (R^2=0.20) between the modality 'pluriv' and DFS was statistically significant (*P*<0.05; Figure 41c).

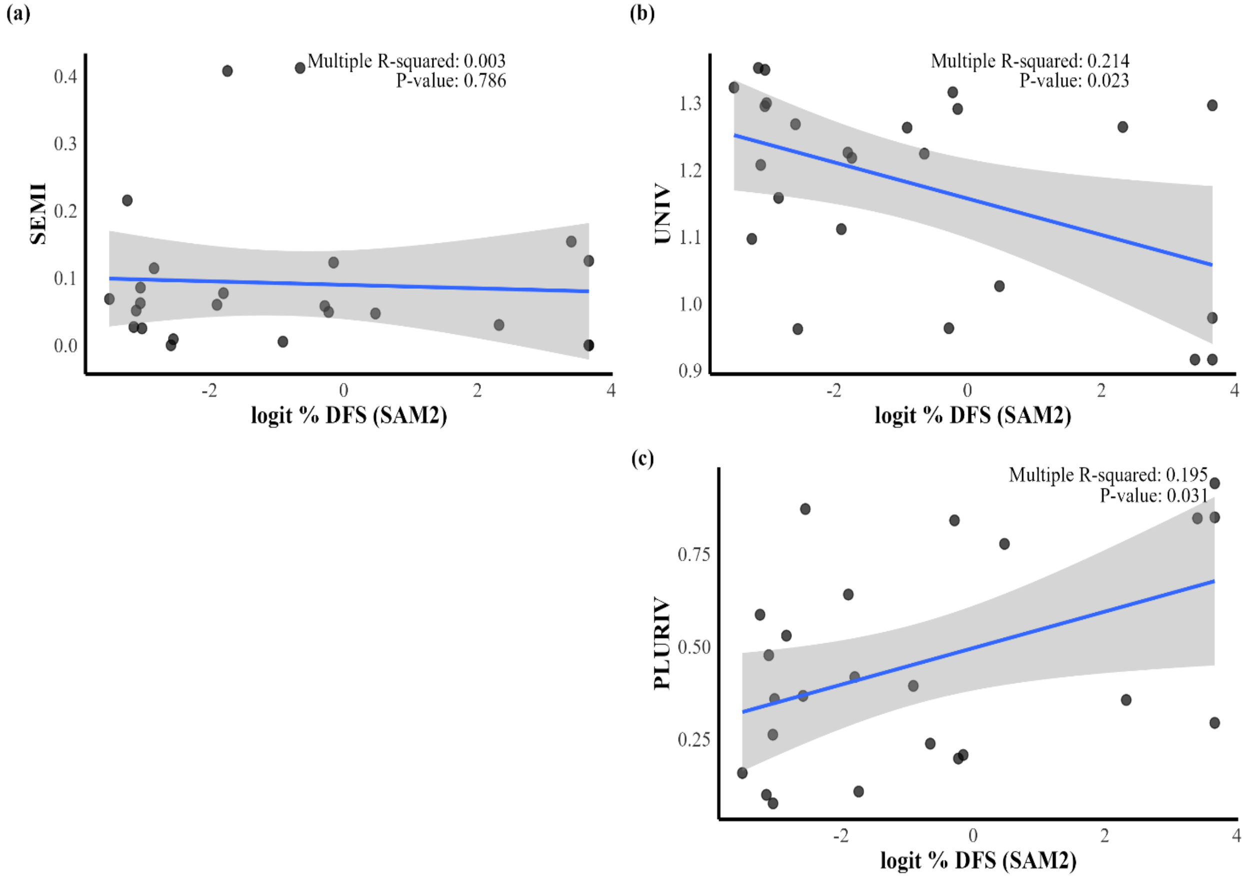

Figure 41. *The relationship between DFS and the modalities of the trait 'maximum number of reproductive cycles per year' in the Te Awanui/Tauranga Harbour basin. a) semi, b) univ, c) pluriv.*

Regarding the trait **'Maximum number of reproductive cycles per individual'**, linear regression analyses were performed to investigate the relationship between the CWM abundances of its two modalities ('cpi1', 'cpi2') and DFS (Figure 42).

The first analysis revealed a weak negative correlation between the modality 'cpi1' and DFS that was not statistically significant (*P*=0.14; Figure 42a). The weak positive correlation between the modality 'cpi2' and DFS was not statistically significant (*P*=0.09; Figure 42b).

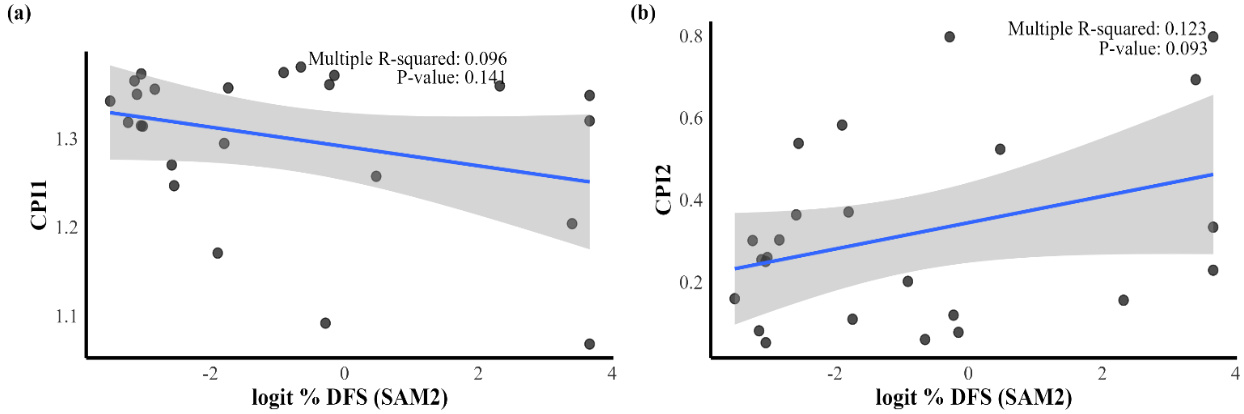

Figure 2. *The relationship between DFS and the modalities of the trait 'number of reproductive cycles per individual' in the Te Awanui/Tauranga Harbour basin. a) cpi1, b) cpi2.*

Regarding the trait **'life duration of adults'**, linear regression analyses were performed to investigate the relationship between the CWM abundances of four modalities ('lda2', 'lda3', 'lda4', 'lda5') and DFS (Figure 43). The modality 'lda1' was excluded as it was absent amongst the macroinvertebrate taxa recorded.

The first analysis revealed a moderate-weak negative correlation (R^2=0.17) between the modality 'lda2' and DFS that was statistically significant (*P*<0.05; Figure 43a). There was an almost non-existent correlation between 'lda3' and DFS that was not statistically significant (*P*=0.92; Figure 43b). The moderate-weak positive correlation (R^2=0.18) between 'lda4' and DFS was statistically significant (*P*<0.05; Figure 43c). There was weak positive correlation between 'lda5' and DFS that was not statistically significant (*P*=0.10; Figure 43d).

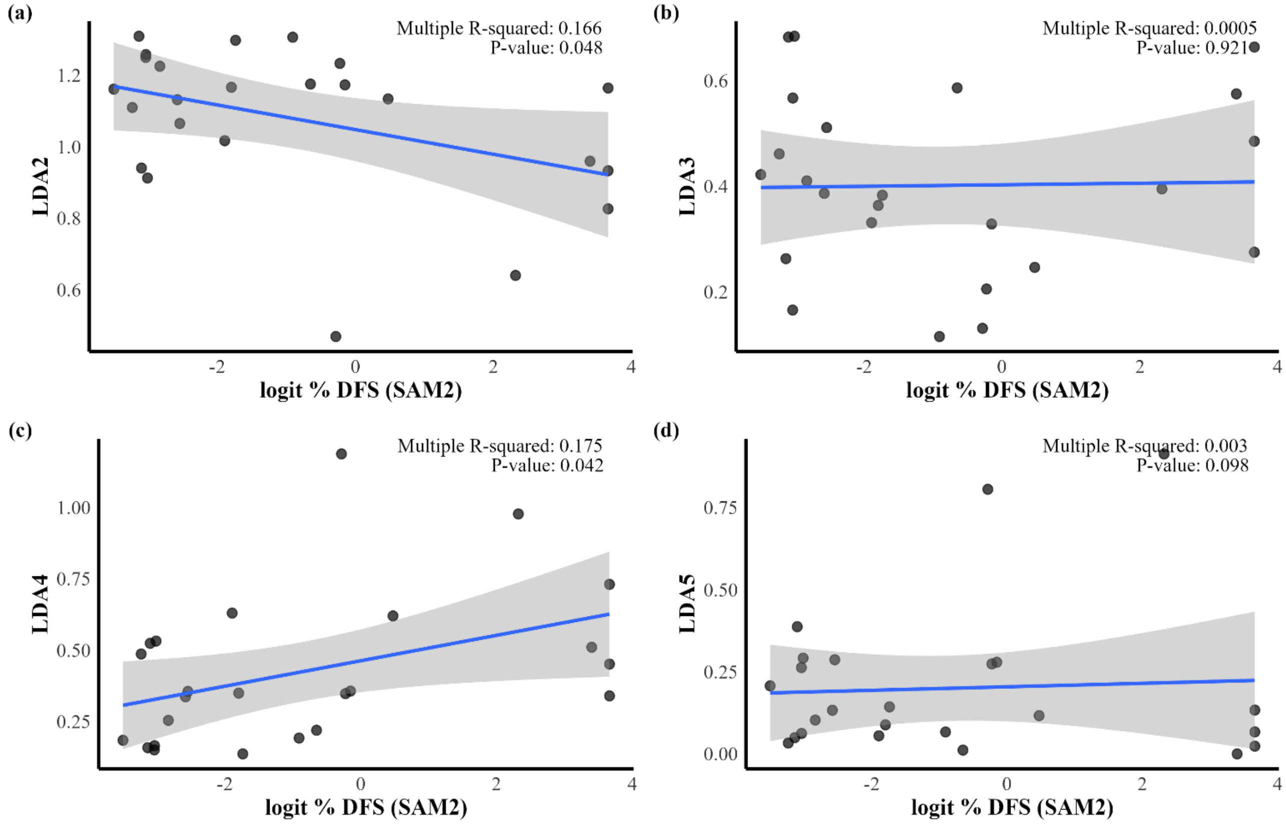

***Figure** 43. The relationship between DFS and the modalities of the trait 'life duration of adults' in the Te Awanui/Tauranga Harbour basin. a) lda2, b) lda3, c) lda4, d) lda5.*

Regarding the trait **'reproductive techniques'**, linear regression analyses were performed to investigate the relationship between the CWM abundances of its three modalities ('single', 'herma', 'two') and DFS (Figure 44).

The first analysis revealed a weak-moderate (R^2=0.15) positive correlation between the modality 'single' and DFS that was not statistically significant at α=0.05 (P=0.06; Figure 44a). The weak negative correlation between the modality 'herma' and DFS was not statistically significant (P=0.15; Figure 44b). The weak negative correlation between 'two' and DFS was not statistically significant (P=0.52; Figure 44c).

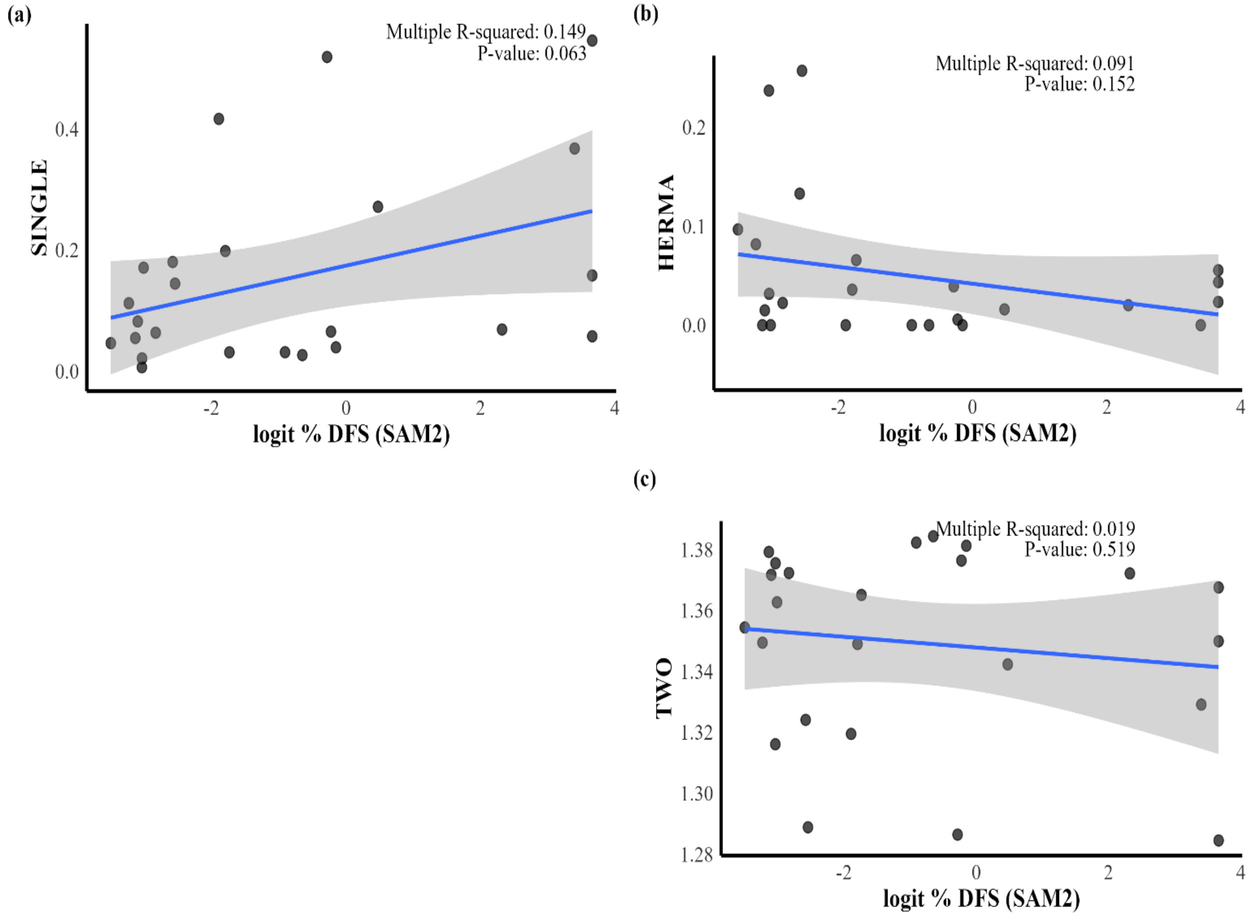

***Figure** 44. The relationship between DFS and the trait 'reproductive techniques' modalities in the Te Awanui/Tauranga Harbour basin. a) single, b) herma, c) two.*

Regarding the trait **'oviposition sites'**, linear regression analyses were performed to investigate the relationship between the CWM abundances of its three modalities ('surface', 'submerged', 'terrestrial', 'eggendo') and DFS (Figure 45).

The first analysis revealed a weak-moderate negative correlation (R^2=0.21) between the modality 'surface' and DFS that was statistically significant (P<0.05; Figure 45a). The moderate positive correlation (R^2=0.25) between 'submerged' and DFS that was statistically significant (P<0.05; Figure 45b). The weakly positive correlation between 'terrestrial' and DFS was not statistically significant (P=0.66; Figure 45c). The weak correlation between the modality 'eggendo' and DFS that was not statistically significant (P=0.32; Figure 45d).

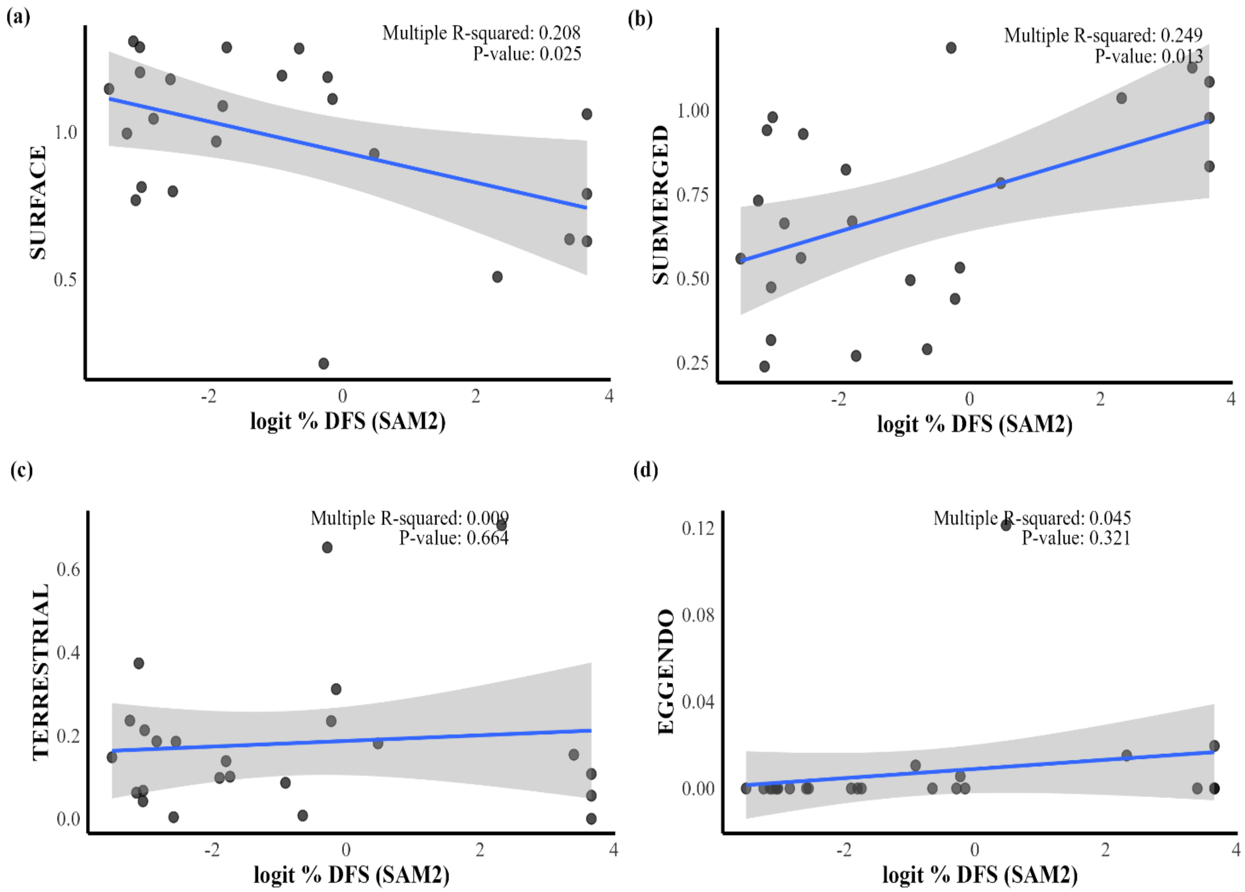

***Figure** 45. The relationship between DFS and the trait 'oviposition sites' modalities in the Te Awanui/Tauranga Harbour basin. a) surface, b) submerged, c) terrestrial, d) eggendo.*

Regarding the trait **'egg/egg mass'**, linear regression analyses were performed to investigate the relationship between the CWM abundances of its three modalities ('eggfree', 'eggcement', 'eggprotected') and DFS (Figure 46).

The first analysis revealed a moderate-weak negative correlation (R^2=0.18) between the modality 'eggfree' and DFS that was statistically significant (P<0.05; Figure 46a). The moderate-weak positive correlation (R^2=0.17) between the modality 'eggcement' and DFS was statistically significant (P<0.05; Figure 46b). The moderate-weak positive correlation (R^2=0.15) between the modality 'eggprotected' and DFS was not statistically significant at α=0.05 (P=0.06; Figure 46c).

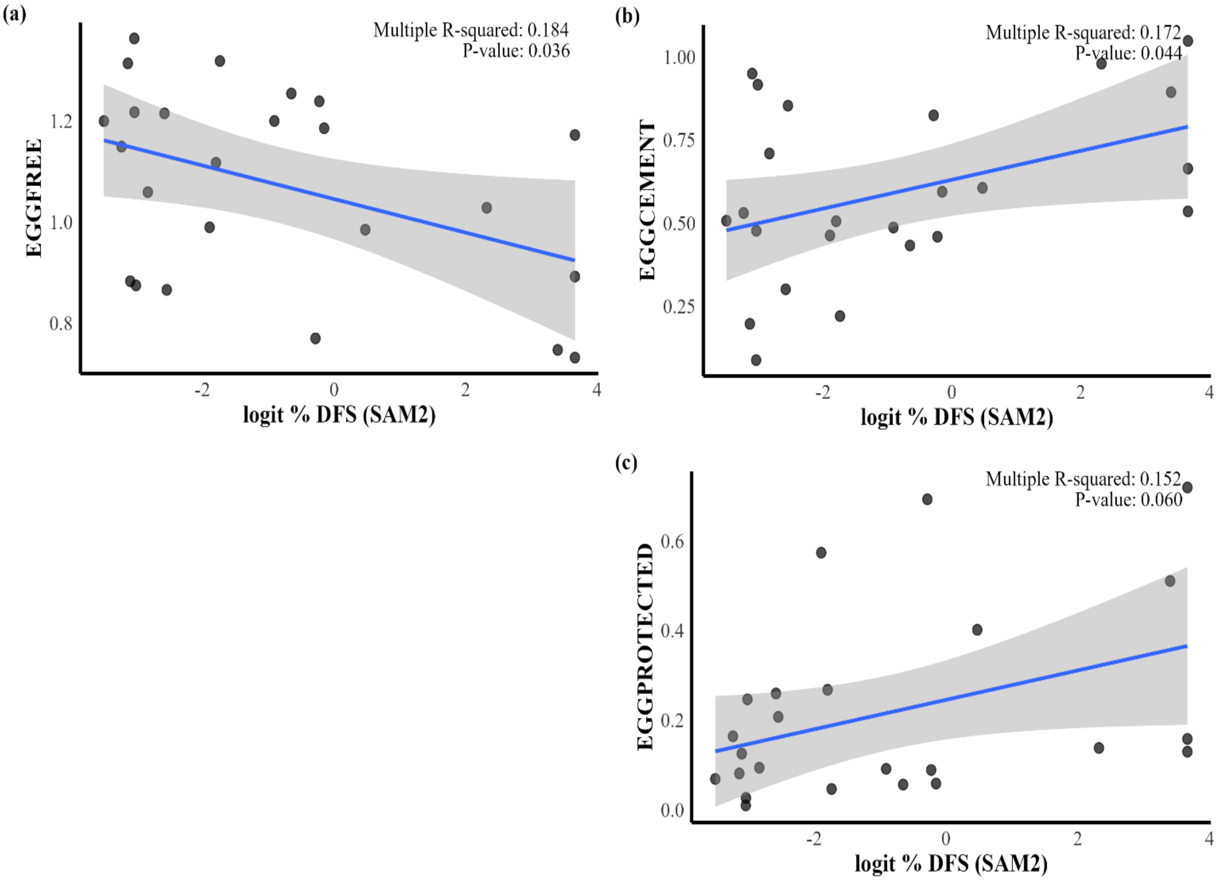

Figure 46. *The relationship between DFS and the trait 'egg/egg mass' modalities in the Te Awanui/Tauranga Harbour basin. a) eggfree, b) eggcement, c) eggprotected.*

Regarding the trait **'aquatic stages'**, linear regression analyses were performed to investigate the relationship between the CWM abundance of its three modalities ('aduandlar', 'aduorlar', 'larandpup') and DFS (Figure 47).

The first analysis revealed a weak positive correlation between the modality 'aduandlar' and DFS that was not statistically significant ($P<0.05$; Figure 47a). The weak-moderate negative correlation between the modality 'aduorlar' and DFS was not statistically significant (Figure 47b). The weak negative correlation between 'larandpup' and DFS was not statistically significant (Figure 47c).

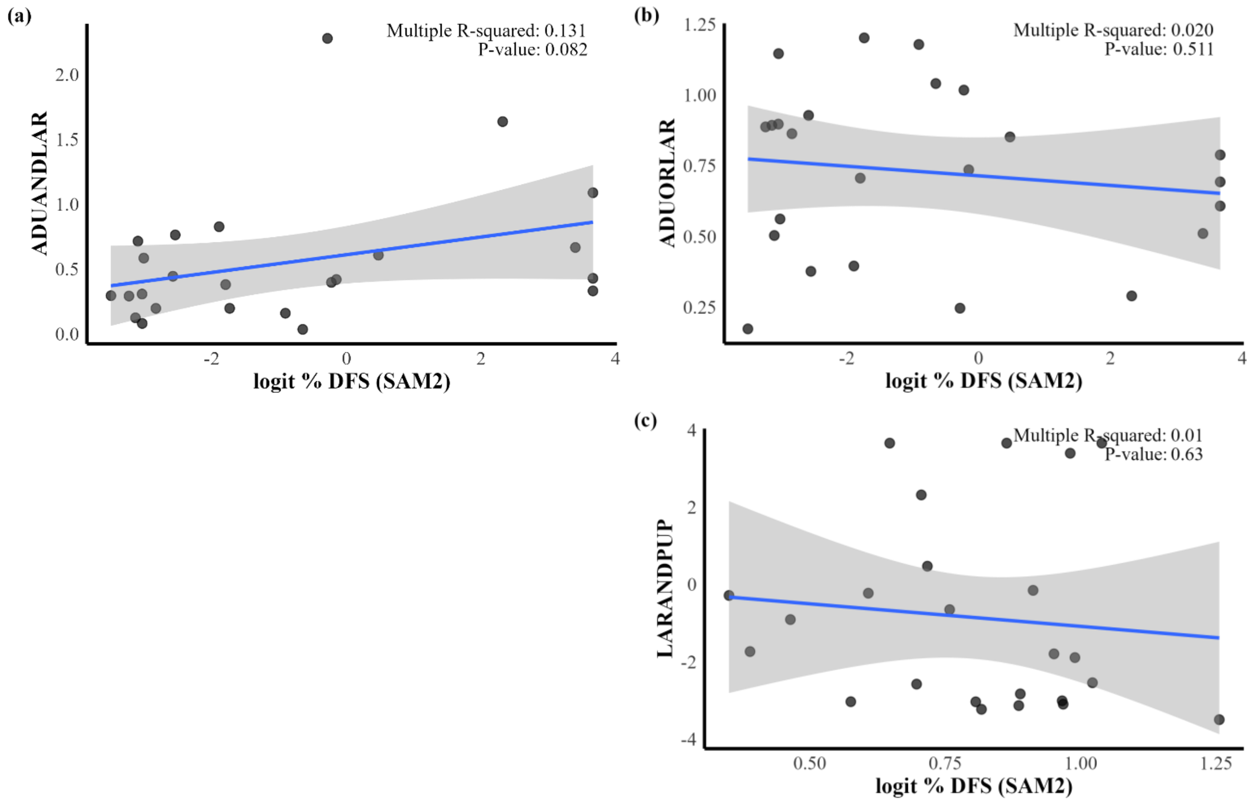

Figure 47*. The relationship between DFS and the trait 'egg/egg mass' modalities in the Te Awanui/Tauranga Harbour basin. a) aduandlar, b) aduorlar, c) larandpup.*

Regarding the trait **'dissemination potential (all stages)'**, linear regression analyses were performed to investigate the relationship between the CWM abundances of its three modalities ('disslow', 'dissmedium', 'disshigh') and DFS (Figure 48).

The first analysis revealed a weak negative correlation between the modality 'disslow' and DFS that was not statistically significant (*P*=0.13; Figure 48a). The weak negative correlation between 'dissmedium' and DFS was not statistically significant (*P*=0.30; Figure 48b). The weak positive correlation between 'disshigh' and DFS was not statistically significant (*P*=0.74; Figure 48c).

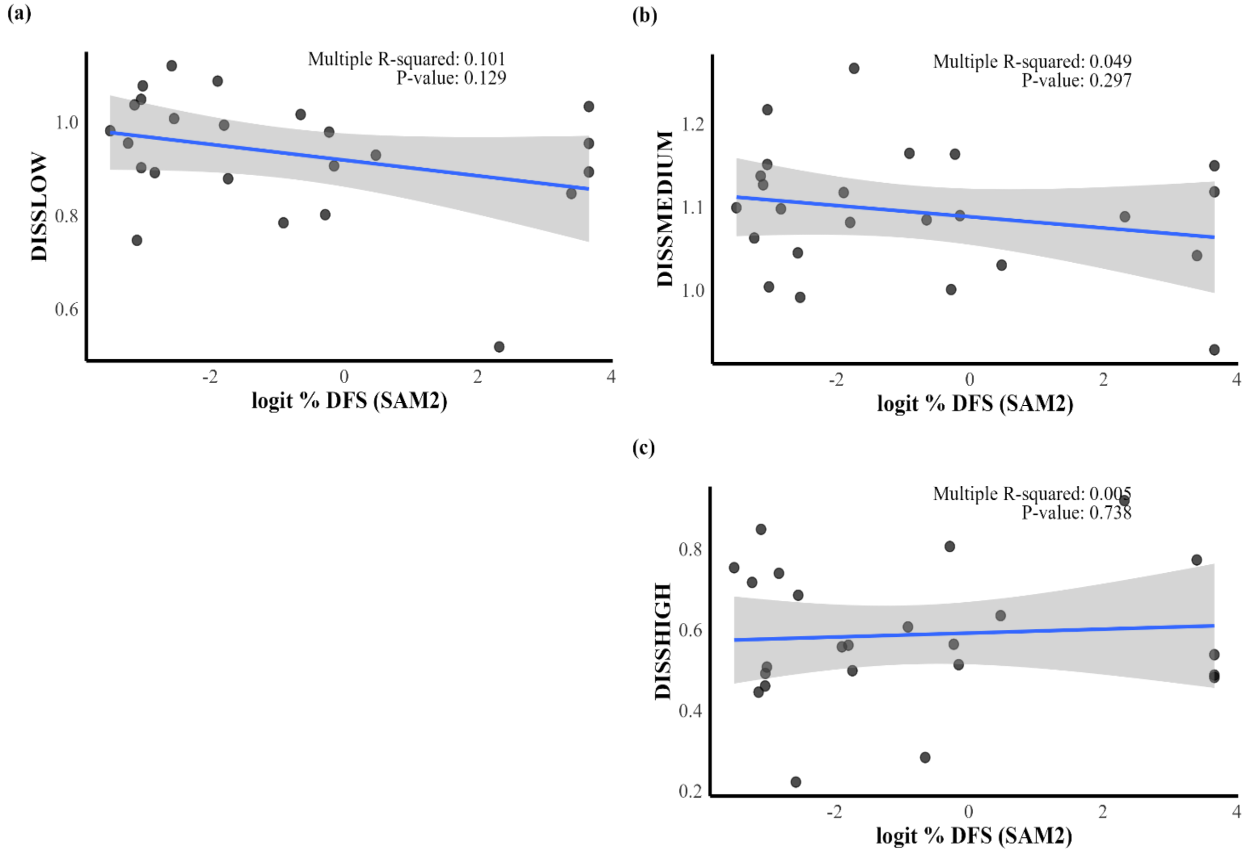

Figure *48. The relationship between DFS and the trait 'egg/egg mass' modalities in the Te Awanui/Tauranga Harbour basin a) disslow, b) dissmedium, c) disshigh.*

Regarding the trait **'attachment to substrate of aquatic stages (excluding eggs)'**, linear regression analyses were performed to investigate the relationship between the CWM abundances of its four modalities ('swimmer', 'crawler', 'burrower', 'attached) and DFS (Figure 49).

The first analysis revealed a weak negative correlation between the modality 'swimmer' and DFS that was not statistically significant (P=0.09; Figure 49a). The weak positive correlation between the modality 'crawler' and DFS was not statistically significant (P=0.50; Figure 49b). The weak positive correlation between the modality 'burrower' and DFS was not statistically significant (P=0.20; Figure 49c). The moderate negative correlation (R^2=0.23) between the modality 'attached' and DFS was statistically significant (P<0.05; Figure 49d).

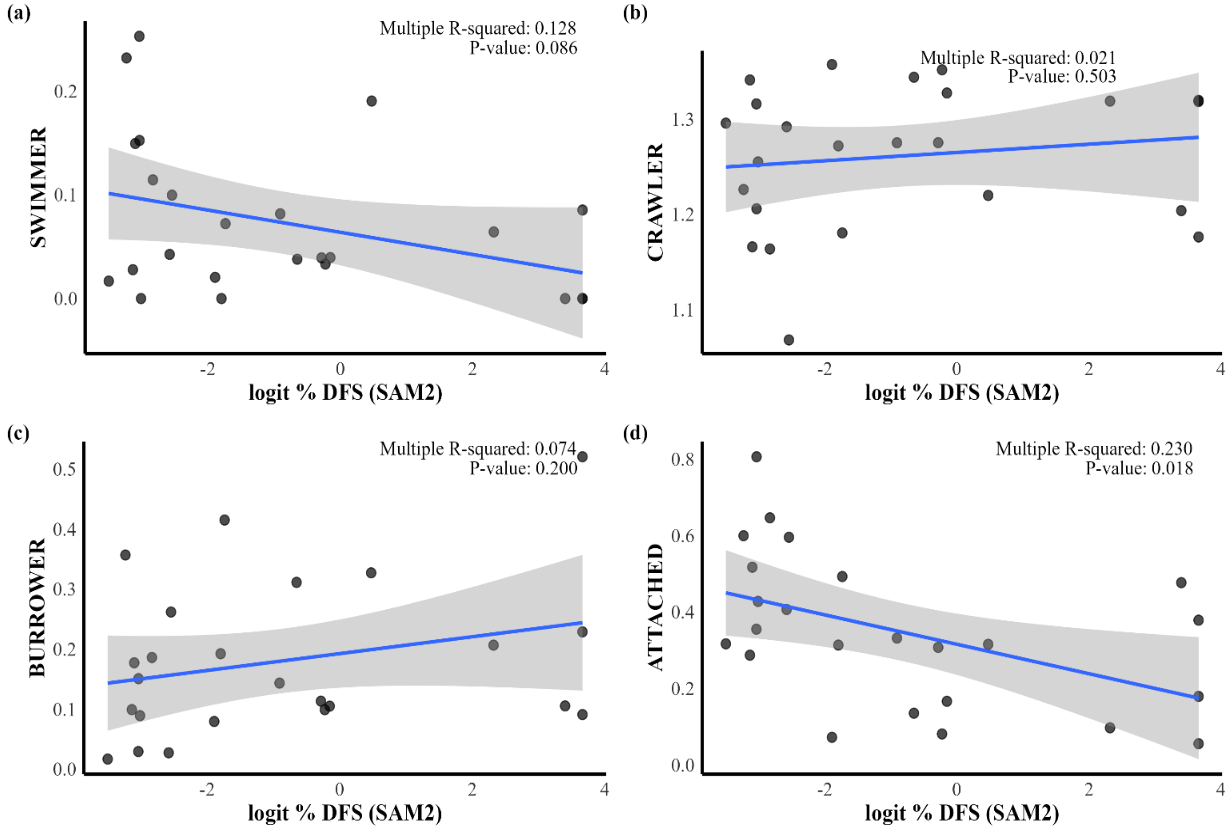

Figure 49. *The relationship between DFS and the trait 'egg/egg mass' modalities in the Te Awanui/Tauranga Harbour basin a) swimmer, b) crawler, c) burrower, d) attached.*

Regarding the trait **'feeding habits'**, linear regression analyses were performed to investigate the relationship between the CWM abundance of its six modalities ('shredder', 'scraper', 'deposit', 'filterfeed', 'predator', 'algalp') and DFS (Figure 50).

The first analysis revealed a moderate positive correlation (R^2=0.37) between the modality 'shredder' and DFS that was statistically significant (P<0.01; Figure 50a). The weak negative correlation between the modality 'scraper' and DFS was not statistically significant (P=0.32; Figure 50b). The weak correlation between the modality 'deposit' and DFS was not statistically significant (P=0.74; Figure 50c). The moderate-weak negative correlation (R^2=0.20) between the modality 'filterfeed' and DFS was statistically significant (P<0.05; Figure 50d). The modality 'predator' had a moderate-weak negative correlation (R^2=0.16) with DFS that not statistically significant at α=0.05 (P=0.06; Figure 50e). The modality 'algalp' had a weak positive relationship with DFS that was not statistically significant (P=0.53; Figure 50f).

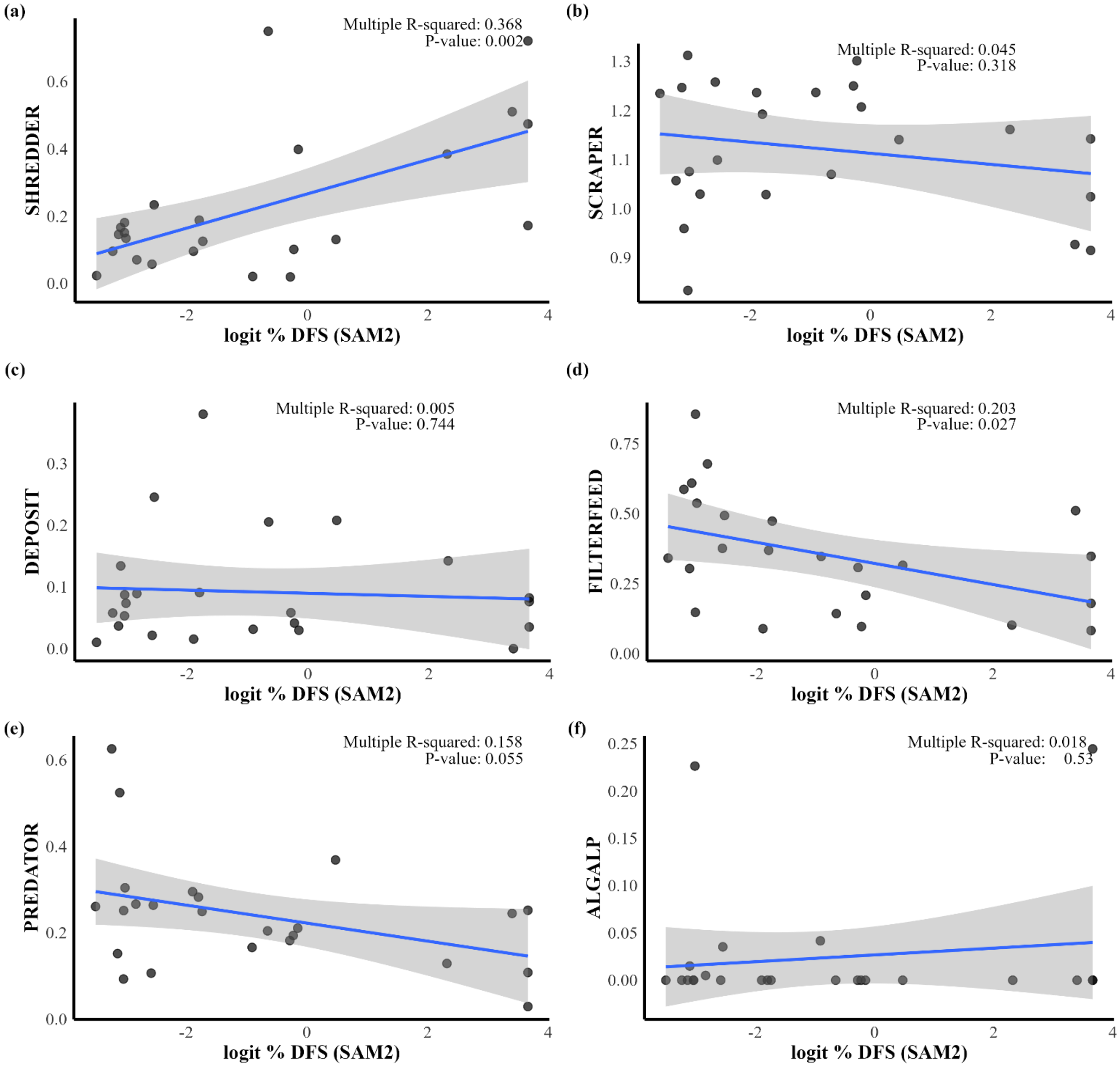

Figure 50. *The relationship between DFS and the modalities of the trait 'feeding habits' in the Te Awanui/Tauranga Harbour basin. a) shredder, b) scraper, c) deposit, d) filterfeed, e) predator, f) algalp.*

Regarding the trait **'dietary preferences'**, linear regression analyses were performed to investigate the relationship between the CWM abundances of its three modalities ('specialist', 'moderatespe', 'generalist') and DFS (Figure 51).

The first analysis revealed a moderate-strong positive correlation (R^2=0.58) between the modality 'specialist' and DFS that was statistically significant (P<0.001; Figure 51a). The moderate-strong negative correlation (R^2=0.44) between 'moderatespe' and DFS was also statistically significant (P<0.001; Figure 51b). The weak negative correlation between the modality 'generalist' and DFS was not statistically significant (P=0.16; Figure 51c).

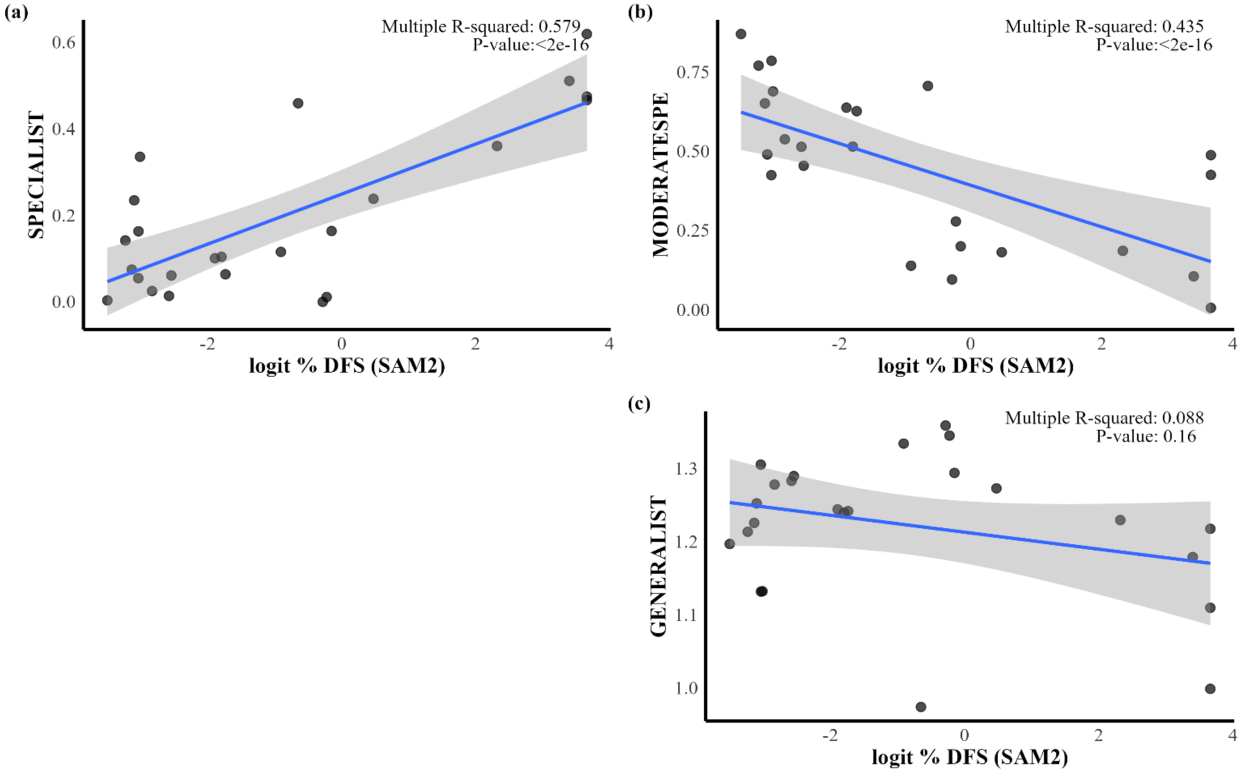

Figure 51. *The relationship between DFS and the trait 'dietary preferences' modalities in the Te Awanui/Tauranga Harbour basin. a) specialist, b) moderatespe, c) generalist.*

4.2 Discussion

4.2.1 Metrics of functional diversity

The functional diversity (FD) results in my thesis were surprising as pastoral streams did not have lower FD than the native forest streams, and values were often equal or greater in these human-affected sites. For instance, pastoral streams had the highest Rao's Q value, followed by native forests and then horticultural streams. Rao's Q is a frequently utilized metric due to its ability to gauge overall levels of FD (Schmera et al., 2017). Pastoral streams also had the highest number of traits (functional richness) and ecological niches filled (functional divergence). This indicates that pastoral streams had the highest level of overall FD, followed by native forest and then horticultural streams. The distribution of traits (functional evenness) and FD accounting for relative abundances (functional dispersion) were relatively similar in pastoral and horticultural streams but lowest in native forest streams. All these differences were not statistically significant. The unexpected discrepancies observed in my results could be attributed to unintentional biases in the sampling methods rather than indicating genuine differences in FD across land uses (Lee & Guénard, 2019), or may be a feature of New Zealand stream invertebrate communities.

The patterns observed in FD metrics across the three land-use types contradicted my expectations. Although they were statistically non-significant, these results were inconsistent with changes in FD in response to land use seen in previous studies (Ding et al., 2016). The most likely culprits for the equivocal results in my thesis are the choice of sampling tool (Correa-Araneda et al., 2021; Pinna et al., 2023; Tubić et al., 2017) and selection/representation of different habitats (Bradley & Ormerod, 2002; Stark et al., 2001); i.e., by using the National Environmental Monitoring Standards (NEMS) kicknet sampling protocol. It is well established that kicknet sampling has been shown to consistently yield higher species richness and abundance than Surber samplers (Murray, 2002; Stark et al., 2001; Stark, 1993; Tubić et al., 2017). This is because the kicknet method has a larger sample volume and area than Surber samplers (Stark, 1993). However, previous studies have indicated that although kicknet sampling generally collects a higher diversity and abundance of species, evidence suggests that Surber samplers may provide a more accurate representation of a stream's true macroinvertebrate community composition (Murray, 2002). Murray (2002) found the most accurate representation of a macroinvertebrate community was achieved with five randomly chosen Surber samples, compared to 30-second, 1-minute and 3-minute kicknet samples, along with individual stone samples. Furthermore, although the 3-minute kicknet samples collected the highest species diversity and abundance, the multivariate ordination analysis indicated it was the furthest away (least similar) from the total community (Murray, 2002). These findings suggest that although kicknet sampling is more efficient and collects a higher number and abundance of different species, Surber samplers are more successful in detecting the true community composition of a stream macroinvertebrate community (Murray, 2002). The 30-second and 1-minute kicknet samples were close to the total community, although lower than the 5-Surber sample method (Murray, 2002). This difference has been suggested due to the differing levels of intensity employed in kicknet and Surber sampling.

Although the number of species collected increases as the number of habitats sampled increases (Downes et al., 2000; Li et al., 2001), the two sampling methods differ in their levels of intensity (Correa-Araneda et al., 2021; Stark, 1993; Tubić et al., 2017). For instance, although kicknet samples are gathered over larger areas, Surber sampling may be more intensive due to its focus on smaller areas. These differing levels of sampling intensity are important to distinguish as previous studies indicate that Surber samplers are more successful in capturing sessile species, in comparison to kicknet sampling, which tends to capture a higher proportion of dislodged, highly-mobile taxa (Ghani et al., 2016). Surber sampling can capture a higher proportion of taxa that would otherwise be more difficult to obtain via kick-net sampling.

As macroinvertebrate community assemblages can differ quite differently between micro-habitats, such as those found in stony riffles in comparison to muddy habitats (Stark, 1993), it implies that Surber sampling is more effective at describing the taxa found in different substrate and habitat types (Ghani et al., 2016). This is important as rare taxa, although low in abundance, can be frequently found within habitat and substrate types that are more difficult to sample (Bradley & Ormerod, 2002). Rutt et al. (1989) found that stream margins contained a higher proportion of rarer species belonging to the orders Odonata and Coleoptera. Therefore, Surber sampling may be a more appropriate method to obtain species more representative of the true community assemblage (Murray, 2002), and thus, better describe FD metrics in response to sedimentation. Murray (2002) proposed that the difference in the ability of the two methods (kicknet vs Surber sampling) to detect the true community assemblage is determined more strongly by the level of sampling effort exerted rather than differences in the tools themselves.

Furthermore, in order for any sampling tool to be able to detect the true species assemblage of a macroinvertebrate community, it must first ensure that an adequate range of habitats representative of the stream must be sampled (Stark et al., 2001; Milne et al., 2022). For example, this study followed the NEMS kicknet protocol for macroinvertebrate sampling, which dictates that a maximum of eight-unit efforts (10 were taken in some instances due to low abundances) from different mesohabitats in proportion to their abundance to the sites reach (Milne et al., 2022). However, although the NEMS protocol dictates that all habitats comprising more than 5% of the streambed area to be sampled, those low in abundance or absent will be unintentionally excluded. This suggests that rarer taxa may have been unintentionally excluded due to the reduced sampling effort exerted upon them (Bradley & Ormerod, 2002; Narr & Krist, 2019). Additionally, sampling rare habitats such as backwaters with DFS levels disproportionate to the overall site would have likely distorted true functional diversity values.

It is important to appropriately select the range and number of habitats to sample, as the study by Narr & Krist (2019) found that if done so incorrectly, rare taxa can potentially be excluded regardless of the sampling method used (kicknet vs Surber sampling). For example, it found that commonly used methods to sample more commonly sought-after macroinvertebrates were ineffective in detecting freshwater gastropod species (Narr & Krist, 2019). This was due to the reduced focus or exclusion of habitat types preferred by gastropod taxa, such as floating algal mats, overhanging vegetation and submerged macrophytes (Narr & Krist, 2019).

The sampling methodology can have significant, although unintended, effects on FD metrics (Lee & Guénard, 2019). For example, in a terrestrial study on ants, Lee & Guénard (2019) found that the trait composition of ant communities differed significantly between pitfall and Winkler traps. Specifically, pitfall traps captured a higher proportion of larger-sized, slender and long-legged ants than those captured by Winkler traps (Lee & Guénard, 2019). This study highlights that the choice of sampling method can induce bias in FD metrics (Lee & Guénard, 2019).

Although the sampling methodology employed in this study aimed to sample macroinvertebrates from mesohabitats most representative of their community, it may have unintentionally caused bias. For example, it is possible that a higher proportion of functionally similar macroinvertebrates were captured due to the sampling of similar mesohabitats across different sites. In other words, since certain mesohabitats, such as riffles and runs, were more common and higher in abundance across sites, macroinvertebrates with traits complementary to those environments would have been unintentionally selected. Also, excluding other sampling methods, such as Surber sampling in favour of kicknet sampling, may have unintentionally selected less sessile species, further excluding species with different functional traits (Collier et al., 2014).

Future research should utilize sampling tools that capture a wider range of functionally different macroinvertebrate taxa. However, it is worth nothing that the kick-net is currently the best tool for macroinvertebrate collection as it can be used in any environment. Additionally, as highlighted by Murray (2002), kick-net samples yielded higher minimum, maximum and average numbers of taxa with the 3-minute kicknet having a substantially higher number of taxa. In comparison, Surber samples can only operate in very limited environments while kick-net samples can be collected from any environment. These tools should also target species known to be rare or more difficult to sample, as in the case of freshwater gastropods (Narr & Krist, 2019). Additionally, more units should be sampled from less abundant habitats with greater effort to detect rarer and or more sessile species. Although the purpose of the NEMS protocol is not necessarily intended for detecting FD, as its primary purpose is to detect the effects of a wide variety of stressors that can affect both habitat and water quality conditions, it could be adapted to detect changes in FD (Milne et al., 2022).

Nevertheless, regardless of the discrepancies and statistical insignificance of the metrics of FD reported in this study, they may still be ecologically important and are deemed worthy of further discussion. It is well-established in ecological theory that natural stream ecosystems generally support higher levels of functional diversity (Burdon et al., 2013; Ding et al., 2016; Kovalenko et al., 2014; Lu et al., 2022; Schmera et al., 2017). This has been attributed to their provision of a diverse range of complex habitats, including leaf litter, woody debris and riffles, which support a broad array of ecological niches (Burdon et al., 2013; Ding et al., 2016). Also, the environmental conditions of natural stream ecosystems are generally of high quality, providing adequate levels of oxygen, low or absent pollution levels, a stable flow regime, and an abundance of different food sources (Burdon et al., 2013; Ding et al., 2016).

Therefore, the notion that native forest streams would exhibit the highest levels of functional diversity, rather than pastoral streams, appears more plausible. Nonetheless, the observed findings might stem from a subsidy-stress response (Clapcott, Collier, et al., 2011; D. K. Niyogi et al., 2007). For instance, studies conducted by Townsend et al. (2008) and (Wagenhoff et al., 2011) revealed that initially, heightened nutrient levels and temperature favoured macroinvertebrates as they observed enhanced growth rates in response to increased algal growth rates and food availability. Furthermore, it was also observed that lower levels of Deposited Fine Sediment (DFS) provided additional habitat space for burrowing macroinvertebrates.

Additionally, the studies by Riley et al. (2003) and Thompson & Townsend (2003) reported that pastoral streams had higher macroinvertebrate species richness, leading to larger and more complex food webs than forested streams. Specifically, pastoral streams displayed heightened levels of connectance and a greater ratio of basal species to prey and predator species. Conversely, forested stream ecosystems exhibited a relatively simplified structure, often characterized by trophic food webs comprising a solitary resource and few primary consumers (Riley et al., 2003; Thompson & Townsend, 2003). These findings were linked to a subsidy stress response mechanism, in which elevated nutrient levels and increased temperatures accelerated algal growth rates, providing a strong autochthonous food base to support a larger and more complex food web (Riley et al., 2003; Thompson & Townsend, 2000).

These studies indicate that the heightened overall FD (e.g., Rao's Q) observed in pastoral streams could be attributed to a favourable 'subsidy' response along a subsidy-stress continuum. Additionally, the high levels of functional divergence observed in pastoral streams further support the potential occurrence of a subsidy-stress relationship with land use. This suggests that a broader range of ecological niches are inhabited by pastoral macroinvertebrate communities, indicating a diverse range of traits and potentially, a more complex trophic food web.

However, greater FD in pastoral streams does not necessarily imply better overall ecosystem health. This idea is highlighted in the study by Bonilla-Valencia et al. (2022), which cautioned against the use of FD metrics to determine the ecological health of an ecosystem if the species composition of the community is unknown. For instance, my study found evidence of the Functional Transformation Hypothesis, which predicts that anthropogenic disturbance can eliminate environmental and biotic filters, creating spatial and temporal niches for species with functional traits that differ from the original community (Bonilla-Valencia et al., 2022). Thus, when functional transformation occurs, species richness and FD may increase. This can be due to the establishment of invasive species possessing different and novel traits than the original community (Bonilla-Valencia et al., 2022). The study by Bonilla-Valencia et al. (2022) described how species richness and FD of a community increased due to the entry of invasive plant species, reinforcing that higher FD does not always mean better ecosystem health.

Similar processes to those described in the Functional Transformation Hypothesis (Bonilla-Valencia et al., 2022) likely explain the heightened values of FD metrics observed in my study. For instance, although Rao's Q and functional richness were highest in pastoral streams, those streams also have the lowest abundances and taxa richness of EPT taxa. This indicates that the heightened FD could likely be attributed towards anthropogenic disturbances creating spatial and temporal niches that allow for the entry of pollutant-tolerant taxa that have traits differing from their original undisturbed pollutant-intolerant community (Bonilla-Valencia et al., 2022; Myslinski & Ginsburg, 1977). In pastoral streams, pollutant-tolerant chironomid species with a Macroinvertebrate Community Index (MCI) score of 2 (Stark & Maxted, 2007) were the third most abundant taxa, and they may have possessed unique traits helping to increase FD at these sites.

Although the MCI and QMCI scores of pastoral and horticultural streams were significantly lower in comparison to native forest streams, it is notable that many pollutant-sensitive taxa remained among the ten most abundant taxa in these human-impacted streams. For example, these taxa included *Austroclima* (MCI score of 9), *Neozephlebia* (MCI score of 7), *Deleatidium* (MCI score of 8) and *Zephlebia* (MCI score of 7). Give the longitudinal distribution of land-uses, it is possible that the relatively high presence of these pollutant-tolerant taxa in these streams is due to mass effects via source-sink dynamics (Heino et al., 2015; Pulliam, 1988).

Heino et al. (2015) discusses how dispersal mechanisms change from low-order to high-order streams. Specifically, in low-order streams (particularly headwaters), species sorting is primarily dictated by environmental filtering, whereas mass effects are more prevalent in higher-order streams. Headwater streams are often isolated from each other as there can be a considerable spatial distance between them. Due to this isolation, the dispersal rate of migrants from local metacommunities will be low, meaning that the survival of the migrants will be dependent on their ability to pass through local environmental filters. This is because the dispersal rate of migrants from local metacommunities will be insufficient to replenish individuals who have died due to incompatible local environmental conditions (Heino et al., 2015).

In comparison, higher-order streams are typically more connected to headwaters and larger rivers, which allows for higher dispersal rates of migrants from local metacommunities due to higher levels of connectivity that can potentially overcome certain species dispersal limitations (Heino et al., 2015). This can allow species to inhabit and persist in environmentally incompatible conditions due to source-sink dynamics (Heino et al., 2015; Pulliam, 1988). Source populations are defined by birth rates higher than morality rates, whereas sink populations are defined by morality rates higher than birth rates (Pulliam, 1988). If the surplus of macroinvertebrates is high enough in source populations, it can compensate for the mortality of macroinvertebrate species in sink populations (Heino et al., 2015; Pulliam, 1988). This means that mass effects can maintain a population of pollutant-sensitive macroinvertebrate taxa in streams degraded by anthropogenic stressors (Heino et al., 2015; Pulliam, 1988).

Mass effects via source-sink dynamics (along with neutral processes) could explain the presence of pollutant-sensitive macroinvertebrate taxa (indicated by higher MCI scores). The median MCI score of pastoral streams (approximately 120) suggests that they may be improving(Bêche & Statzner, 2009; Mouchet et al., 2010; Pulliam, 1988; Stark & Maxted, 2007). Specifically, only three pastoral streams had MCI scores below a NOF band of B (Sites 13, 18, 17), whilst the remaining sites were above this NOF band. However, it should be noted that the results do not accurately represent each site's actual NOF bands. This is because NOF bands are determined from data spanning five years of annual sampling (Davis et al., 2022), whereas the results in my study were derived from a single sampling date.

Surprisingly, my results suggested horticultural streams appear to be more severely impacted than pastoral streams. Although horticultural streams had higher relative abundances and taxa richness of EPT taxa in comparison to pastoral streams, they still had the lowest MCI and QMCI scores. This discrepancy can be explained by the high relative abundance of lower-scoring EPT taxa in horticultural sites such as *Pycnocentrodes* (MCI score of 5)*, Aoteapsyche* (MCI score of 4), and *Triplectides* (MCI score of 5) (Stark & Maxted, 2007). Horticultural streams also had low FD with the lowest values for Rao's Q, functional richness and functional divergence.

This significant reduction could be due to functional homogenization, characterized by a reduction in functional diversity and an increase in species diversity (Bonilla-Valencia et al., 2022). Functional homogenization occurs when anthropogenic disturbances act as environmental filters that can eliminate species within the original community, allowing the establishment of species with similar functional traits and pollution tolerances (Bonilla-Valencia et al., 2022).

The low MCI and QMCI scores, combined with the indications of reduced FD, suggest that a pattern of functional homogenisation may be present in horticultural streams. Many pollutant-tolerant taxa such as *Potamopyrgus* (MCI score of 4), *Chironomidae* (MCI score of 2), and *Austrosimulium* (MCI score of 3) were among the ten most abundant species in horticultural streams. Although EPT relative abundances and taxa richness were higher in comparison to pastoral streams, my results suggest that anthropogenic stressors are causing horticultural stream macroinvertebrate communities to become homogenized by pollutant-tolerant taxa.

Despite some indications of functional homogenization, an apparent contradiction was observed in my study, as horticultural streams had the lowest taxa richness out of the three land uses. In contrast, species richness is predicted to increase under the Functional Transformation Hypothesis (Bonilla-Valencia et al., 2022). However, in my study there was no sampling over time, meaning it is unclear whether taxa richness in these communities increased or decreased in response to land-use change. Further, Piano et al. (2020) found evidence of functional homogenization occurring within macroinvertebrate communities with a reduction in species richness. In response to flow alterations, the community displayed a significant shift from taxa with a preference for aquatic respiration and medium to fast-flowing water to taxa that preferred air respiration and lentic habitats, along with having a larval dormancy phase that protected them from low stream flows (Piano et al., 2020). This indicates that heightened species richness is not always characteristic of functional homogenization (Bonilla-Valencia et al., 2022; Piano et al., 2020).

Interestingly, although native forest streams had the second highest values out of the three land-use types for Rao’s Q, functional richness and divergence, they also had significantly higher biotic indices scores for the ASPM, MCI and QMCI metrics, the highest EPT relative abundances and taxa richness, along with the highest overall abundances and taxa diversity. The combination of reduced levels of FD yet high taxa richness displayed in native forest streams indicate that these communities have high levels of functional redundancy (Schmera et al., 2017).

With the heightened taxa diversity in native forest streams, functional diversity should also be higher. Many studies, such as those done by Heino (2008), Bêche & Statzner (2009) and Schmera et al. (2017), show a positive relationship between functional diversity and species diversity, with the study by Bêche & Statzner (2009) displaying a positive and saturating relationship. However, the paper by Reynaga (2013) showed that functional diversity can vary independently of taxonomic diversity. This point was further highlighted in the paper by Cadotte et al. (2011), which attributed this finding to high levels of functional redundancy.

Functional redundancy occurs when there is a high number of species that have the same trait and, therefore, perform the same ecological function (Bêche & Statzner, 2009; Schmera et al., 2017). In order for functional diversity to increase with species diversity, there must be a high number of species with unique traits (Schmera et al., 2017). However, there is a limit to the number of traits that could occur within an ecosystem, as highlighted by Poff et al. (2006), due to trait syndromes. Trait syndromes explain how multiple traits an organism possesses are typically linked together due to evolutionary, ecological or physiological reasons (Poff et al., 2006; Schmera et al., 2017). Therefore, there is a limit to the number of traits a single organism can possess (Poff et al., 2006; Schmera et al., 2017).

Functional redundancy typically occurs when organisms such as macroinvertebrates are subjected to high degrees of environmental pressures that act as abiotic filters, only allowing species with compatible traits to establish (Bêche & Statzner, 2009). Thus, this limits the number of traits present within a community, leading to an abundance of organisms with the same or similar traits (Bêche & Statzner, 2009). For example, the study (Bêche & Statzner, 2009) found that all functional diversity and species richness metrics were strongly influenced by environmental conditions such as fine substrate, conductivity, stream width, land use and precipitation.

Additionally, the low levels of functional evenness, divergence and dispersion present in native forest streams indicate a strong preference towards specific traits, providing additional evidence of functional redundancy (Schmera et al., 2017). As native forest streams are not affected by the same stressors as those adjacent to human land uses, changes in the representation of functional traits are more strongly influenced by natural environmental conditions (Bêche & Statzner, 2009; Dolédec et al., 2011). Furthermore, the study (Statzner & BÊChe, 2010) showed that natural environmental gradients strongly influenced the distribution of functional feeding groups in pristine stream ecosystems.

These studies align with the River Continuum Concept (RCC) proposed by Vannote et al. (1980), which describes the changes in functional feeding groups (FFG) throughout the longitudinal profile of a river/stream as a result of environmental gradients, such as the input of organic matter. For example, according to the RCC by Vannote et al. (1980), stream/river ecosystems transition from small, narrow and shaded headwaters with Coarse Particulate Organic Matter (CPOM) inputs and high nutrient uptake rates to lower reaches which are wider and have increased light availability, higher temperatures and reliance on Fine Particulate Organic Matter (FPOM) inputs and algal productivity. The change in environmental conditions and the volume and type of organic matter along the longitudinal profile of a river creates a gradient that strongly influences the distribution of macroinvertebrates belonging to different FFGs (Vannote et al., 1980).

According to the RCC, macroinvertebrates belonging to the shredder FFG are more prevalent in headwaters due to high CPOM inputs via allochthonous inputs. The mid-reach of a river is predicted to be dominated by macroinvertebrates belonging to the grazer/browser and algal piecer FFGs. Finally, the lower reaches of a river are predicted to be dominated by macroinvertebrates belonging to the filter-feeder and predator FFG (Vannote et al., 1980). However, New Zealand native forest stream ecosystems may display significant differences from their North American counterparts which the RCC was developed upon (Thompson & Townsend, 2000; Vannote et al., 1980; Winterbourn et al., 1981). The distribution of FFGs throughout the longitudinal profile of a stream from the headwaters to the lower reaches in New Zealand differs significantly from macroinvertebrate communities found internationally (Thompson & Townsend, 2000; Winterbourn et al., 1981). Winterbourn et al. (1981) describe that the representation of FFGs do not differ significantly across the natural longitudinal profile of a stream in New Zealand. Furthermore, Winterbourn et al. (1981) also found an absence of a temporal replacement or change of species in response to seasonal changes in organic matter inputs.

These differences were directly attributed to the climatically unpredictable nature of stream ecosystems and the evergreen nature of native tree species in New Zealand (Winterbourn et al., 1981). Due to the evergreen nature of New Zealand native tree species, inputs of allochthonous organic matter into streams are low and irregular. As a result, the prevalence of macroinvertebrates belonging to the shredder functional group is significantly lower in New Zealand streams than those found internationally (Winterbourn et al., 1981). This is due to the lack of CPOM in the form of allochthonous organic matter, which shredder macroinvertebrates depend upon (Winterbourn et al., 1981). New Zealand macroinvertebrates may mainly comprise of those belonging to the browsing FFG with a generalist dietary preference, displaying ecological flexibility to the unpredictable inputs of food sources via allochthonous organic matter. It was explained that the prevalence of macroinvertebrates in New Zealand with these feeding traits is due to the high input of FPOM, mediated primarily via external inputs (wind blow, surface runoff, and bank erosion) and through the decomposition of organic matter by microbial organisms such as bacteria and fungi (Winterbourn et al., 1981).

This theory appears to be supported in my study, as highlighted by the relatively low CWM abundances of shredders within native forest streams along with the higher CWM abundance of scrapers, deposit-feeders, and trophic generalists. Interestingly, the CWM abundances of filter feeders were high and comparable to scrapers in the native forest streams. Although filter feeders can be relatively low in abundance in New Zealand native forest streams, the high abundances of the mayfly species *Coloburiscus*, known to be a filter feeder, could explain this discrepancy (Winterbourn, 1978).

Nevertheless, regardless of the unpredictable nature of the environmental conditions in New Zealand's native forest streams, they remain stable. This is due to the highly flexible nature of New Zealand macroinvertebrate communities, which display high levels of functional redundancy demonstrated by the prevalence of browsing, trophic generalist macroinvertebrates in response to unreliable sources of CPOM via allochthonous organic matter (Bêche & Statzner, 2009; Winterbourn, 1978; Winterbourn et al., 1981).

It is also possible that the functional flexibility characteristic of New Zealand macroinvertebrates could provide another explanation for the statistical insignificance of all functional diversity metrics reported in this study. For example, the CWM trait abundance of the shredder and generalist modalities were not just high in native forest streams but relatively high in pastoral and horticultural streams in comparison to the other modalities associated with the traits 'Feeding habits' and 'Dietary preferences'. The functional similarity of New Zealand macroinvertebrates might explain why there were no statistically significant differences between communities in different land uses in this study.

However, as discussed previously, it is also possible that the sampling methods were inappropriate for describing the relationship between macroinvertebrate FD and the environment. The sampling protocol used potentially did not reflect the true diversity and abundances of macroinvertebrates at the different site types. For instance, macroinvertebrates adapted to living in highly polluted environments might be poorly resolved by the NEMS protocol, which relies on relatively coarse taxonomy for certain taxa groups. Different sampling methods can also have biases. For example, the study by Correa-Araneda et al. (2021) was able to clearly distinguish between macroinvertebrate communities collected by hand nets and sediment corers. This was due to the high abundance of *Chironomidae* and *Oligochaeta* taxa found in sampled sediment cores, as these taxa thrive in highly sedimented environments due to their natural life cycle (Correa-Araneda et al., 2021).

Nevertheless, my study did not find strong evidence of sedimentation influencing the functional diversity of macroinvertebrate communities sampled in this study, as the majority of the relationships were not statistically significant. For instance, there was a relatively weak but statistically significant positive relationship observed between functional divergence ($P<0.05$) and fine sediment. This result indicates that the sampled macroinvertebrate communities may have exhibited a broad range of adaptations (functional traits) in response to rising sediment levels. This response will be further explored in the following sections.

4.2.2 ***Trait relationships with sediment***

The study by Poff (1997) discusses the importance of environmental filters in determining the assemblage of stream ecosystems, as species must be able to 'pass through these filters in order to join a local community'. What allows these species to pass through these initial environmental filters is their ecological traits or, in other words, their habitat preferences (Dolédec et al., 1999; Usseglio-Polatera et al., 2000a). However, in response to pollution, many stream macroinvertebrates are lost due to adverse environmental conditions in their habitats. The species that remain do so due to possessing biological traits that allow them to withstand and overcome the pressure placed upon them by degraded conditions (Usseglio-Polatera et al., 2000b).

This means that biological traits are better predictors of anthropogenic stressors rather than ecological traits (Dolédec et al., 1999; Menezes et al., 2010). For instance, the study by Dolédec et al. (1999) employed both biological and ecological traits and found that biological traits were the best predictor of anthropogenic stressors, as they were not confounded by naturally occurring spatial gradients. Biological traits can distinguish the difference between changes in macroinvertebrate community composition caused by anthropogenic stressors, and those because of naturally occurring environmental conditions and/or spatial gradients.

Numerous New Zealand studies have employed macroinvertebrate biological traits to assess the effects of anthropogenic stressors. The studies most relevant to this study pertain to those investigating the effects of anthropogenic land use (particularly pastoral land use) and deposited fine sediment (Clapcott et al., 2017; Dolédec et al., 2011; Lange et al., 2014; Phillips & Reid, 2012b; Phillips & Smith, 2018; Townsend et al., 1997; Townsend et al., 2008). These studies are highly relevant and will be used to compare changes in biological traits in response to land use and sedimentation. There are a large number of modalities that could be potentially affected by both human land uses and sediment. Only those that have been identified in previous studies as having a strong response to these two stressors will be discussed.

The comprehensive analysis by Clapcott et al. (2017) investigated the use of biological functional traits as biomonitoring tool under the National Policy Statement for Freshwater Management

(NPS-FM) and identified a number of modalities that respond strongly to sediment: 'submerged', 'crawler', 'semi', 'pluriv', 'size4', size5', 'scraper', 'crawler', 'burrower' and 'surface'. Of these traits, the four modalities with the strongest response to sediment were 'submerged', 'crawler', 'plurivoltine', and 'size1' (Clapcott et al., 2017). A New Zealand study conducted by Dolédec et al. (2011) found that the modalities 'submerged' and 'eggfree' strongly responded to sediment. The biological trait analysis conducted by the Auckland Regional Council (ARC) identified the following two modalities as having a strong response to sediment: 'surface' and 'eggprotected' (Phillips & Reid, 2012a). The analysis by the ARC also identified the modalities 'surface' and 'eggprotected' as having strong responses to sediment, along with 'eggfree', 'crawlers', and 'swimmers'. The South Island study by Lange et al. (2014) identified a large number of modalities that responded to sediment, which was attributed towards the smothering and infilling of substrate. According to these studies combined, modalities that have the strongest response to sediment are 'submerged', 'crawler', 'plurivoltine', 'size1', 'surface' and 'eggprotected' (Clapcott et al., 2017; Dolédec et al., 2011; Philips & Smith, 2012a; Phillips & Reid, 2012b). Modalities that had a statistically significant response to sediment ($P<0.05$) in my study and the previously cited studies were 'submerged', 'crawler', 'surface', 'eggfree', 'swimmer', and 'plurivoltine'.

Traits provide a mechanistic understanding of how stressors affect stream macroinvertebrate communities (Menezes et al., 2010), making it important to discuss the specific interaction between the modalities mentioned above in response to sediment. For instance, the modalities 'submerged', 'eggfree', and 'plurivoltine' all increased with greater levels of deposited fine sediment in my study and the others referred to above (Clapcott et al., 2017; Dolédec et al., 2011; Lange et al., 2014; Philips & Smith, 2012a; Phillips & Reid, 2012b). In contrast, the modality 'surface' decreased as sediment levels increased. The explanation behind this pattern was that eggs laid at the water's surface were more susceptible to being smothered by sediment than those laid below the water ('submerged') or remained attached to their parent were less susceptible (Conroy et al., 2018). In the studies mentioned above and my study, the modality 'plurivoltine' increased in response with greater levels of deposited fine sediment. This is because the ability to reproduce multiple times a year is favoured as it confers higher levels of resilience in response to sediment (Dolédec et al., 2011).

Interestingly, the modality 'crawler' decreased as sediment levels rose in the other studies (Clapcott et al., 2017), but slightly increased as sediment levels increased in my study. The modality 'crawler' should decrease as sediment levels increased due to the infilling of interstitial spaces between substrates, which served as a primary habitat for these macroinvertebrates (Pollard & Yuan, 2009). However, it is possible that the slightly positive relationship observed between ‘crawlers’ and sediment in my study could be due to sampling bias and processing errors. For instance, it is possible that a higher proportion of ‘crawling’ macroinvertebrate taxa recorded were due to them being unable to seek refuge in their natural interstitial habitats and thus making them easier to capture via kicknet (Pollard & Yuan, 2009). Additionally, the modalities ‘swimmer’ and ‘attached’ decreased as sediment levels increased. High levels of sediment deposition may have degraded the habitat of these invertebrates (Pollard & Yuan, 2009). High concentrations of sediment were not measured but may significantly hinder ‘swimming’ macroinvertebrates by making it more difficult for them to move through the water column (Clapcott et al., 2017).

Overall, the modalities most strongly associated with higher levels of DFS were traits relating to life history and mobility. Many other modalities also had statistically significant responses to sediment, such as ‘desc1’, ‘desc4’, ‘lda4’, ‘two’, ‘eggcement’, ‘attached’, ‘filterfeed’, and ‘lda2’. I predicted that the modality ‘desc4’ (Maximum number of descendants per reproductive cycle: >3000) would increase in response to rising sediment levels, but the results reported in this study and by Clapcott et al. (2017) displayed a decrease, with the modality ‘desc1’ (≤100) increasing instead. A higher number of descendants produced should confer resilience due to compensating for increased mortality in response to high sediment levels. However, it is possible that producing higher numbers of descendants may have trade-offs with other traits sensitive to sedimentation (Clapcott et al., 2017). Sediment deposition upon the streambed likely severely degraded potential egg-laying sites, thus reducing the number of descendants that can be produced.

Although not statistically significant in my study, the modalities ‘size4’, ‘size5’ and ‘burrower’ are still ecologically important to discuss. This is because these three modalities were predicted to display a strong response to sediment (Clapcott et al., 2017; Lange et al., 2014). Specifically, it was predicted by Clapcott et al. (2017) and Phillips & Smith (2012) that larger body sizes (‘size4’ and ‘size5’) would decrease and smaller body sizes (< ‘size3’) would increase as fine sediment levels increased. Surprisingly, in this study, only the modality ‘size4’ decreased slightly ($P<0.1$) with all other body sizes modalities showing equivocal responses as sediment increased. It was predicted that smaller body sizes are associated with higher rates of reproduction and, in turn, increased resilience due to compensating for high mortality rates (Clapcott et al., 2017; Phillips & Reid, 2012a; Phillips & Reid, 2012b).

Larger body size might confer some benefits in silted habitats because it might provide macroinvertebrates a greater ability to escape after burial (Conroy et al., 2018). However, the study by Conroy et al. (2018) found that the body size of a macroinvertebrate did not affect its ability to escape from sediment, with swimming ability being a better predictor. Another possibility that could explain the equivocal results in body size in response to rising sediment levels are the increased levels of specialization associated with larger body size (Usseglio-Polatera et al., 2000a). Usseglio-Polatera et al. (2000a) discussed how macroinvertebrates with larger body sizes are more specialized to access specific food sources. Therefore, it may be possible that larger body sizes provide some larger macroinvertebrates with a competitive advantage to access specific food sources in highly sedimented streams (Usseglio-Polatera et al., 2000a).

Additionally, it has been predicted that deposit-feeding and burrowing macroinvertebrates will increase in response to sediment while filter feeders will decrease (Phillips & Reid, 2012b). The prediction made regarding deposit-feeders is due to the higher rates of detrital (FPOM) deposition associated with streams affected by sedimentation (Phillips & Reid, 2012b; Thompson & Townsend, 2000), while the prediction made regarding burrowing macroinvertebrates is due to the provision of additional habitat space in the form of fine sediment (Dolédec et al., 2011). The prediction regarding filter-feeding is due to sediment clogging their specialized mouth parts and/or degrading food quality (Merritt & Wallace, 1981). Although two of these predictions were correct as ‘filter-feeding’ and ‘burrowing’ increased as deposited fine sediment levels rose, ‘deposit’ feeding macroinvertebrates did not. A possible explanation for this is that high levels of DFS decrease substrate heterogeneity and degrade the habitat and food resources of deposit feeders (Sutherland et al., 2012).

4.2.3 *Trait relationship with land-use*

It is important to note that as stream ecosystems often face multiple stressors that are related due to originating from a common source, such as anthropogenic land-use, it can be challenging to identify the specific effects of each separate stressors (Lange et al., 2014; Townsend et al., 2008). This means certain traits can respond to multiple stressors, meaning traits that have responded to sediment can also respond to land use (Lange et al., 2014; Townsend et al., 2008).

Regarding trait responses to pastoral land use, it has been identified that modalities associated with shorter generation times, asexual reproduction, hermaphroditism, laying eggs beneath the water's surface, scrapers, algal piecers, egg protection and respiration times tolerant of oxygen depletion increase in response to pastoral land-use (Dolédec et al., 2011). Also, it has been identified that modalities associated with shortened adult life duration, laying eggs at the water surface, sexual reproduction and low body flexibility decreased in response to pastoral land use (Dolédec et al., 2011).

In this study I found that the following modalities were higher in pastoral streams in comparison to native forest streams: 'pluriv' (shorter generation time); 'single' (asexual reproduction); 'submerged', (laying eggs below the water's surface); 'scraper', (algal feeding); 'agalp', (algal piercer); 'eggprotected', (female bears eggs in/on body); and 'plastron' (respiration). The following modalities were lower in pastoral streams: 'lda2', 'lda3', (Shorter adult life duration); 'surface', (laying eggs at the surface of the water); and 'lowflex'. All the trends in modalities observed in my study were consistent with the predictions made by Dolédec et al. (2011), except for 'herma' which did not increase, and 'two' (sexual reproduction) which did not decrease in response to pastoral land use. Another contradictory result was for the modality 'shredder', which was highest in pastoral streams. This result might be explained by the changes in riparian vegetation, specifically from native evergreen to exotic deciduous species, which could provide higher and more frequent inputs of CPOM (Winterbourn et al., 1981). Alternatively, more depositional habitats might explain a strong positive relationship between the modality 'shredder' and deposited fined sediment. More consistent with previous studies, the modalities 'filter-feeder' and 'predator' showed negative relationships with deposited fine Sediment.

The strong positive relationship displayed by 'shredders' is surprising as previous studies have shown that the availability of CPOM decreases due to it being smothered by deposited fine sediment, decreasing its quality and accessibility (Ntloko et al., 2021). Some shredders can be effective bioturbators capable of resuspending fine sediment, which is then carried downstream, preventing it from accumulating on the leaf packs that they feed on and increasing the availability of CPOM to shredder species (Sanpera-Calbet et al., 2012). This mechanism might explain the unexpected positive relationship observed between 'shredders' and deposited fine sediment in my study.

The negative relationship displayed by 'filter-feeders' can be explained by the fact that sediment can clog their specialised mouth apparatus, impairing their ability to feed (Conroy et al., 2018; McKenzie et al., 2022a). This mechanism could also explain the negative relationship between the modality ‘gill’ and deposited fine sediment, as sediment can also clog macroinvertebrate's gills, impairing their ability to respire (Conroy et al., 2018; McKenzie et al., 2022a).

The negative response of predators might be mediated through trophic pathways, with bottom-up effects of sediment constraining energy flows (Burdon et al., 2020). Alternatively, predatory invertebrates (e.g., *Stenoperla prasina*, *Archichauliodes diversus*, *Antipodochlora braueri*) are typically larger-bodied and longer-lived meaning they may have possessed life history traits (e.g., maximum number of reproductive cycles per year) that conferred greater sensitivity to anthropogenic disturbances such as sedimentation (Devonport & Winterbourn, 1976; Stark et al., 2001; Winterbourn, 1974).

Another surprising result was the relatively strong positive relationship between the modality ‘specialist’ and deposited fine sediment. ‘Specialisation’ is a modality for the trait resource acquisition (dietary preferences) and indicates that taxa with more specific dietary requirements had greater abundances in sites affected by sedimentation (Phillips & Smith, 2018). This result may have been linked to the ‘shredder’ result, where an obligate shredder like the leptocerid caddisfly *Triplectides* spp. possessed life history traits conferring resistance to sediment impacts (Brand & Miserendino, 2011). It is also plausible that obligate deposit feeders were more abundant at sites with greater deposited sediment levels.

Interestingly, the resource acquisition modalities ‘moderate specialisation’ and ‘generalist’ had a negative relationship with sediment, which was consistent with the predicted effects of sediment on trophic plasticity (Dolédec et al., 2011). For instance, it was found in the study by Dolédec et al. (2011) that both of these modalities decreased as sediment levels increased due to a lack of high levels of specialisation in dietary habitats required to confer resilience. Taxa with more specialised modes of food acquisition, such as biofilm grazers like the mayfly *Deleatidium,* would be expected to be negatively affected by sediment due to the loss of rheophilic habitat and contamination of food resources (Dolédec et al., 2011; Lange et al., 2014; Winterbourn, 1974).

The potential conflict caused by trait syndromes in which multiple traits can all confer sensitivity to sedimentation (Menezes et al., 2010). For example, although eggs laid on submerged surfaces may be able to find a suitable microhabitat where they will not be smothered, other related traits have been shown to influence this suitability (Conroy et al., 2018). For instance, the modality ‘eggfree’ had a negative relationship with sediment, while ‘eggcement’ displayed the opposite result. Such habitat-mediated ‘life stage bottlenecks’ could help explain why some taxa were more sensitive to sedimentation (Claptcott et al., 2017).

It is also worth discussing which taxa were indicative of a specific land use type (native forest, pastoral and horticultural). For instance, the following taxa were indicator of native forest streams: *Coloburiscus, Zephelbia*, *Austroperla*, *Zelandoperla*, *Beraeoptera* and *Heliopsyche* (Chapter 3). Collectively, these taxa show a strong affinity (score of 3) towards the following modalities ‘Size1’, ‘Size2’, ‘Size3’, ‘desc2’, ‘desc4’, ‘semi’, ‘univ’, ‘cpi1’, ‘lda2’, ‘lda3’, ‘lda4’, ‘two’, ‘surface’, ‘eggfree’, ‘dissmedium’, ‘crawler’, ‘attached’, ‘noflex’, ‘lowflex’, ‘flattened’, ‘cylindrical’, ‘shredder’, ‘scraper’, ‘filterfeed’, ‘generalist’, ‘tegument’, ‘gill’, ‘aduorlar’ and ‘larandpup’ (Phillips & Smith, 2018).

However, pastoral streams only had a singular, exclusive indicator species, a singular beetle species in the Hydrophilidae family (Chapter 3). This species showed a strong affinity (score of 3) towards the following modalities “size2’, ‘univ’, ‘cpi1’, ‘lda4’, ‘two’, ‘submerged’, ‘terrestrial’, ‘eggfree’, ‘predator’, ‘plastron’ and ‘aduandlar’ (Phillips & Smith, 2018). Also, it was found that horticultural streams had no exclusive indicator species, as chironomid midge larvae were indicative of both horticultural and pastoral streams (Chapter 3). Chironomid taxa showed a strong affinity (score of 3) towards the following modalities, ‘size3’, ‘desc3’, ‘pluriv’, ‘cpi1’, ‘lda2’, ‘single’, ‘two’, ‘surface’, ‘submerged’, ‘eggcement’, ‘burrower’, ‘highflex’, ‘cylindrical’, ‘deposit’, ‘generalist’, ‘tegument’ and ‘larandpup’ (Phillips & Smith, 2018).

As these indicator species (Chapter 3) show a strong affinity towards many different modalities, further research should be undertaken to understand better the mechanistic relationship between these modalities and different land-use types (Menezes et al., 2010). In other words, although existing literature can explain how a strong preference for specific modalities can aid a macroinvertebrate in surviving in certain environmental conditions, other modalities remain largely misunderstood (Clapcott et al., 2017; Lange et al., 2014). Nevertheless, specific modalities associated with pastoral and horticultural stream indicator species likely confer resilience towards anthropogenic stressors, while specific modalities associated with native forest stream indicator taxa are more susceptible (Clapcott et al., 2017; Lange et al., 2014). This is highlighted by the fact that all of the macroinvertebrate species indicative of pastoral and horticultural streams have low MCI scores (pollutant-tolerant), while the purely EPT taxa indicative of native forest streams have high MCI scores (pollutant-intolerant) (Stark & Maxted, 2007).

5 Chapter 5: Synthesis and conclusions

5.1 Hypothesis outcomes

In summary, I found evidence to support all my hypotheses except for the third one (negative impacts of sediment and/or land use on functional diversity), which was equivocal. Firstly, I hypothesized that deposited fine sediment would explain the most variation between stream sites in different land uses but that any remaining differences not explained by sediment would indicate the influence of other stressors such as contaminant pollution (pesticides), nutrient enrichment, flow alteration, and loss of riparian vegetation (Jackson et al., 2021; Lange et al., 2014; Matthaei et al., 2010; Townsend et al., 2008; Villeneuve et al., 2018; Wagenhoff et al., 2011). This was confirmed by the significant quantities of DFS present in pastoral and horticultural streams, along with clear and significant declines in the Macroinvertebrate Community Index (MCI), QMCI, and Average Score Per Metric (ASPM) scores along with EPT and fine sediment 'decreaser' abundances and richness (Chapter 3).

For the second part of my hypothesis, I found some evidence for this alternative hypothesis predicting that any remaining differences not explained by sediment would indicate the influence of other stressors. Specifically, nitrate concentrations were significantly higher in pastoral and horticultural streams in comparison to native forest streams. Also, Riparian Condition Index (RCI) scores were significantly lower in pastoral and horticultural streams in comparison to native forest streams, indicated that riparian condition was degraded in the human land-use types. However, the only environmental variable tested that was not statistically significant across the three land-use types was stream flow velocity. This was despite stream flow velocities showing the trend of being highest in horticultural streams, followed by pastoral and then native forest streams. The lack of a significant difference at α=0.05 reflected the fact that stream flow velocities were highly variable across sites.

I also found evidence to support my hypothesis that sediment-specific indicators should better detect the impacts of this stressor. Many traditional (bioindicators) and functional (trait-based) macroinvertebrate indicators responded significantly to sediment. The Macroinvertebrate Community Index responded to sediment, which reflects the use of tolerance scores for different taxa to assess pollution levels (Stark & Maxted, 2007). Although the Macroinvertebrate Community Index was originally designed to assess levels of organic/nutrient enrichment within lotic ecosystems, in this study, it appears to indicate levels of sedimentation present (and/or other co-occurring stressors, such as saprobic conditions) within the study sites. However, there was a stronger result with the fine sediment "decreasers" indices, indicating that sediment-specific indices may be better placed to detect the impacts of this stressor and match DFS-macroinvertebrate relationships.

The influence of multiple stressors remains an unresolved issue in my analyses, that could be further explored with more complex models and sophisticated approaches. The entanglement of multiple stressors is currently an area of active ecological research. This is highlighted by the study conducted by Lange et al. (2014), which sought to disentangle biological trait responses to multiple stressors. Those authors identified which biological traits were most strongly associated with multiple specific stressors such as farming intensity, water abstraction, nutrients and sediment (Lange et al., 2014). Although the statistical testing in my thesis was not as comprehensive as that employed by Lange et al. (2014), many published literature sources were used to compare similarities and differences in the response of certain traits to sediment, helping to confirm that the functional indicators (traits) used were previously found to be stressor-specific to sediment. Further research quantifying land use and land cover (LULC) attributes at the catchment scale would help better link specific land use types to stream responses, since cumulative impacts could be a feature of my sites.

My third hypothesis was that non-random species loss would lead to negative land-use impacts on functional diversity (Chapter 4). This hypothesis was partially true, as horticultural sites had the lowest levels of functional diversity overall. Notably, many horticultural sites were in the lower reaches, meaning they experienced the cumulative impact of stressors upstream (Vries et al., 2019). However, stressors specific to this land-use type, such as pesticides are also highly likely to affect horticultural streams substantially, reducing functional diversity (Knillmann & Liess, 2019; Magbanua et al., 2013).

However, pastoral streams displayed higher levels of functional diversity than native forest streams, which was unexpected. For example, metrics such as Rao's Quadratic Entropy (*hereafter* Rao's Q), functional richness and divergence were highest in pastoral streams. In Chapter 4, I explained that this heightened functional diversity is not necessarily positive (Bonilla-Valencia et al., 2022). For example, evidence showed that functional divergence was positively correlated with deposited fine sediment. This suggests that the heightened functional diversity exhibited by pastoral streams was in response to sedimentation (Bonilla-Valencia et al., 2022). Although high functional diversity is linked to enhanced ecosystem functioning, it does not mean the ecosystem overall is healthy (Bonilla-Valencia et al., 2022; Tilman et al., 1997). This was highlighted by the fact that pastoral streams had the lowest MCI scores. This was likely due to the establishment of pollutant-tolerant taxa with additional traits conferring resilience to sedimentation, resulting in increased functional diversity (Bonilla-Valencia et al., 2022). A further explanation for my reasoning is provided below in my comparison of the ability of traditional taxonomic and trait-based approaches to determine ecosystem health.

Another possible explanation for the heightened functional diversity present in pastoral streams was that high inputs of organic matter appeared to support a larger and more complex trophic food web in comparison to native forest streams (Riley et al., 2003), and these sites may have a greater proportion of depositional habitats possessing more organic matter.

It was also notable that certain functional traits responded to sediment. Traits have a strong mechanistic relationship with different stressors, such as sediment, that are not confounded by differences in community assemblage, spatial scale or geographic region (Menezes et al., 2010). Therefore, functional traits may be better able to distinguish individual stressors due to their more mechanistic relationships with the environment (Menezes et al., 2010). My fourth hypothesis predicted that specific macroinvertebrate trait modalities would respond to sediment stress and pastoral land use, respectively. Specifically, I predicted that the following modalities would respond to Deposited Fine Sediment (DFS), 'submerged', 'crawler', 'plurivoltine', 'size1', 'surface', 'eggprotected', 'filter-feeding' and 'swimmer' (Clapcott et al., 2017; Dolédec et al., 2011; Philips & Smith, 2012a; Phillips & Reid, 2012b). All modalities responded to sediment stress with the predicted pattern, with the only two that did not display a statistically significant result being 'size1' and 'eggprotected'. Specifically, the modalities 'submerged', 'plurivoltine' and 'eggprotected' increased in response to sediment, while the modalities 'crawler', 'size1', 'surface', 'filter-feeding' and 'swimmer' decreased in response to sediment.

Regarding agricultural land use, I predicted that the following modalities would respond 'pluriv', 'single', 'submerged', 'scraper', 'algalp', 'eggprotected', 'plastron', 'lda2', 'lda3', 'surface', 'lowflex', 'herma' and 'two'. All responded to sediment with the predicted pattern except for 'herma', which did not increase and 'two', which did not decrease in response to agricultural land use. Specifically, the following modalities increased in response to land use 'pluriv', 'single', ‘submerged’, ‘scraper’, ‘algalp’, ‘eggprotected’ and ‘plastron’. The following modalities decreased in response to agricultural land use: ‘lda2’, ‘lda3’, ‘surface’ and ‘lowflex’.

These results imply that these modalities can be used as functional indicators of sediment stress and land use, respectively (Clapcott et al., 2017; Philips & Smith, 2012a; Phillips & Reid, 2012b). However, as discussed above, the same traits can respond to multiple stressors due to the entanglement of multiple stressors (Lange et al., 2014). For instance, the modalities ‘pluriv’, ‘submerged’, ‘surface’ and ‘eggprotected’ responded to both deposited fine sediment and land use. This is likely due to the fact that agricultural land uses are associated with sedimentation (Magbanua et al., 2010), along with other stressors (Allan, 2004).

Nevertheless, further research should be conducted to identify which specific modalities are associated with individual stressors using statistical models that control for multiple stressors. The study by Lange et al. (2014) serves as a good foundation for this research, along with the more recent study by Clapcott et al. (2017), which investigated the potential for traits to be used as a functional indicator under the New Zealand National Policy Statement for Freshwater Management (NPS-FM). Clapcott et al. (2017) found evidence that the four traits most indicative of sediment stress, backed by ecological theory, are ‘herma’, ‘two’, ‘univ’ and ‘lowflex’. While the four traits most reflective of agricultural land use are ‘eggsprotected’, ‘cpi1’, ‘submerged’, and ‘desc1’ (Clapcott et al., 2017). However, although these traits are not shared between responses to sediment stress and agricultural land use, not all of them responded in a predicted manner. For example, this study found instances of the modality ‘desc1’ (maximum number of descendants per reproductive cycle: ≤100) responding in the opposite direction to what was expected based on my interpretation of ecological theory. In other words, more descendants were not always favourable in response to agricultural land use (Clapcott et al., 2017). This suggests that further research needs to be conducted to understand the mechanistic relationship between a trait and its stressor (Menezes et al., 2010).

5.2 Comparison of taxonomic and trait-based approaches

The report by Clapcott et al. (2017) provides an important insight into the comparison between traditional taxonomic and trait-based approaches to assessing ecosystem health. While taxonomic-based approaches (such as the Macroinvertebrate Community Index) and trait-based approaches are valuable tools for evaluating ecosystem health, they offer different perspectives.

Trait-based approaches excel at identifying individual stressors faced by specific stream communities, providing a nuanced understanding of the ecological impacts (Clapcott et al., 2017). This granularity is particularly beneficial for local assessments where the type and intensity of stressors can vary significantly among sites (Clapcott et al., 2017).

On the other hand, the Macroinvertebrate Community Index (MCI) stands out in its capacity to incorporate the combined response of macroinvertebrates to the entirety of stressors in an ecosystem (Clapcott et al., 2017). Furthermore, it provides a standardized metric of ecosystem health in the form of a score that can be used to make comparisons across many different regions (Clapcott et al., 2017). Additionally, taxonomic approaches help resolve the complexity of trait-based approaches by consolidating intricate datasets of individual taxa into simpler metrics, such as MCI scores.

Essentially, trait-based approaches provide a detailed, mechanistic perspective, whereas taxonomic approaches provide a generalist perspective, with both being valuable (Clapcott et al., 2017). For instance, taxonomic approaches alert us to the general 'sickness' of an ecosystem (loss of ecosystem health) in the form of the loss of sensitive species, whereas trait-based approaches can identify specific symptoms in the form of change in trait composition, which can help diagnose the issue causing them (stressors). Given this logic, it would be wiser to utilise the quantitative version of the MCI (QMCI). This is because the MCI only responds to complete losses or replacement of sensitive taxa due to environmental pollution, meaning that typical changes in relative abundances are not detected in response to more subtle changes in environmental conditions. Therefore, the QMCI might provide a better early warning signal of ecosystem degradation.

However, biomonitoring using these indicators is more of a reactive than a pro-active response to ecological degradation since management actions would only be taken after sufficient disturbance, causing the loss of sensitive tax. Therefore, it might be helpful to use trait-based approaches to detect early signs of ecological degradation so to take management actions before significant adverse changes in the ecosystem occur (Hewitt et al., 2014). For example, research indicates a discernible trend as functional diversity decreases as environmental stress or constraints increase (Judi Hewitt et al., 2014; Mouillot et al., 2006). Moreover, evidence suggests that the escalation of environmental constraints corresponds with an increase in the similarity of functional traits between species (Statzner et al., 2004). This phenomenon can be attributed towards the narrowing of environmental filters, which imposes heightened selective pressure (Statzner et al., 2004). As a result, the species capable of passing through these restricted filters exhibit shared biological characteristics (Bonilla-Valencia et al., 2022; Judi Hewitt et al., 2014; Statzner et al., 2004). Consequently, this process leads to the functional homogenization of ecological communities, where species richness increases but functional diversity decreases due to the functional similarity shared between species (Bonilla-Valencia et al., 2022). This functional homogenization is an early warning sign of ecological degradation (Bonilla-Valencia et al., 2022).

Most importantly, these changes can occur before the loss of sensitive species (Mouillot et al., 2012). Therefore, trait-based approaches provide a quantifiable method to estimate functional diversity linked to ecosystem functioning. However, although trait-based approaches have been suggested as an alternative and even a replacement for traditional taxonomic indicators in assessing the health of an ecosystem, they pose a fundamental issue (Bonilla-Valencia et al., 2022; Ghilarov, 2000; Judi Hewitt et al., 2014).

Just because an ecosystem has higher levels of functional diversity does not always imply that it is healthy (Bonilla-Valencia et al., 2022). This can be explained using the functional transformation hypothesis, which predicts that anthropogenic disturbances can eliminate existing environmental filters, allowing novel and/or invasive species that would otherwise be excluded to establish themselves within an ecosystem. Consequently, this can increase functional diversity if the functional traits possessed by the invasive species are different from those present in the original community (Bonilla-Valencia et al., 2022). Additionally, species richness is predicted to increase under the functional transformation hypothesis. However, in this study it was unclear if sufficient evidence was present to accept this hypothesis as insufficient sampling over a longer time period was not performed. Although species richness overlapped between some native forest and pastoral stream sites (Figure 5b & 6b), it is unclear if this heighted diversity was due to species turnover as a result of predictions made by the functional transformation hypothesis or due to other causes.

Nevertheless, it would be wise to use traditional taxonomic approaches alongside trait-based approaches to accurately measure all aspects of ecosystem health (Bonilla-Valencia et al., 2022).

For instance, the three core aspects of ecosystem health have been defined as vigour, resilience and organization (Costanza & Mageau, 1999). Using only trait-based approaches means that although the vigour or function may be described it may not accurately measure the organization (structure) or resilience of an ecosystem. Essentially, although an ecosystem may maintain its vigour in the face of external stress (resilience), its organization may have changed significantly as functions previously performed by an ecosystem's original native species may now be performed by novel or exotic species (Bonilla-Valencia et al., 2022; Costanza & Mageau, 1999). If an ecosystem has these properties, then it can be questioned whether it is resilient and thus healthy, as it cannot maintain its original organization in response to external stress (Bonilla-Valencia et al., 2022; Costanza & Mageau, 1999).

Therefore, it would be wise also to employ the use of traditional taxonomic metrics to observe changes in the community composition of an ecosystem (organization) alongside trait-based metrics to ensure all aspects of ecosystem health can measured accurately and without bias (Bonilla-Valencia et al., 2022; Costanza & Mageau, 1999). Such a combined multivariate metric has already been proposed in the study by Clapcott et al. (2017) to be used as an official metric under the National Policy Statement for Freshwater Management (NPS-FM).

5.3 Limitations

Although evidence was found to support my first, second and fourth hypotheses, the inability to clearly accept or reject my third hypothesis, which predicted that non-random species loss (i.e. trait-mediated) would lead to negative land-use impacts on functional diversity, highlights the limitations of this study. For instance, it is likely that the method used to select habitat for sampling and the sampling tool used were biased in assessing functional diversity. Consequently, specific habitats were likely not sampled as equally or intensively as others, resulting in discrepancies in functional diversity.

Another limitation of this study is that sampling was conducted as a singular occurrence, preventing the identification of temporal trends occurring in the macroinvertebrate communities sampled. For instance, NOF (National Objective Framework) bands for macroinvertebrate indicators are calculated over averaging five years of data to account for fluctuations in temporal trends (Clapcott et al., 2017). Additionally, not all sites were purely independent of each other as they were sampled at different locations across the longitudinal profile of the same stream, although this influence was accounted for in my mixed models. According to metacommunity ecology, it is possible that species present in these sites (mass effects) could have originated from another site further upstream (Heino et al., 2015). However, care was taken to ensure sufficient distance between such sampling sites to increase their level of independence (Stark & Maxted, 2007).

Also, the land-use categories assigned to different sites may not have entirely reflected all land-cover types present (Ministry for the Environment, 2010). The land cover classifications assigned to each site derived from the New Zealand River Environment Classification (REC), which reflects the dominant land cover type upstream differed in some instances from my land-use determinations (Ministry for the Environment, 2010). For example, if more than 50% of the land cover upstream is native forest but 40% is pastoral, it is still classified as native forest (Ministry for the Environment, 2010). The REC database also does not discern between pastoral and horticultural land uses.

Additionally, due to the large number of individual macroinvertebrates that were processed, mistakes will inevitably be made when identifying taxa (Stark et al., 2001). This was controlled by ensuring that quality control methods were implemented (Stark & Maxted, 2007). This involved reinspecting macroinvertebrate individuals to ensure they were identified correctly (Stark & Maxted, 2007). However, it is unlikely that the number of taxonomic mistakes made was not numerous enough to significantly alter assessments made, such as the Macroinvertebrate Community Index (Stark & Maxted, 2007).

5.4 Implications and proposed actions

Nevertheless, the results of this study provide clear evidence of the adverse effects of sedimentation on New Zealand stream macroinvertebrate communities. It is critical that the issue of sedimentation is addressed to prevent further degradation of stream ecosystems. As established previously, sedimentation is a significant issue within the Te Awanui/Tauranga Moana catchment within the Bay of Plenty. Although average sediment accumulation rates (SAR) within the Tauranga Harbor are small (0.75-1.57 mm/yr) in comparison to the maximum SAR derived from NIWA (25 mm/yr) or to the average SAR of other harbours such as the Auckland's Kaipara Harbour (5mm/yr) or in the Bay of Islands and the Pauatahanui Inlet in Porirua (2.5 mm/yr), the issue of climate change dramatically intensifies the issue.

An investigation into the issue of sedimentation by NIWA (National Institute of Water & Atmospheric Research) in collaboration with the Bay of Plenty Regional Council predicted that modelled SAR rates are expected to increase by 42.8% by 2051 (Elliot et al., 2010). This is due to the predicted increases in mean annual rainfall (4.4%) by 2050, with SAR rates increasing with the square of mean annual precipitation (Elliot et al., 2010). Although a 4.4% increase in mean annual rainfall does not logically seem to correspond to a 42.8% increase in SAR rates, the dramatic increase in SAR rates can be explained by the increased variability in the maximum range of the highest rainfall (mm) over the next 27 years. For example, under the model developed by NIWA, the highest daily rainfall (240.6 mm) was predicted to increase by 9.7% by 2051, corresponding to a 23% increase in SAR. The study determined that the combined increase of the modelled highest rainfalls resulted in an average SAR increase of 42.8% (Elliot et al., 2010).

These findings are significant as it has been estimated that as of 2014, a total of 120,000 tonnes of sediment enter the Tauranga Harbour annually (McKenzie, 2014; McKenzie et al., 2022b). This figure, combined with the modelled increases in Sediment Accumulation Rates (SAR), indicates potentially dire consequences for not just stream and estuarine ecosystems within the Tauranga Moana catchment but on a national level too. Therefore, the issue of sedimentation must be addressed with greater urgency immediately.

Since these results were released by McKenzie (2014) and Hume et al. (2010b), six mitigation actions were proposed and have since been implemented to address the issue of sedimentation. The first action was riparian planting, which prevents bank erosion, as it has been indicated that it contributes to approximately 10% of the sediment load in North Island streams (Hume et al., 2010b). The low Riparian Condition Index values for pastoral and horticultural streams shown in this study (Figure 2c) indicate that there is still a strong need for more riparian planting in anthropogenic land uses in the Bay of Plenty.

The second action was the retirement of steep, grazed pasture, which was found to reduce sediment loads by a factor of 5−10 times (Blascke et al., 2008). The third action was to establish additional pine plantations as it has been found that they contribute low sediment loads comparable to those of native forests (Blascke et al., 2008). The fourth action was to implement additional controls on forestry, such as buffers, as most sediment released from pine forests occurs during harvesting (Quinn et al., 2004). The fifth action was to divert the flow of sediment away from main river channels and into floodplains, as it has been found that a floodplain area comprising only 1% of the catchment can reduce sediment loads by 50% (Hume et al., 2010b). The sixth action was to ensure that all earthworks complied with the Erosion and Sediment Control Guidelines enforced by the Bay of Plenty Regional Council (Clemens & Dunphy, 2010). However, it was noted that the contribution towards sediment loads by uncontrolled earthworks would be minor, but not insignificant (Hume et al., 2010b). It was suggested that in severe cases, measures such as flocculation ponds could be implemented (Hume et al., 2010b).

These actions have probably had a favourable impact on diminishing sediment loads entering the streams and rivers of the Bay of Plenty. Nevertheless, the findings presented in my thesis indicate that many of the sampled streams are currently grappling with the detrimental consequences of sedimentation. However, not enough time has likely passed for the full effectiveness of these actions to be observed. For instance, existing research suggests that erosion and sediment control methods may take up to 15 years to exhibit their complete effectiveness. Although sediment control practices such as livestock fencing and sediment ponds can become effective immediately, riparian vegetation and pine afforestation can take up to 15 years to become effective (Basher et al., 2020). This is simply due to the time it takes for vegetation to reach maturity (Basher et al., 2020).

5.5 Areas for further research

As recommended in the study by Hume et al. (2010b), future research should address the following points to improve estimates of sediment loads entering the Tauranga Harbor via freshwater sources and improve the effectiveness of implemented mitigation strategies. Firstly, further monitoring of sediment loads in different land uses needs to be undertaken. Secondly, long-term monitoring of sediment loads in relation to rainfall needs to be continued. Thirdly, it would be wise to conduct further research into the rate at which stream bank erosion and surface runoff occurs in the Bay of Plenty. Fourthly, it would be wise to investigate further the feasibility and effectiveness of the use of floodplains in controlling sediment deposits. Fifthly, further monitoring of the effects of earthworks in relation to erosion and sediment deposition needs to be conducted. Finally, further research must be undertaken to understand better the components of the existing stream sediment transport model developed for the Bay of Plenty Region (Bryan et al., 2023; Dorner et al., 2018 Hume et al., 2010b).

Regarding the use of macroinvertebrate functional traits to assess the effects of sedimentation, future research should address the following points. Firstly, there is a strong need to understand better the mechanistic relationship between functional traits and the issue of sedimentation (Lange et al., 2014). However, several issues pose a barrier to developing further insight and thus need to be addressed.

Firstly, the entanglement of multiple stressors makes it challenging to distinguish which specific stressor functional traits are responding to (Lange et al., 2014). For example, in the case of this study, deposited fine sediment is strongly correlated with pastoral land use, making it difficult to distinguish precisely which stressor (sediment vs land use) certain traits are responding to, although my statistical models attempted to disentangle these effects. Other statistical approaches with larger datasets, such as the use of a Gradient Forest analysis with the large amount of site data used by Clapcott et al. (2017) appear to help disentangle multiple stressors, thus making it easier to identify which stressor functional traits are responding.

Secondly, trait syndromes make it challenging to identify which functional traits respond directly to specific stressors. For example, the study by Poff et al. (2006) highlighted that the evolutionary intercorrelation between traits makes distinguishing between traits that have developed due to evolution or response to stressor-related environmental changes difficult. Furthermore, trait trade-offs make it difficult to distinguish which traits respond to stressors and which are just a result of their life history strategy, separate from stressors (Menezes et al., 2010).

Thirdly, the inconsistencies among freshwater macroinvertebrate trait databases globally need to be reconciled to ensure comparability across different trait-based studies (Kunz et al., 2022). For instance, trait databases across New Zealand, Australia, North America and Europe differ in their definition of certain traits and coding to describe them (Kunz et al., 2022). This difference in trait definition across countries occurred due to the use of multiple traits to describe the same function, with the way certain functions were described varying (Kunz et al., 2022). For example, many different traits were used to describe the function of feeding, but the description of this function also varied as some focused more on traits associated with the macroinvertebrate's food source while others focused on the morphology of its mouthparts (Kunz et al., 2022).

Also, the New Zealand trait database uses fuzzy coding while the North American database used binary coding, making it difficult to distinguish the degree to which a macroinvertebrate species expresses a particular trait (Phillips & Smith, 2018). Binary coding indicates whether a trait is present or absent in each macroinvertebrate species, while fuzzy coding provides insights into the degree to which specific traits are expressed. This choice of coding method holds significance, influencing the accuracy with which different trait modalities are assigned to various macroinvertebrate species. For example, given the multitude of sources informing the expression of different trait modalities across macroinvertebrate species, taxa may exhibit associations with more than one modality. Fuzzy coding helps to navigate these complexities, offering a more nuanced and comprehensive understanding of trait modality expression with different macroinvertebrate species.

Moreover, the variation in taxonomic resolutions employed across diverse trait databases introduces another challenge. Since trait databases commonly attribute trait information to macroinvertebrates at distinct taxonomic levels, such as genus or family, the precision of the data may fluctuate. For instance, when databases possess trait information at higher taxonomic resolutions than others, the approach is often to utilize the trait data assigned to lower taxonomic resolutions. This aggregation method can potentially lead to a loss of specificity in the trait data, as trait expression can vary across different taxonomic resolutions.

However, the study by Kunz et al. (2022) is an excellent example of how these issues can be addressed. First, they grouped all traits in different databases into commonly used trait categories such as life history and morphology. Then, they normalized the varying definitions of the same trait by combining them into a singular trait. After that, the trait affinity scores for each species were normalized. Median affinity scores were used instead of the mean, as they more accurately reflected the affinity score of binary-coded traits. Species with taxonomic precision below the family level were omitted, and duplicate entries were consolidated by taking the median for fuzzy-coded traits or the maximum for binary-coded traits. Only data that was available in one database but not another was used. Fuzzy and binary-coded traits were then normalized to a scale between 0 and 1 (Kunz et al., 2022).

The results of the study by Kunz et al (2022) found that traits aggregated at a family level were similar to those assigned by experts, especially when using median trait values. This indicated that the aggregation of trait definitions and affinity across all databases was correct. Furthermore, they found that harmonized and aggregated data identified relationships between traits and the environment that were similar to non-aggregated data. This indicates that the harmonization and aggregation of traits still accurately reflected their relationship with environmental conditions. However, a discrepancy was observed when this harmonized and aggregated data was compared with functional diversity metrics presented in a relevant case study. This suggests that although this consolidation of trait databases was largely successful, it failed to capture subtle nuances in functional diversity, likely caused by the aggregation of trait affinity scores. Nevertheless, this study is an excellent example of addressing this issue and should serve as a foundation for future research (Kunz et al., 2022). It is important that these inconsistencies are resolved to ensure comparability between different studies internationally in order to better the effects that deposited fine sediment along with other anthropogenic stressors have on stream ecoystems.

Another area of further research that should be undertaken is the identification of accurate deposited fine sediment thresholds for New Zealand macroinvertebrate species (McKenzie et al., 2024). McKenzie et al. (2024) indicates that deposited fine sediment thresholds may be too high for New Zealand macroinvertebrate communities. For instance, in the study by Burdon et al (2013) it was found that as little as 4% sediment cover could have adverse effects on macroinvertebrate communities. However, any threshold is likely to be scale and context dependent (Burdon et al. 2013). The scale of observation (e.g., patch vs reach) can influence the exact threshold identified, with Burdon et al. (2013) finding a threshold of ≈13% deposited fine sediment at the patch scale as compared with ≈20% at the reach scale in lowland, gravel-bed Canterbury streams. Using quantitative sampling is essential for detecting robust threshold, and in my thesis, I only sampled using the semiquantitative NEMS kick-net protocol. Further, it has been identified that the accumulation of deposited fine sediment is highest in lowland gravel-bed rivers with stable flow regimes and a high presence of agricultural land use (Burdon et al., 2013; Naden et al., 2016a). In contrast, rivers with coarser substrates but run-off mainly derived from agricultural land use may have comparably lower levels of deposited fine sediment (McKenzie et al., 2024). However, this knowledge has led to a bias as lowland rivers/streams are studied in relation to deposited fine sediment more frequently than upland rivers (McKenzie et al., 2024).

This could cause bias in accurately determining the threshold at which significant taxonomic and functional changes occur in macroinvertebrate communities in response to deposited fine sediment (Mathers et al., 2022). The study by McKenzie et al. (2024) identified the threshold at which such changes occurred ranged between 30−50%. However, (Mathers et al., 2022) also highlighted that threshold values are highly context-dependent This was because lowland macroinvertebrate species had a higher tolerance to deposited fine sediment than upland species due to environmental filtering. In other words, lowland macroinvertebrate species in some regions may be more accustomed to living in high sediment conditions (i.e. soft-bottomed streams) than upland species.

This suggests that upland macroinvertebrate species may be more susceptible to taxonomic and functional changes at lower levels of deposited fine sediment (McKenzie et al., 2024). However, it should be noted that the threshold at which changes in functional diversity occurred was higher than taxonomic thresholds (McKenzie et al., 2024). This is due to functional redundancy's role in replacing the loss of macroinvertebrate species with those with similar functional roles (McKenzie et al., 2024). Nevertheless, these findings highlight the need for a context-specific role in determining accurate deposited fine sediment threshold for macroinvertebrates across all river typologies (McKenzie et al., 2024).

Lastly, I suggest that future research further investigate horticulture's role in the ecological degradation of New Zealand's freshwater ecosystems. For instance, it has been highlighted in the report by Larned et al. (2018) that there is a substantial lack of research focusing on horticultural pesticide inputs into New Zealand lotic ecosystems and any resulting ecological changes. This research gap needs to be filled, as the results of many international studies have provided strong evidence of the adverse effects of horticultural pesticide inputs on stream ecosystems (Larned et al., 2018; Sharma et al., 2019; Stone et al., 2014).

Additionally, the statistically significant differences in MCI score between only native forest and horticultural streams suggest that the usage of pesticides might be a possible explanation. For example, although pastoral streams had high levels of deposited fine sediment and nitrate and low riparian habitat condition index scores, the post hoc analysis revealed no statistically significant differences. However, it is worth mentioning that stream flow velocities were higher in horticultural streams, and more intensive sampling could detect differences in hydrogeomorphology. Nevertheless, I strongly recommend that future studies further investigate the impacts of horticultural land use on freshwater stream/riverine ecosystems.

In summary, future research must address the issues posed by multiple stressors, trait syndromes, trade-offs and databases, along with further determining macroinvertebrate sediment thresholds so that these stressors can be addressed through actions implemented through the National Policy Statement for Freshwater Management ("National Policy Statement for Freshwater Management," 2020)

6 Cited literature

Allan, J., & Johnson, L. (1997). Catchment-scale analysis of aquatic ecosystems. *Freshwater Biology*, *37*(1), 107-111. https://doi.org/https://doi.org/10.1046/j.1365-2427.1997.00155.x

Allan, J. D. (2004). Landscapes and Riverscapes: The Influence of Land Use on Stream Ecosystems. *Annual Review of Ecology, Evolution, and Systematics*, *35*, 257-284. https://doi.org/10.1146/annurev.ecolsys.35.120202.110122

Anderson, M. J. (2001). A new method for non-parametric multivariate analysis of variance. *Austral Ecology*, *26*(1), 32-46. https://doi.org/https://doi.org/10.1111/j.1442-9993.2001.01070.pp.x

Antwi, F. B., & Reddy, G. V. (2015). Toxicological effects of pyrethroids on non-target aquatic insects. *Environmental Toxicology and Pharmaco*logy, *40*(3), 915-923. https://doi.org/10.1016/j.etap.2015.09.023

Barquín, P., & Death, R. (2006). Spatial patterns of macroinvertebrate diversity in New Zealand springbrooks and rhithral streams. *Journal of The North American Benthological Society 25*(4), 768-786. https://doi.org/10.1899/0887-3593(2006)025[0768:SPOMDI]2.0.CO;2

Basher, L., Phillips, C., & Spiekermann, R. (2020). *Biophysical performance of erosion and sediment control techniques in New Zealand: a review*. Maanaki Whenua - Landcare Research Contract Report: LC3761.

Bay of Plenty Regional Council. (2016). *Tauranga Harbour catchment action plan progress report 2013-2016*. https://atlas.boprc.govt.nz/api/v1/edms/document/A2455450/content

Bay of Plenty Regional Council. (2023a). *Land use*. Bay of Plenty Regional Council. https://www.boprc.govt.nz/environment/land

Bay of Plenty Regional Council. (2023b). *Tauranga Harbour*. Bay of Plenty Regional Council. https://www.boprc.govt.nz/environment/coast-and-ocean/harbours/tauranga-harbour

Bêche, L. A., & Statzner, B. (2009). Richness gradients of stream invertebrates across the USA: taxonomy- and trait-based approaches. *Biodiversity and Conservation*, *18*(14), 3909-3930. https://doi.org/10.1007/s10531-009-9688-1

Bighiu, M. A., Höss, S., Traunspurger, W., Kahlert, M., & Goedkoop, W. (2020). Limited effects of pesticides on stream macroinvertebrates, biofilm nematodes, and algae in intensive agricultural landscapes in Sweden. *Water Research*, *174*, 115640. https://doi.org/https://doi.org/10.1016/j.watres.2020.115640

Bisson, P., A.,, Quinn, T., P.,, Revees, R., H.,, & Gordon, V., S.,. (1992). Best Management Practices, cumulative effects, and long term trends in fish abundance in Pacific Northwest river systems. In R. Naiman, J. (Ed.), *Watershed Management: Balancing Sustainability and Environmental Change* (pp. 189-233). Springer. https://doi.org/10.1007/978-1-4612-4382-3_7

Blascke, P., Hicks, D., & Meister, A. (2008). *Quantification of the flood and erosion reduction benefits, and costs, of climate change mitigation measures in New Zealand*. Report prepared by Blaschke and Rutherfor Environmental Consultants for MfE. Ministry for the Environment. https://environment.govt.nz/assets/Publications/Files/quanitification-flood-erosion-reduction-apr08.pdf

Blöcher, J. R., Ward, M. R., Matthaei, C. D., & Piggott, J. J. (2020). Multiple stressors and stream macroinvertebrate community dynamics: Interactions between fine sediment grain size and flow velocity. *Science of The Total Environment*, *717*. https://doi.org/https://doi.org/10.1016/j.scitotenv.2020.137070

Bonilla-Valencia, L., Castillo-Agüero, S., Zavala-Hurtado, J. A., Espinosa García, F. J., Lindig-Cisneros, R., & Martínez-Orea, Y. (2022). Linking functional diversity to ecological indicators: a tool to predict anthropogenic effects on ecosystem functioning. *Environmental Reviews*, *30*(2), 175-183. https://doi.org/10.1139/er-2021-0093

Bournaud, M., Richoux, P., & Usseglio-Polatera, P. (1992). An approach to the synthesis of qualitative ecological information from aquatic coleoptera communities. *Regulated Rivers: Research & Management*, *7*(2), 165-180. https://doi.org/https://doi.org/10.1002/rrr.3450070205

Bradley, D., & Ormerod, S. (2002). Evaluating the precision of kick-sampling in upland streams for assessments of long-term change: The effects of sampling effort, habitat and rarity. *Archiv fur Hydrobiologie*, *155*, 199-221.

Brand, C., & Miserendino, M. L. (2011). Life history strategies and production of caddisflies in a perennial headwater stream in Patagonia. *Hydrobiologia*, *673*(1), 137-151. https://doi.org/10.1007/s10750-011-0768-3

Briggs, R., M.,, Hall, R., J.,, Harmsworth, G., R.,, Hollis, A., G.,, Houghton, B., F.,, Hughes, G., R.,, Morgan, M., D.,, & Whitbread-Edwards, A., R. (1996). *Geology of the Tauranga area*. Occasional Report No. 22. University of Waikato. https://atlas.boprc.govt.nz/api/v1/edms/document/A1705483/content

Bryan, K., Stewart, B., Rahdarian, A. and Rautenbach, C. (2023). Ōhiwa Harbour Delft3D sediment transport modelling to support the National Policy Statement on Freshwater (NPS-FM). Prepared for the the Bay of Plenty Regional Council. Environmental Research Report No. 162, 16pp.

Bunn, S. E., & Arthington, A. H. (2002). Basic Principles and Ecological Consequences of Altered Flow Regimes for Aquatic Biodiversity. *Environmental Management*, *30*(4), 492-507. https://doi.org/10.1007/s00267-002-2737-0

Burdon, F. J., McIntosh, A. R., & Harding, J. S. (2013). Habitat loss drives threshold response of benthic invertebrate communities to deposited sediment in agricultural streams. *Ecological Applications*, *23*(5), 1036-1047. https://doi.org/https://doi.org/10.1890/12-1190.1

Burdon FJ, McIntosh AR, Harding JS. Mechanisms of trophic niche compression: Evidence from landscape disturbance. J Anim Ecol. 2020; 89: 730–744. https://doi.org/10.1111/1365-2656.13142

Cadotte, M. W., Carscadden, K., & Mirotchnick, N. (2011). Beyond species: functional diversity and the maintenance of ecological processes and services. *Journal of Applied Ecology*, *48*(5), 1079-1087. https://doi.org/https://doi.org/10.1111/j.1365-2664.2011.02048.x

Champagne, E. J., Guzzo, M. M., Gutgesell, M. K., & McCann, K. S. (2022). Riparian buffers maintain aquatic trophic structure in agricultural landscapes. *Biology Letters*, *18*(3). https://doi.org/10.1098/rsbl.2021.0598

Chappell, P., R. (2014). *The climate and weather of the Bay of Plenty* (3rd ed.). NIWA. https://niwa.co.nz/static/BOP%20ClimateWEB.pdf

Charvet, S., Kosmala, A., & Statzner, B. (1998). Biomonitoring through biological traits of benthic macroinvertebrates: Perspectives for a general tool in stream management. *Archiv fur Hydrobiologie*, *142*, 415-432.

Chevene, F., Dolédec, S., & Chessel, D. (2006). A fuzzy coding approach for the analysis of long-term ecological data. *Freshwater Biology*, *31*(3), 295-309. https://doi.org/10.1111/j.1365-2427.1994.tb01742.x

Chien, N. (1985). Changes in river regime after the construction of upstream reservoirs. *Earth Surface Processes and Landforms*, *10*(2), 143-159. https://doi.org/https://doi.org/10.1002/esp.3290100207

Clapcott, J., Casanovas, P., & Doehring, K. (2020). *Indicators of freshwater quality based on deposited sediment and Rapid Habitat Assessment.* Cawthron Report No: 3402. https://environment.govt.nz/assets/Publications/Files/indicators-of-freshwater-quality.pdf

Clapcott, J., Collier, K., Death, R., Goodwin, E., Harding, J., Kelly, D., Leathwick, J., & Young, R. (2011). Quantifying Relationships between Land-Use Gradients and Structural and Functional Indicators of Stream Ecological Integrity. *Freshwater Biology*, *57*(1). https://doi.org/10.1111/j.1365-2427.2011.02696.x

Clapcott, J., Harding, J., Matthaei, C., Quinn, J., Death, R., & Young, R. (2011). *Sediment Assessment Methods: Protocols and guidelines for assessing the effects of deposited fine sediment on in-stream values*. Cawthron Institute, Nelson, New Zealand. 108 p. https://www.envirolink.govt.nz/assets/R4-1-Sediment-Assessment-Methods-Protocol-and-guidelines.pdf

Clapcott, J., Wagenhoff, A., Neale, M., Storey, R., Smith, B., Death, R., Harding, J., Matthaei, C., Quinn, J., Collier, K., J, A., Goodwin, E., Rabel, H., Mackman, J., & Young, R. (2017). *Macroinvertebrate metrics for the National Policy Statement for Freshwater Management*. Prepared for the Ministry for the Environment. Cawthron Report No. 3073. 205p.

Clemens, D., & Dunphy, R. (2010). *Erosion and sediment control guidelines for land disturbing activities*. Prepared by Dudley Clemens, the Bay of Plenty Regional Council and Ridley Dunphy Environmental Limited. 248p. https://environment.govt.nz/assets/Publications/Files/quanitification-flood-erosion-reduction-apr08.pdf

Close, M., & Banasiak, L. (2023). *National Survey of Pesticides in Groundwater 2022.* Client Report No: CSC23010. https://www.knowledgeauckland.org.nz/media/m1ncztyh/national-survey-pesticides-in-groundwater-2022-esr-june-2023.pdf

Close, M. E. (1993). Assessment of pesticide contamination of groundwater in New Zealand. *New Zealand Journal of Marine and Freshwater Research*, *27*(2), 257-266. https://doi.org/10.1080/00288330.1993.9516565

Coccia, C., Almeida, B. A., Green, A. J., Gutiérrez, A. B., & Carbonell, J. A. (2021). Functional diversity of macroinvertebrates as a tool to evaluate wetland restoration. *Journal of Applied Ecology*, *58*(12). https://doi.org/https://doi.org/10.1111/1365-2664.14038

Collier, K. (1992). *Assessing River Stability: Use of the Pfankuch method*. Science and Research Internal Report No. 131. Department of Conservation. https://www.doc.govt.nz/globalassets/documents/science-and-technical/srir131.pdf

Collier, K., Clapcott, J., & Neale, M. W. (2014). *A macroinvertebrate attribute to assess ecosystem health for New Zealand waterways for the National Objective Framework-Issues and options*. Environmental Research Institute Report 36, University of Waikato, Hamilton.

Collier, K. J. (2008). Average score per metric: An alternative metric aggregation method for assessing wadeable stream health. *New Zealand Journal of Marine and Freshwater Research*, *42*(4), 367-378. https://doi.org/10.1080/00288330809509965

Collier, K. J., & Halliday, J. N. (2000). Macroinvertebrate-wood associations during decay of plantation pine in New Zealand pumice-bed streams: stable habitat or trophic subsidy? *Journal of the North American Benthological Society*, *19*(1), 94-111. https://doi.org/10.2307/1468284

Collier, K. J., & Quinn, J. M. (2003). Land-use influences macroinvertebrate community response following a pulse disturbance. *Freshwater Biology*, *48*(8). https://doi.org/https://doi.org/10.1046/j.1365-2427.2003.01091.x

Comiti, F., Mao, L., Wilcox, A., Wohl, E. E., & Lenzi, M. A. (2007). Field-derived relationships for flow velocity and resistance in high-gradient streams. *Journal of Hydrology*, *340*(1), 48-62. https://doi.org/https://doi.org/10.1016/j.jhydrol.2007.03.021

Connolly, N., Crossland, M., & Pearson, R. (2004). Effect of low dissolved oxygen on survival, emergence, and drift of tropical stream macroinvertebrates. *Journal of The North American Benthological Society 23(2),* 251- 270. https://doi.org/10.1899/0887-3593(2004)023

Conroy, E., Turner, J., Rymszewicz, A., Bruen, M., O'Sullivan, J., Lawler, D., Stafford, S., & Kelly-Quinn, M. (2018). Further insights into the responses of macroinvertebrate species to burial by sediment. *Hydrobiologia*, *805*. https://doi.org/10.1007/s10750-017-3328-7

Copp, G. H. (1989). The habitat diversity and fish reproductive function of floodplain ecosystems. *Environmental Biology of Fishes*, *26*(1), 1-27. https://doi.org/10.1007/BF00002472

Correa-Araneda, F., Núñez, D., Díaz, M., Gómez Capponi, F., Figueroa Jara, R., Acuña, J., Boyero, L., & Esse, C. (2021). Comparison of sampling methods for benthic macroinvertebrates in forested wetlands. *Ecological Indicators*, *125*(2021), 1470-1160. https://doi.org/10.1016/j.ecolind.2021.107551

Costanza, R., & Mageau, M. (1999). What is a healthy ecosystem? *Aquatic Ecology*, *33*, 105-115. https://doi.org/10.1023/A:1009930313242

Davies-Colley, R. J., Hickey, C. W., Quinn, J. M., & Ryan, P. A. (1992). Effects of clay discharges on streams. *Hydrobiologia*, *248*(3), 215-234. https://doi.org/10.1007/BF00006149

Davis, J. C. (1975). Minimal dissolved oxygen requirements of aquatic life with emphasis on Canadian species: a review. *Journal of the Fisheries Board of Canada*, 2295-2332. https://doi.org/10.1139/f75-268

Davis, N. G., Hodson, R., & Matthaei, C. D. (2022). Long-term variability in deposited fine sediment and macroinvertebrate communities across different land-use intensities in a regional set of New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research*, *56*(2), 191-212. https://doi.org/10.1080/00288330.2021.1884097

Dawson, S. K., Carmona, C. P., González-Suárez, M., Jönsson, M., Chichorro, F., Mallen-Cooper, M., Melero, Y., Moor, H., Simaika, J. P., & Duthie, A. B. (2021). The traits of "trait ecologists": An analysis of the use of trait and functional trait terminology. *Ecology and Evolution*, *11*(23), 16434-16445. https://doi.org/10.1002/ece3.8321

Dean, T. L., & Richardson, J. (1999). Responses of seven species of native freshwater fish and a shrimp to low levels of dissolved oxygen. *New Zealand Journal of Marine and Freshwater Research*, *33*(1), 99-106. https://doi.org/10.1080/00288330.1999.9516860

Décamps, H., Naiman, R. J., & McClain, M. E. (2009). The ecology of interfaces: riparian zones. *Annual Review of Ecology and Systematics 28*(1), 621-658. https://doi.org/https://doi.org/10.1016/B978-012370626-3.00053-3

Delerue, F., & Michalet, R. (2022). Effect and response traits in severe environments in the context of positive plant-plant interactions. A commentary on: 'Interspecific interactions alter plant functional strategies in a revegetated shrub-dominated community in the Mu Us Desert'. *Annuls of Botany*, *130*(2), 149-158. https://doi.org/10.1093/aob/mcac073

Department of Conservation. (2023). *Freshwater habitats*. https://www.doc.govt.nz/nature/habitats/freshwater/#:~:text=New%20Zealand's%20freshwater%20habitats%20range,geothermal%20areas%20and%20underground%20aquifers.

Devonport, B. F., & Winterbourn, M. J. (1976). The feeding relationships of two invertebrate predators in a New Zealand river. *Freshwater Biology*, *6*(2), 167-176. https://doi.org/https://doi.org/10.1111/j.1365-2427.1976.tb01601.x

Dewson, Z., & Death, R. (2007). *The effects of water supply intakes on macroinvertebrate communities in the Waikato Region*. Waikato Regional Council Technical Report: 2007/32. https://www.waikatoregion.govt.nz/assets/WRC/WRC-2019/tr07-32.pdf

Ding, N., Yang, W., Zhou, Y., González-Bergonzoni, I., Zhang, J., Chen, K., Vidal, N., Jeppesen, E., Liu, Z., & Wang, B. (2016). Different responses of functional traits and diversity of stream macroinvertebrates to environmental and spatial factors in the Xishuangbanna watershed of the upper Mekong River Basin, China. *The Science of the Total Environment*, *574*, 288-299. https://doi.org/10.1016/j.scitotenv.2016.09.053

Dodds, W. K. (2006). Eutrophication and trophic state in rivers and streams. *Limnology and Oceanography*, *51*(1), 671-680. https://doi.org/https://doi.org/10.4319/lo.2006.51.1_part_2.0671

Dodds, W. K., & Smith, V. H. (2016). Nitrogen, phosphorus, and eutrophication in streams. *Inland Waters*, *6*(2), 155-164. https://doi.org/10.5268/IW-6.2.909

Dolédec, S., & Chessel, D. (1994). Co-inertia analysis: an alternative method for studying species–environment relationships. *Freshwater Biology*, *31*(3), 277-294. https://doi.org/https://doi.org/10.1111/j.1365-2427.1994.tb01741.x

Dolédec, S., Phillips, N., Scarsbrook, M., Riley, R., & Townsend, C. (2006). Comparison of structural and functional approaches to determining landuse effects on grassland stream communities. *Journal of The North American Benthological Society* 25(1), 44-60. https://doi.org/10.1899/0887-3593(2006)25[44:COSAFA]2.0.CO;2

Dolédec, S., Phillips, N., & Townsend, C. (2011). Invertebrate community responses to land use at a broad spatial scale: Trait and taxonomic measures compared in New Zealand rivers. *Freshwater Biology*, *56*(8), 1670-1688. https://doi.org/10.1111/j.1365-2427.2011.02597.x

Dolédec, S., Statzner, B., & Bournard, M. (1999). Species traits for future biomonitoring across eco-regions: Patterns along a human-impacted river. *Freshwater Biology*, *42*(4), 737-758. https://doi.org/10.1046/j.1365-2427.1999.00509.x

Dorner, Z., Doole, G., Bermeo, S., Paterson, J. (2018). Economic assessment of options to mitigate sediment loss from New Zealand agriculture – in the context of managing freshwater quality. Bay of Plenty Regional Council Environmental Publication 2018/07, 61pp.

Downes, B. J., Lake, P. S., Schreiber, E. S. G., & Glaister, A. (2000). Habitat Structure, Resources and Diversity: The Separate Effects of Surface Roughness and Macroalgae on Stream Invertebrates. *Oecologia*, *123*(4), 569-581. http://www.jstor.org/stable/4222656

Elliot, S., Parshotam, A., & Wadhwa, S. (2010). *Tauranga Harbour Sediment Study: Catchment Model Results*. NIWA Client Report HAM2009-046.

Fahrig, L. (1992). Relative importance of spatial and temporal scales in a patchy environment. *Theoretical Population Biology*, *41*(3), 300-314. https://doi.org/https://doi.org/10.1016/0040-5809(92)90031-N

Fausch, K. D., Torgersen, C. E., Baxter, C. V., & Li, H. W. (2002). Landscapes to Riverscapes: Bridging the Gap between Research and Conservation of Stream Fishes: A Continuous View of the River is Needed to Understand How Processes Interacting among Scales Set the Context for Stream Fishes and Their Habitat. *BioScience*, *52*(6), 483-498. https://doi.org/10.1641/0006-3568(2002)052[0483:LTRBTG]2.0.CO;2

Fitzpatrick, F., Waite, I., D'Arconte, P., Meador, M., Maupin, M., & Gurtz, M. (1998). *Revised Methods for Characterizing Stream Habitat in the National Water Quality Assessment Program*. U.S. Geological Survey Water-Resources Investigations Report: 98-4052. https://pubs.usgs.gov/wri/wri984052/pdf/wri98-4052.pdf

Fleeger, J. W., Carman, K. R., & Nisbet, R. M. (2003). Indirect effects of contaminants in aquatic ecosystems. *Science of The Total Environment*, *317*(1-3), 207-233. https://doi.org/10.1016/s0048-9697(03)00141-4

Flint, J. J. (1974). Stream gradient as a function of order, magnitude, and discharge. *Water Resources Research*, *10*(5), 969-973. https://doi.org/https://doi.org/10.1029/WR010i005p00969

Forbes-Green, M. J., & Cyr, H. (2023). The effects of physical disturbance and sediment refuge on the growth of young native freshwater mussels Elliptio complanata (Eastern Elliptio). *Hydrobiologia*, *850*(5), 1123-1140. https://doi.org/10.1007/s10750-023-05147-0

Francoeur, S. N., & Biggs, B. J. F. (2006). Short-term Effects of Elevated Velocity and Sediment Abrasion on Benthic Algal Communities. *Hydrobiologia*, *561*(1), 59-69. https://doi.org/10.1007/s10750-005-1604-4

Frissell, C., Liss, W., Warren, C., & Hurley, M. (1986). A hierarchical framework for stream habitat classification: Viewing streams in a watershed context. *Environmental Management*, *10*(2), 199-214. https://doi.org/10.1007/BF01867358

Ghani, W. M., Rawi, C. S., Hamid, S. A., & Al-Shami, S. A. (2016). Efficiency of Different Sampling Tools for Aquatic Macroinvertebrate Collections in Malaysian Streams. *Tropical Life Sciences Research*, *27*(1), 115-133.

Ghilarov, A. M. (2000). Ecosystem Functioning and Intrinsic Value of Biodiversity. *Oikos*, *90*(2), 408-412. http://www.jstor.org/stable/3547152

Glenjarman, N. (2017). *The impact of suspended and deposited fine inorganic sediment on New Zealand Freshwater Fishes* [Masters thesis, University of Cantebury]. The University of Canterbury Library. https://ir.canterbury.ac.nz/server/api/core/bitstreams/c210a1a3-a1e1-4bf4-95ea-7d14df31b4d2/content

Goodman, J. (2018). *Conservation, ecology and of migratory galaxiids and the whitebait fishery: A summary of current knowledge and information gaps*. Department of Conservation. https://www.doc.govt.nz/globalassets/documents/conservation/land-and-freshwater/freshwater/conservation-ecology-and-management-of-migratory-galaxiids.pdf

Goodyear, K. L., & McNeill, S. (1999). Bioaccumulation of heavy metals by aquatic macro-invertebrates of different feeding guilds: a review. *Science of The Total Environment*, *229*(1), 1-19. https://doi.org/https://doi.org/10.1016/S0048-9697(99)00051-0

Graeber, D., Pusch, M. T., Lorenz, S., & Brauns, M. (2013). Cascading effects of flow reduction on the benthic invertebrate community in a lowland river. *Hydrobiologia*, *717*(1), 147-159. https://doi.org/10.1007/s10750-013-1570-1

Graham, E., & Franklin, P. (2017). *Dissolved Oxygen Threshold and Management* (2017/EXT/1564). Horizons Report No: 2017/EXT/1564. https://www.manawaturiver.co.nz/wp-content/uploads/2018/12/Horizons-Regional-Council-Dissolved-Oxygen-Thresholds-and-Management-2017.pdf

Grainger, N., Harding, J., Drinan, T., Collier, K., Smith, B., Death, R., Makan, T., & Rolfe, J. (2018). *Conservation status of New Zealand freshwater invertebrates, 2018*. New Zealand Threat Classification Series 28. Department of Conservation.

Gray, C., Baird, D. J., Baumgartner, S., Jacob, U., Jenkins, G. B., O'Gorman, E. J., Lu, X., Ma, A., Pocock, M. J., Schuwirth, N., Thompson, M., & Woodward, G. (2014). Ecological networks: the missing links in biomonitoring science. *Journal of Applied Ecology*, *51*(5), 1444-1449. https://doi.org/10.1111/1365-2664.12300

Grime, J. P. (1974). Vegetation classification by reference to strategies. *Nature*, *250*(5461), 26-31. https://doi.org/10.1038/250026a0

Gutiérrez-Rial, D., Soto González, B., García Vázquez, D., Méndez-Martínez, G., Pombal Diego, M. Á., & Garrido González, J. (2023). Freshwater biodiversity loss in urbanised rivers. *Ecological Indicators*, *156*, 111150. https://doi.org/https://doi.org/10.1016/j.ecolind.2023.111150

Hageman, K. J., Aebig, C. H. F., Luong, K. H., Kaserzon, S. L., Wong, C. S., Reeks, T., Greenwood, M., Macaulay, S., & Matthaei, C. D. (2019). Current-use pesticides in New Zealand streams: Comparing results from grab samples and three types of passive samplers. *Environmental Pollution*, *254* (Pt A), 112973. https://doi.org/https://doi.org/10.1016/j.envpol.2019.112973

Harding, J., Clapcott, J., Quinn, J., Hayes, J., Joy, M., Storey, R., Greig, H., Hay, J., James, T., Beech, M., Ozane, R., Meredith, A., & Boothroyd, I. (2009). *Stream Habitat Assessment Protocols for wadeable rivers and streams of New Zealand*. https://envirolink.govt.nz/assets/Envirolink/Stream20Habitat20Assessment20Protocols.pdf

Harding, J. S. (1994). *Lotic Eco-regions of New Zealand*. Department of Conservation Technical Series No.11. https://ir.canterbury.ac.nz/items/d5b499db-7c81-4842-94d7-be91d666456c/full

Hauer, F., & Resh, V. (2007). Macroinvertebrates. In Hauer, F. & Lamberti, G., Methods in Stream Ecology (pp. 435-454). https://doi.org/10.1016/B978-012332908-0.50028-0

Heino, J. (2008). Patterns of functional biodiversity and function-environment relationships in lake littoral macroinvertebrates. *Limnology and Oceanography*, *53*(4), 1446-1455. https://doi.org/https://doi.org/10.4319/lo.2008.53.4.1446

Heino, J., Melo, A. S., Siqueira, T., Soininen, J., Valanko, S., & Bini, L. M. (2015). Metacommunity organisation, spatial extent and dispersal in aquatic systems: patterns, processes and prospects. *Freshwater Biology*, *60*(5), 845-869. https://doi.org/https://doi.org/10.1111/fwb.12533

Hladik, M., & Kuivila, K. (2009). Assessing the Occurrence and Distribution of Pyrethroids in Water and Suspended Sediments. *Journal of agricultural and food chemistry*, *57*(19), 9079-9085. https://doi.org/10.1021/jf9020448

Holt, E., A.,, & Miller, S., W. (2010). Bioindicators: Using Organisms to measure Environment Impacts. *Nature Education Knowledge*, 2(2) 8pp. https://www.nature.com/scitable/knowledge/library/bioindicators-using-organisms-to-measure-environmental-impacts-16821310/

Houpt, E. (2022). Bay of Plenty's population growth second highest in the country. *Bay of Plenty Times*. https://www.nzherald.co.nz/bay-of-plenty-times/news/bay-of-plentys-population-growth-second-highest-in-country/TAWXF3BAHKY7SIAJCHWEUPGIRA/

Hume, T., M.,, Green, M., O., & Elliott, S. (2010a). *Tauranga Harbour Sediment Study: Assessment of predictions for management*. NIWA Client Report: HAM2009-139. NIWA. https://atlas.boprc.govt.nz/api/v1/edms/document/A3888411/content

Hume, T., M.,, Green, M., O.,, & Elliott, S. (2010b). *Tauranga Harbour Sediment Study: Assessment of predictions for management*. NIWA Client Report No: HAM2009-139. https://atlas.boprc.govt.nz/api/v1/edms/document/A3888411/content

Irvine, J. R., & Henriques, P. R. (1984). A preliminary investigation on effects of fluctuating flows on invertebrates of the Hawea River, a large regulated river in New Zealand. *New Zealand Journal of Marine and Freshwater Research*, *18*(3), 283-290. https://doi.org/10.1080/00288330.1984.9516050

Jackson, M. C., Pawar, S., & Woodward, G. (2021). The Temporal Dynamics of Multiple Stressor Effects: From Individuals to Ecosystems. *Trends in Ecology & Evolution*, *36*(5), 402-410. https://doi.org/10.1016/j.tree.2021.01.005

Jha, S. K., Negi, A. K., & Alatalo, J. M. (2022). Quantification of ecosystem services providing socio-economic benefits to customary owners of natural resources in Pauri, western Himalaya. *Current Research in Environmental Sustainability*, *4* (2022). https://doi.org/https://doi.org/10.1016/j.crsust.2021.100121

Johnson, W. C. (1994). Woodland Expansions in the Platte River, Nebraska: Patterns and Causes. *Ecological Monographs*, *64*(1), 45-84. https://doi.org/10.2307/2937055

Jonsson, M., Burrows, R. M., Lidman, J., Fältström, E., Laudon, H., & Sponseller, R. A. (2017). Land use influences macroinvertebrate community composition in boreal headwaters through altered stream conditions. *Ambio*, *46*(3), 311-323. https://doi.org/10.1007/s13280-016-0837-y

Jowett, I., G.,, Hayes, J., W.,, & Ducan, M., J. (2008). *A guide to instream habitat survey methods and analysis* (54). NIWA Science and Technology Series No. 54. https://niwa.co.nz/sites/niwa.co.nz/files/a_guide_to_instream_habitat_survey_methods_and_analysis.pdf

Jowett, I., & Richardson, J. (1990). Microhabitat preferences of benthic invertebrates in a New Zealand river and the development of in-stream flow-habitat models for Deleatidium spp. *New Zealand Journal of Marine and Freshwater Research*, *24*(1), 19-30.

Joy, M. K., & Death, R. G. (2013). Freshwater biodiversity. In Dymond, J., (Ed.), *Ecosystem services in New Zealand: conditions and trends* (p. 448-459). Manaaki Whenua Landcare Research.

Judi Hewitt, Silvia De Juan, Drew Lohrer, Micheal Townsend, & Roberta D' Archino. (2014). *Functional traits as indicators of ecological integrity.* NIWA Client Report No: HAM2014-001.

Julian, J. P., de Beurs, K. M., Owsley, B., Davies-Colley, R. J., & Ausseil, A. G. E. (2017). River water quality changes in New Zealand over 26 years: response to land use intensity. *Hydrology and Earth System Sciences Disscussions, 21*(2), 1149-1171. https://doi.org/10.5194/hess-21-1149-2017

Keke, U. N., Omoigberale, M. O., Ezenwa, I., Yusuf, A., Biose, E., Nweke, N., Edegbene, A. O., & Arimoro, F. O. (2021). Macroinvertebrate communities and physicochemical characteristics along an anthropogenic stress gradient in a southern Nigeria stream: Implications for ecological restoration. *Environmental and Sustainability Indicators*, *12*. https://doi.org/https://doi.org/10.1016/j.indic.2021.100157

Kirkeby, L. (2018). Residents reassured after popular South Waikato stream poisoned twice in one year. *Stuff*. https://www.stuff.co.nz/environment/100792781/residents-reassured-after-popular-stream-poisoned-twice-in-one-year

Knillmann, S., & Liess, M. (2019). Pesticide Effects on Stream Ecosystems. In M. Schröter, A. Bonn, S. Klotz, R. Seppelt, & C. Baessler, *Atlas of Ecosystem Services: Drivers, Risks, and Societal Responses* (pp. 211-214). Springer International Publishing. https://doi.org/10.1007/978-3-319-96229-0_33

Kondolf, G. M., & Curry, R. R. (1986). Channel erosion along the Carmel River, Monterey County, California. *Earth Surface Processes and Landforms*, *11*(3), 307-319. https://doi.org/https://doi.org/10.1002/esp.3290110308

Kovalenko, K. E., Brady, V. J., Ciborowski, J. J., Ilyushkin, S., & Johnson, L. B. (2014). Functional changes in littoral macroinvertebrate communities in response to watershed-level anthropogenic stress. *PLOS ONE*, *9*(7), e101499. https://doi.org/10.1371/journal.pone.0101499

Krajenbrink, H. J., White, J. C., Dunbar, M. J., & Wood, P. J. (2022). Macroinvertebrate and diatom community responses to thermal alterations below water supply reservoirs. *River Research and Applications*, *38*(3), 595-612. https://doi.org/https://doi.org/10.1002/rra.3922

Kunz, S., Kefford, B. J., Schmidt-Kloiber, A., Matthaei, C. D., Usseglio-Polatera, P., Graf, W., Poff, N. L., Metzeling, L., Twardochleb, L., Hawkins, C. P., & Schäfer, R. B. (2022). Tackling inconsistencies among freshwater invertebrate trait databases: harmonising across continents and aggregating taxonomic resolution. *Freshwater Biology*, *67*(2), 275-291. https://doi.org/https://doi.org/10.1111/fwb.13840

Lake, P. S. (2000). Disturbance, patchiness, and diversity in streams. *Journal of the North American Benthological Society*, *19*(4), 573-592. https://doi.org/10.2307/1468118

Laliberté, E., & Legendre, P. (2010). A distance-based framework for measuring functional diversity from multiple traits. *Ecology*, *91*(1), 299-305. https://doi.org/https://doi.org/10.1890/08-2244.1

Landman, M. J., Van Den Heuvel, M. R., & Ling, N. (2005). Relative sensitivities of common freshwater fish and invertebrates to acute hypoxia. *New Zealand Journal of Marine and Freshwater Research*, *39*(5), 1061-1067. https://doi.org/10.1080/00288330.2005.9517375

Lange, K., Townsend, C. R., & Matthaei, C. D. (2014). Can biological traits of stream invertebrates help disentangle the effects of multiple stressors in an agricultural catchment? *Freshwater Biology*, *59*(12), 2431-2446. https://doi.org/https://doi.org/10.1111/fwb.12437

Larned, S. T., Booker, D., Dudley, B., Moore, S., Monaghan, R., Baillie, B., Schallenberg, M., Moriarty, E., Zeldis, J., & Short, K. (2018). *Land-use impacts on freshwater and marine environments in New Zealand. NIWA.* https://environment.govt.nz/assets/Publications/Files/land-use-impacts-on-freshwater-and-marine-environments-.pdf

Larned, S. T., Booker, D., Dudley, B., Moores, J., Monaghan, R., Baillie, B., Schallenberg, M., Moriarty, E., Zeldis, J., & Short, K. (2018). *Land-use impacts on freshwater and marine environments in New Zealand*. NIWA. https://environment.govt.nz/assets/Publications/Files/land-use-impacts-on-freshwater-and-marine-environments-.pdf

Larned, S. T., Moores, J., Gadd, J., Baillie, B., & Schallenberg, M. (2019). Evidence for the effects of land use on freshwater ecosystems in New Zealand. *New Zealand Journal of Marine and Freshwater Research*, *54*(3), 551-591. https://doi.org/10.1080/00288330.2019.1695634

Laskowski, D. A. (2002). Physical and chemical properties of pyrethroids. *Review of Environmental Contamination and Toxicology*, *174*, 49-170. https://doi.org/10.1007/978-1-4757-4260-2_3

Laureto, L. M. O., Cianciaruso, M. V., & Samia, D. S. M. (2015). Functional diversity: an overview of its history and applicability. *Natureza & Conservação*, *13*(2), 112-116. https://doi.org/https://doi.org/10.1016/j.ncon.2015.11.001

LAWA. (2018). *Bay of Plenty Region: Land Cover*. Land Air and Water Aotearoa. Retrieved 20/08/23 from https://www.lawa.org.nz/explore-data/land-cover/

LAWA. (2021). *LAWA River Water Quality National Picture Summary 2021*. LAWA. https://www.lawa.org.nz/explore-data/river-quality/

LAWA. (2023a). *Bay of Plenty region: Land cover*. Land Air and Water Aotearoa. Retrieved 23/09/23 from https://www.lawa.org.nz/explore-data/land-cover/

LAWA. (2023b). *River quality: New Zealand*. Land Air and Water Aotearoa. https://www.lawa.org.nz/explore-data/river-quality/#/tb-region

Lee, R. H., & Guénard, B. (2019). Choices of sampling method bias functional components estimation and ability to discriminate assembly mechanisms. *Methods in Ecology and Evolution*, *10*(6), 867-878. https://doi.org/https://doi.org/10.1111/2041-210X.13175

Lemly, A. D. (1982). Modification of benthic insect communities in polluted streams: combined effects of sedimentation and nutrient enrichment. *Hydrobiologia*, *87*(3), 229-245. https://doi.org/10.1007/BF00007232

Leopold, L. B. (1968). *Hydrology for urban land planning - A guidebook on the hydrologic effects of urban land use*. Geological Survey Circular: 554.

LGNZ. (2023). *Local Government in New Zealand*. https://www.lgnz.co.nz/local-government-in-nz/#:~:text=The%20local%20government%20sector%20consists,authorities%20with%20regional%20council%20responsibilities.

Li, J., Herlihy, A., Gerth, W., Kaufmann, P., Gregory, S., Urquhart, S., & Larsen, D. P. (2001). Variability in stream macroinvertebrates at multiple spatial scales. *Freshwater Biology*, *46*(1), 87-97. https://doi.org/https://doi.org/10.1111/j.1365-2427.2001.00628.x

Li, Z., Heino, J., Chen, X., Liu, Z., Meng, X., Jiang, X., Ge, Y., Chen, J., & Xie, Z. (2021). Understanding macroinvertebrate metacommunity organization using a nested study design across a mountainous river network. *Ecological Indicators*, *121*, 107188. https://doi.org/https://doi.org/10.1016/j.ecolind.2020.107188

Ling, N. (2019). *Toxicological assessment of Oraka Stream crayfish mortalities in December 2017*. The University of Waikato.

Liu, G., Lin, Z., Qi, X., Wang, Y., Wang, Y., Jiang, W., He, F., & Wu, N. (2023). Environmental filtering, spatial processes and biotic interactions jointly shape different traits communities of stream macroinvertebrates. *Frontiers in Ecology and Evolution*, *11*. https://doi.org/10.3389/fevo.2023.1196296

Liu, W., Gan, J. J., Lee, S., & Kabashima, J. N. (2004). Phase distribution of synthetic pyrethroids in runoff and stream water. *Biologia 23*(1), 7-11. https://doi.org/10.1897/03-183

Local Government Act 2002, s. 5. (2002). New Zealand Legislation. https://www.legislation.govt.nz/act/public/2002/0084/latest/whole.html#DLM170881

Lu, A., Li, J., Zheng, B., & Yin, X. (2022). Effect of Different Land Use Types on the Taxonomic and Functional Diversity of Macroinvertebrates in an Urban Area of Northern China. *Water*, *14*(23).

Lu, Y., Wang, R., Zhang, Y., Su, H., Wang, P., Jenkins, A., Ferrier, R. C., Bailey, M., & Squire, G. (2015). Ecosystem health towards sustainability. *Ecosystem Health and Sustainability*, *1*(1), 1-15. https://doi.org/10.1890/EHS14-0013.1

Lu, Z., Gan, J., Cui, X., Delgado-Moreno, L., & Lin, K. (2019). Understanding the bioavailability of pyrethroids in the aquatic environment using chemical approaches. *Environment International*, *129*, 194-207. https://doi.org/https://doi.org/10.1016/j.envint.2019.05.035

Mac Loughlin, T. M., Peluso, M. L., & Marino, D. J. G. (2022). Multiple pesticides occurrence, fate, and environmental risk assessment in a small horticultural stream of Argentina. *Science of The Total Environment*, *802*, 149893. https://doi.org/https://doi.org/10.1016/j.scitotenv.2021.149893

Magbanua, F., Townsend, C., Blackwell, G., Phillips, N., & Matthaei, C. (2010). Responses of stream macroinvertebrates and ecosystem function to conventional, integrated and organic farming. *Journal of Applied Ecology*, *47*, 1014-1025. https://doi.org/10.1111/j.1365-2664.2010.01859.x

Magbanua, F. S., Townsend, C. R., Hageman, K. J., & Matthaei, C. D. (2013). Individual and combined effects of fine sediment and the herbicide glyphosate on benthic macroinvertebrates and stream ecosystem function. *Freshwater Biology*, *58*(8), 1729-1744. https://doi.org/https://doi.org/10.1111/fwb.12163

Mahler, B. J., Schmidt, T. S., Nowell, L. H., Qi, S. L., Van Metre, P. C., Hladik, M. L., Carlisle, D. M., Munn, M. D., & May, J. (2020). Biofilms Provide New Insight into Pesticide Occurrence in Streams and Links to Aquatic Ecological Communities. *Environmental Science & Technology*, *54*(9), 5509-5519. https://doi.org/10.1021/acs.est.9b07430

Makiola, A., Compson, Z. G., Baird, D. J., Barnes, M. A., Boerlijst, S. P., Bouchez, A., Brennan, G., Bush, A., Canard, E., Cordier, T., Creer, S., Curry, R. A., David, P., Dumbrell, A. J., Gravel, D., Hajibabaei, M., Hayden, B., van der Hoorn, B., Jarne, P., Bohan, D. A. (2020). Key Questions for Next-Generation Biomonitoring [Review]. *Frontiers in Environmental Science, 7*. https://doi.org/10.3389/fenvs.2019.00197

Manktelow, D., Stevens, P., Walker, J., S, G., Park, N. M., Zabkiewicz, J., Teulon, D., & A, R. (2005). *Trends in Pesticide Use in New Zealand: 2004*. Report to the Ministry for the Environment Report. Project SMF4193. HortResearch Client Report No. 17962.

Mason, N. W. H., Mouillot, D., Lee, W. G., & Wilson, J. B. (2005). Functional richness, functional evenness and functional divergence: the primary components of functional diversity. *Oikos*, *111*(1), 112-118. https://doi.org/https://doi.org/10.1111/j.0030-1299.2005.13886.x

Mathers, K. L., Doretto, A., Fenoglio, S., Hill, M. J., & Wood, P. J. (2022). Temporal effects of fine sediment deposition on benthic macroinvertebrate community structure, function and biodiversity likely reflects landscape setting. *Science of The Total Environment*, *829*, 154612. https://doi.org/https://doi.org/10.1016/j.scitotenv.2022.154612

Mathers, K. L., White, J. C., Fornaroli, R., & Chadd, R. (2020). Flow regimes control the establishment of invasive crayfish and alter their effects on lotic macroinvertebrate communities. *Journal of Applied Ecology*, *57*(5), 886-902. https://doi.org/https://doi.org/10.1111/1365-2664.13584

Matthaei, C., & Piggott, J. (2019). Multiple Stressors in Australia and New Zealand: Key Stressors and Interactions. In Sabater et al., *Multiple Stressors in River Ecosystems* (pp. 221-233). https://doi.org/10.1016/B978-0-12-811713-2.00013-3

Matthaei, C. D., Piggott, J. J., & Townsend, C. R. (2010). Multiple stressors in agricultural streams: interactions among sediment addition, nutrient enrichment and water abstraction. *Journal of Applied Ecology*, *47*(3), 639-649. https://doi.org/https://doi.org/10.1111/j.1365-2664.2010.01809.x

Mayfield, A. E., Seybold, S. J., Haag, W. R., Johnson, M. T., Kerns, B. K., Kilgo, J. C., Larkin, D. J., Lucardi, R. D., Moltzan, B. D., Pearson, D. E., Rothlisberger, J. D., Schardt, J. D., Schwartz, M. K., & Young, M. K. (2021). Impacts of Invasive Species in Terrestrial and Aquatic Systems in the United States. In T. M. Poland, T. Patel-Weynand, D. M. Finch, C. F. Miniat, D. C. Hayes, & V. M. Lopez (Eds.), *Invasive Species in Forests and Rangelands of the United States: A Comprehensive Science Synthesis for the United States Forest Sector* (pp. 5-39). Springer International Publishing. https://doi.org/10.1007/978-3-030-45367-1_2

McGill, B. J., Enquist, B. J., Weiher, E., & Westoby, M. (2006). Rebuilding community ecology from functional traits. *Trends in Ecology & Evolution*, *21*(4), 178-185. https://doi.org/https://doi.org/10.1016/j.tree.2006.02.002

McKenzie, J. S. (2014). *Predicted hydrodynamic and sediment transport impacts of breakwater construction in Tauranga Harbour, New Zealand* [Masters thesis, University of Waikato].

McKenzie, M., Brooks, A., Callisto, M., Collins, A. L., Durkota, J. M., Death, R. G., Jones, J. I., Linares, M. S., Matthaei, C. D., Monk, W. A., Murphy, J. F., Wagenhoff, A., Wilkes, M., Wood, P. J., & Mathers, K. L. (2024). Freshwater invertebrate responses to fine sediment stress: A multi-continent perspective. *Global Change Biology*, *30*(1). https://doi.org/https://doi.org/10.1111/gcb.17084

McKenzie, M., England, J., Foster, I., & Wilkes, M. (2022a). Evaluating the performance of taxonomic and trait-based biomonitoring approaches for fine sediment in the UK. *Ecological Indicators*, *134*, 108502. https://doi.org/https://doi.org/10.1016/j.ecolind.2021.108502

McKenzie, M., England, J., Foster, I. D. L., & Wilkes, M. A. (2022b). Abiotic predictors of fine sediment accumulation in lowland rivers. *International Journal of Sediment Research*, *37*(1), 128-137. https://doi.org/https://doi.org/10.1016/j.ijsrc.2021.06.003

McKenzie, M., Mathers, K. L., Wood, P. J., England, J., Foster, I., Lawler, D., & Wilkes, M. (2020). Potential physical effects of suspended fine sediment on lotic macroinvertebrates. *Hydrobiologia*, *847*(3), 697-711. https://doi.org/10.1007/s10750-019-04131-x

McKinnon, M. (2016). *The Bay of Plenty Region*. The New Zealand Government. https://teara.govt.nz/en/bay-of-plenty-region

Medupin, C. (2020). Spatial and temporal variation of benthic macroinvertebrate communities along an urban river in Greater Manchester, UK. *Environmental Monitoring and Assessment*, *192*(2), 84. https://doi.org/10.1007/s10661-019-8019-6

Menezes, S., Baird, D., & Soares, A. (2010). Beyond taxonomy: A review of macroinvertebrate trait-based community descriptors as tools for freshwater biomonitoring. *Journal of Applied Ecology*, *47*, 711-719. https://doi.org/10.1111/j.1365-2664.2010.01819.x

Merritt, R. W., & Wallace, J. B. (1981). Filter-feeding Insects. *Scientific American*, *244*(4), 132-147. http://www.jstor.org/stable/24964386

Milardi, M., Gavioli, A., Soininen, J., & Castaldelli, G. (2019). Exotic species invasions undermine regional functional diversity of freshwater fish. *Scientific Reports*, *9*(1). https://doi.org/10.1038/s41598-019-54210-1

Miller, A., Weiss, S., Pickles, S., & Zaman, N. (2015). *Pollution prevention and compliance report*. Environmental Publication 2015/09. Bay of Plenty Regional Council. https://www.boprc.govt.nz/media/520780/2014_2015-pollution-prevention-and-compliance-report.pdf

Milne, J., Suren, A., Williams, A., Smith, B., Nicholson, C., Gray, D., Harrison, E., & Ellery, G. (2022). *National Environmental Monitoring Standards*. National Environmental Monitoring Standards. https://www.nems.org.nz/documents/macroinvertebrates/

Ministry for the Environment. (2002). *Towards a Pesticides Risk Reduction Policy for New Zealand*. https://www.oecd.org/newzealand/40804999.pdf

Ministry for the Environment. (2010). *River Environment Classification Bay of Plenty*. Ministry for the Environment. https://data.mfe.govt.nz/layer/51825-river-environment-classification-bay-of-plenty-2010-deprecated/

Ministry for the Environment & Stats NZ (2022).*New Zealand Environmental Reporting Series: Environment Aotearoa 2022.* https://environment.govt.nz/assets/publications/Environmental-Reporting/environment-aotearoa-2022.pdf

Ministry for the Environment. (2023). *New Zealand Environmental Reporting Series: Our Freshwater.* https://environment.govt.nz/publications/our-freshwater-2023/

Ministry For the Environment & Ministry for Primary Industries (2020). *Te Mana o te Wai factsheet*. https://environment.govt.nz/assets/Publications/Files/essential-freshwater-te-mana-o-te-wai-factsheet.pdf

Ministry for the Environment, & NZ, S. (2020). *New Zealand Environmental Reporting Series: Our Freshwater*. https://environment.govt.nz/assets/Publications/Files/our-freshwater-2020.pdf

Moldoveanu, S., & David, V. (2021). Chapter 5 - Phase transfer in sample preparation. In S. Moldoveanu & V. David (Eds.), *Modern Sample Preparation for Chromatography (Second Edition)* (pp. 151-190). Elsevier. https://doi.org/https://doi.org/10.1016/B978-0-12-821405-3.00011-3

Moon, V. G. (1993). Geotechnical characteristics of ignimbrite: A soft pyroclastic rock type. *Engineering Geology*, *35*(1), 33-48. https://doi.org/https://doi.org/10.1016/0013-7952(93)90068-N

Mouchet, M. A., Villéger, S., Mason, N. W. H., & Mouillot, D. (2010). Functional diversity measures: an overview of their redundancy and their ability to discriminate community assembly rules. *Functional Ecology*, *24*(4), 867-876. https://doi.org/https://doi.org/10.1111/j.1365-2435.2010.01695.x

Mouillot, D., Graham, N., Villéger, S., Mason, N., & Bellwood, D. (2012). A functional approach reveals community responses to disturbance. *Trends in Ecology & Evolution*, *28*(3), 167-177. https://doi.org/10.1016/j.tree.2012.10.004

Mouillot, D., Spatharis, S., Reizopoulou, S., Thierry, L., Sabetta, L., Basset, A., & Chi, T. (2006). Alternatives to taxonomic-based approaches to assess changes in transitional water communities. *Aquatic Conservation Marine and Freshwater Ecosystems*, *16*, 469-482. https://doi.org/10.1002/aqc.769

Murray, A. (2002). *Methods of assessment of macroinvertebrate biodiversity in New Zealand streams* [Masters thesis, Massey University]. Massey University Library. https://mro.massey.ac.nz/handle/10179/7234

Myslinski, E., & Ginsburg, W. (1977). Macroinvertebrates as Indicators of Pollution. *Journal (American Water Works Association)*, *69*(10), 538-544. http://www.jstor.org/stable/41268824

Phillips, N. & Reid, D. (2012a). *Biological trait analysis: application to the Waikato Regional Council Monitoring Programme*. Waikato Regional Council Technical Report 2012/04. https://www.waikatoregion.govt.nz/assets/WRC/WRC-2019/TR201204.pdf

Phillips, N. & Reid, D. (2012b). *Biological trait analysis: application to the Auckland Council River Monitoring Programme*. Auckland Council Technical Report 2012/001. Waikato Regional Council. https://knowledgeauckland.org.nz/media/1678/tr2012-001-biological-traits-application-to-auckland-council-river-ecology-monitoring-programme.pdf

Naden, P. S., Murphy, J. F., Old, G. H., Newman, J., Scarlett, P., Harman, M., Duerdoth, C. P., Hawczak, A., Pretty, J. L., Arnold, A., Laizé, C., Hornby, D. D., Collins, A. L., Sear, D. A., & Jones, J. I. (2016a). Understanding the controls on deposited fine sediment in the streams of agricultural catchments. *Science of The Total Environment*, *547*, 366-381. https://doi.org/https://doi.org/10.1016/j.scitotenv.2015.12.079

Naden, P. S., Murphy, J. F., Old, G. H., Newman, J., Scarlett, P., Harman, M., Duerdoth, C. P., Hawczak, A., Pretty, J. L., Arnold, A., Laizé, C., Hornby, D. D., Collins, A. L., Sear, D. A., & Jones, J. I. (2016b). Understanding the controls on deposited fine sediment in the streams of agricultural catchments. *Science of the Total Environment*, *547*, 366-381. https://doi.org/10.1016/j.scitotenv.2015.12.079

Narr, C., & Krist, A. (2019). Improving estimates of richness, habitat associations, and assemblage characteristics of freshwater gastropods. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *30*(1). https://doi.org/10.1002/aqc.3247

National Policy Statement for Freshwater Management 2020.

Ngāi Te Rangi Iwi. (2023). *About*. Ngāi Te Rangi Iwi. https://www.ngaiterangi.com/about.html

Ngāti Pūkenga. (2023). *Our History*. Ngāti Pūkenga. https://ngatipukenga.co.nz/our-history

NIWA. (2020). *SHMAK: Stream health monitoring and asssement kit user manual*. NIWA. https://niwa.co.nz/static/web/SHMAK_Manual.pdf

Niyogi, D. K., Koren, M., Arbuckle, C. J., & Townsend, C. R. (2007). Longitudinal changes in biota along four New Zealand streams: Declines and improvements in stream health related to land use. *New Zealand Journal of Marine and Freshwater Research*, *41*(1), 63-75. https://doi.org/10.1080/00288330709509896

Niyogi, D. K., Koren, M., Arbuckle, C. J., & Townsend, C. R. (2007). Stream communities along a catchment land-use gradient: subsidy-stress responses to pastoral development. *Environmental Management*, *39*(2), 213-225. https://doi.org/10.1007/s00267-005-0310-3

Nock, C. A., Vogt, R. J., & Beisner, B. E. (2016). Functional Traits. In *Encyclopedia of Life Sciences* (p1-8). https://doi.org/https://doi.org/10.1002/9780470015902.a0026282

Ntloko, P., Palmer, C. G., Akamagwuna, F. C., & Odume, O. N. (2021). Exploring Macroinvertebrates Ecological Preferences and Trait-Based Indicators of Suspended Fine Sediment Effects in the Tsitsa River and Its Tributaries, Eastern Cape, South Africa. *Water*, *13*(6).

O'Brien, A., Townsend, K., Hale, R., Sharley, D., & Pettigrove, V. (2016). How is ecosystem health defined and measured? A critical review of freshwater and estuarine studies. *Ecological Indicators*, *69*, 722-729. https://doi.org/https://doi.org/10.1016/j.ecolind.2016.05.004

Oliveira, P. C., Kraak, M. H. S., Pena-Ortiz, M., van der Geest, H. G., & Verdonschot, P. F. M. (2020). Responses of macroinvertebrate communities to land use specific sediment food and habitat characteristics in lowland streams. *Science of The Total Environment*, *703*, 135060. https://doi.org/https://doi.org/10.1016/j.scitotenv.2019.135060

Osmundson, D. B., Ryel, R. J., Lamarra, V. L., & Pitlick, J. (2002). Flow-Sediment Biota relations: Implications for river regulation, effects on native fish abundance. *Ecological Applications*, *12*(6), 1719-1739. https://doi.org/https://doi.org/10.1890/1051-0761(2002)012[1719:FSBRIF]2.0.CO;2

Pardo, I., & García, L. (2016). Water abstraction in small lowland streams: Unforeseen hypoxia and anoxia effects. *Science of The Total Environment*, *568*, 226-235. https://doi.org/10.1016/j.scitotenv.2016.05.218

Parmar, T. K., Rawtani, D., & Agrawal, Y. K. (2016). Bioindicators: the natural indicator of environmental pollution. *Frontiers in Life Science*, *9*(2), 110-118. https://doi.org/10.1080/21553769.2016.1162753

Parshotam, A., Hume, T., Elliott, A., & Green, M. (2008). *Tauranga Harbour sediment study. Specification of scenarios.* NIWA Client Report: HAM2008-117. https://www.researchgate.net/publication/273576252_Tauranga_Harbour_sediment_study_Specification_of_scenarios_Prepared_for_Environment_Bay_of_Plenty

Parsons, M., Thoms, M., & Noris, R. H. (2003). Scales of Macroinvertebrate Distribution in Relation to the Hierarchical Organization of River Systems. *Journal of the North American Benthological Society*, *22*, 105-122. https://doi.org/10.2307/1467981

Phillips, N., & Smith, B. (2018). *New Zealand Freshwater Macroinvertebrate Trait Database*. NIWA Client Report No: 201879HN.

Piano, E., Doretto, A., Mammola, S., Falasco, E., Fenoglio, S., & Bona, F. (2020). Taxonomic and functional homogenisation of macroinvertebrate communities in recently intermittent Alpine watercourses. *Freshwater Biology*, *65*(12), 2096-2107. https://doi.org/https://doi.org/10.1111/fwb.13605

Pingram, M., Collier, K., Hamer, M., David, B., & Williams, A. (2019). Improving region-wide ecological condition of wadeable streams: Risk analyses highlight key stressors for policy and management. *Environmental Science & Policy*, *92*, 170-181. https://doi.org/10.1016/j.envsci.2018.11.028

Pinna, M., Gjoni, V., Bozzeda, F., Marini, G., Cristiano, G., & Di Sabatino, A. (2023). How Do Sampling Methods Affect the Body Size–Abundance Relationships of Benthic Macroinvertebrates in Freshwater Ecosystems? A Comparative Analysis. *Water*, *15*(3).

Pisani, O., Bosch, D. D., Coffin, A. W., Endale, D. M., Liebert, D., & Strickland, T. C. (2020). Riparian land cover and hydrology influence stream dissolved organic matter composition in an agricultural watershed. *Science of The Total Environment*, *717*, 137165. https://doi.org/https://doi.org/10.1016/j.scitotenv.2020.137165

Poff, N., Olden, J., Vieira, N., Finn, D., Simmons, M., & Kondratieff, B. (2006). Functional Trait Niches of North American Lotic Insects: Traits-Based Ecological Applications in Light of Phylogenetic Relationships. *Journal of The North American Benthological Society 25*, 730-755. https://doi.org/10.1899/0887-3593(2006)025[0730:FTNONA]2.0.CO;2

Poff, N. L. (1997). Landscape Filters and Species Traits: Towards Mechanistic Understanding and Prediction in Stream Ecology. *Journal of the North American Benthological Society*, *16*(2), 391-409. https://doi.org/10.2307/1468026

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., Sparks, R. E., & Stromberg, J. C. (1997). The Natural Flow Regime. *BioScience*, *47*(11), 769-784. https://doi.org/10.2307/1313099

Pollard, A., & Yuan, L. (2009). Assessing the consistency of response metrics of the invertebrate benthos: A comparison of trait- and identity-based measures. *Freshwater Biology*, *55*, 1420-1429. https://doi.org/10.1111/j.1365-2427.2009.02235.x

Prestegaard, K. L. (1988). Morphological controls on sediment delivery pathways. In *Sediment Budgets* (pp. 533-540). International Association of Hydrological Science.

Pulliam, H. R. (1988). Sources, Sinks, and Population Regulation. *The American Naturalist*, *132*(5), 652-661. http://www.jstor.org/stable/2461927

Quinn, J. (2000). Effects of pastoral development. In Barmuta, L., Collier, K., & Winterbourn, M. New Zealand stream invertebrates: ecology and implications for management (pp. 208-229). Caxton Press, Christchurch.

Quinn, J., & Stroud, M. (2002). Water Quality and Sediment and Nutrient Export From New Zealand Hill-Land Catchments of Contrasting Land Use. *New Zealand Journal of Marine and Freshwater Research*, *36*(2), 409-429. https://doi.org/10.1080/00288330.2002.9517097

Quinn, J. M., Boothroyd, I. K. G., & Smith, B. J. (2004). Riparian buffers mitigate effects of pine plantation logging on New Zealand streams: 2. Invertebrate communities. *Forest Ecology and Management*, *191*(1), 129-146. https://doi.org/https://doi.org/10.1016/j.foreco.2003.11.013

Quinn, J. M., Cooper, A. B., Davies-Colley, R. J., Rutherford, J. C., & Williamson, R. B. (1997). Land use effects on habitat, water quality, periphyton, and benthic invertebrates in Waikato, New Zealand, hill-country streams. *New Zealand Journal of Marine and Freshwater Research*, *31*(5), 579-597. https://doi.org/10.1080/00288330.1997.9516791

Rajmohan, K. S., Chandrasekaran, R., & Varjani, S. (2020). A Review on Occurrence of Pesticides in Environment and Current Technologies for Their Remediation and Management. *Indian Journal of Microbiology*, *60*(2), 125-138. https://doi.org/10.1007/s12088-019-00841-x

Ranginui, N. (2023). *Our Iwi*. Ngāti Ranginui. https://ranginui.co.nz/our-iwi/

Rao, C. R. (1982). Diversity and dissimilarity coefficients: A unified approach. *Theoretical Population Biology*, *21*(1), 24-43. https://doi.org/https://doi.org/10.1016/0040-5809(82)90004-1

Reynaga, M. (2013). Contrasting taxonomical and functional responses of stream invertebrates across space and time in a Neotropical basin. *Fundamental and Applied Limnology / Archiv für Hydrobiologie*, *183*(2), 121-133. https://doi.org/10.1127/1863-9135/2013/0501

Rideout, N. K., Compson, Z. G., Monk, W. A., Bruce, M. R., Hajibabaei, M., Porter, T. M., Wright, M. T. G., & Baird, D. J. (2022). Environmental filtering of macroinvertebrate traits influences ecosystem functioning in a large river floodplain. *Functional Ecology*, *36*(11), 2791-2805. https://doi.org/https://doi.org/10.1111/1365-2435.14168

Rijkse, C. W., & Guinto, F. D. (2010). *Soils of the Bay of Plenty*. Bay of Plenty Regional Council Environmental Publication: 2012/11-2. Bay of Plenty Regional Council. https://atlas.boprc.govt.nz/api/v1/edms/document/A472925/content

Riley, R., Townsend, C., Niyogi, D., Arbuckle, C., & Peacock, K. (2003). Headwater Stream Response to Grassland Agricultural Development in New Zealand. *New Zealand Journal of Marine and Freshwater Research - New Zealand Journal of Marine and Freshwater Research*, *37*(2), 389-403. https://doi.org/10.1080/00288330.2003.9517175

Rowe, D. K., & Dean, T. L. (1998). Effects of turbidity on the feeding ability of the juvenile migrant stage of six New Zealand freshwater fish species. *New Zealand Journal of Marine and Freshwater Research*, *32*(1), 21-29. https://doi.org/10.1080/00288330.1998.9516803

Ruru, I., Kanz, W., Afoa, E., Clarke, C., Nutsford, D., Lowe, M., A, J., & Shivnan, S. (2022). *Implementing Mahinga Kai as a Māori Freshwater Value: A kete for implementing mahinga kai in the context of the National Policy Statement for Freshwater Management 2020*. Prepared by Maumahara Consultancy Services Ltd, Awamona Ltd, Morphum Environmental Ltd and Tektus Consultants Ltd for the Ministry For the Environment. https://environment.govt.nz/assets/publications/Implementing-mahinga-kai-as-a-Maori-freshwater-value.pdf

Rutt, G. P., Weatherley, N. S., & Ormerod, S. J. (1989). Microhabitat availability in Welsh moorland and forest streams as a determinant of macroinvertebrate distribution. *Freshwater Biology*, *22*(2), 247-261. https://doi.org/https://doi.org/10.1111/j.1365-2427.1989.tb01098.x

Sanpera-Calbet, I., Chauvet, E., & Richardson, J. (2012). Fine sediment on leaves: Shredder removal of sediment does not enhance fungal colonisation. *Aquatic Sciences*, *74*(3). https://doi.org/10.1007/s00027-011-0245-0

Scarsbrook, M., McIntosh, A., Wilcock, B., & Matthaei, C. (2016). *Effects of agriculture on water quality*. In Jellyman, P., Davie T., Pearson, C., & Harding, J. Advances in New Zealand Freshwater Sciences. New Zealand Freshwater Sciences Society & New Zealand Hydrological Society.

Scarsbrook, M. R., & Townsend, C. R. (1993). Stream community structure in relation to spatial and temporal variation: a habitat templet study of two contrasting New Zealand streams. *Freshwater Biology*, *29*(3), 395-410. https://doi.org/https://doi.org/10.1111/j.1365-2427.1993.tb00774.x

Schmera, D., Heino, J., & Podani, J. (2022). Characterising functional strategies and trait space of freshwater macroinvertebrates. *Scientific Reports*, *12*(1). Nature. https://doi.org/10.1038/s41598-022-16472-0

Schmera, D., Heino, J., Podani, J., Eros, T., & Dolédec, S. (2017). Functional diversity: a review of methodology and current knowledge in freshwater macroinvertebrate research. *Hydrobiologia*, *787*(1), 27-44. https://doi.org/10.1007/s10750-016-2974-5

Sear, D. A. (1995). Morphological and sedimentological changes in a gravel-bed river following 12 years of flow regulation for hydropower. *Regulated Rivers: Research & Management*, *10*(2-4), 247-264. https://doi.org/https://doi.org/10.1002/rrr.3450100219

Sharma, A., Kumar, V., Shahzad, B., Tanveer, M., Sidhu, G. P. S., Handa, N., Kohli, S. K., Yadav, P., Bali, A. S., Parihar, R. D., Dar, O. I., Singh, K., Jasrotia, S., Bakshi, P., Ramakrishnan, M., Kumar, S., Bhardwaj, R., & Thukral, A. K. (2019). Worldwide pesticide usage and its impacts on ecosystem. *SN Applied Sciences*, *1*(11), 1446. Springer. https://doi.org/10.1007/s42452-019-1485-1

Shaw, E. A., & Richardson, J. S. (2001). Direct and indirect effects of sediment pulse duration on stream invertebrate assemblages and rainbow trout (Oncorhynchus mykiss) growth and survival. *Canadian Journal of Fisheries and Aquatic Sciences*, *58*(11), 2213-2221. https://doi.org/10.1139/f01-160

Shearer, K. A., & Young, R. G. (2011). Influences of geology and land use on macroinvertebrate communities across the Motueka River catchment, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, *45*(3), 437-454. https://doi.org/10.1080/00288330.2011.587823

Sheldon, F., Peterson, E. E., Boone, E. L., Sippel, S., Bunn, S. E., & Harch, B. D. (2012). Identifying the spatial scale of land use that most strongly influences overall river ecosystem health score. *Ecological Applications*, *22*(8), 2188-2203 pp. https://doi.org/10.1890/11-1792.1

Singers, N., & Rogers, G. (2014). A classification of New Zealand's terrestrial ecosystems. *Science for Conservation*, *325*, 87 pp.

Singh, S., Anil, A. G., Kumar, V., Kapoor, D., Subramanian, S., Singh, J., & Ramamurthy, P. C. (2022). Nitrates in the environment: A critical review of their distribution, sensing techniques, ecological effects and remediation. *Chemosphere*, *287*, 131996. https://doi.org/https://doi.org/10.1016/j.chemosphere.2021.131996

Sinner, J., Clark, D., Ellis, J., Jiang, W., Goodwin, E., Hale, L., Rolleston, S., Patterson, M., Hardy, D., Prouse, E., Brown, S. *Health of the Te Awanui Tauranga harbour*. Report No. 1. Mannaki Taha Moana Research Team. https://doi.org/10.13140/2.1.2879.5204

Smith, V. H., Joye, S. B., & Howarth, R. W. (2006). Eutrophication of freshwater and marine ecosystems. *Limnology and Oceanography*, *51*(1part2), 351-355. https://doi.org/https://doi.org/10.4319/lo.2006.51.1_part_2.0351

Snelder, T., H.,, Dey, K., & Suren, A. (2019). *Analysis of habitat factors influencing invertebrate communities in Streams of the Bay of Plenty Region*. LWP Client Report No: 2018-14. LWP.

Snelder, T. H., & Biggs, B. J. F. (2002). Multiscale River Environment Classifications for Water Sources Management. *Journal of the American Water Resources Association*, *38*(5), 1225-1239. https://doi.org/https://doi.org/10.1111/j.1752-1688.2002.tb04344.x

Snelder, T. H., Fraser, C., Larned, S. T., Monaghan, R., De Malmanche, S., & Whitehead, A. L. (2022). Attribution of river water-quality trends to agricultural land use and climate variability in New Zealand. *Marine and Freshwater Research*, *73*(1), 1-19. https://doi.org/10.1071/MF21086

Snelder, T. H., Whitehead, A. L., Fraser, C., Larned, S. T., & Schallenberg, M. (2020). Nitrogen loads to New Zealand aquatic receiving environments: comparison with regulatory criteria. *New Zealand Journal of Marine and Freshwater Research*, *54*(3), 527-550. https://doi.org/10.1080/00288330.2020.1758168

Southwood, T. R. E. (1977). Habitat, the Templet for Ecological Strategies? *Journal of Animal Ecology*, *46*(2), 337-365. https://doi.org/10.2307/3817

Stark, J., D.,. (1985). A Macroinvertebrate Community Index of Water Quality for Stony Streams. *Water and Soil Miscellaneous Publication* (87), 55. https://docs.niwa.co.nz/library/public/w&smp87.pdf

Stark, J., D.,, Boothroyd, I., K, G.,, Harding, J., S.,, Maxted, J., R.,, & Scarsbrook, M. R. (2001). *Protocols for sampling macroinvertebrates in wadeable streams*. New Zealand Macroinvertebrate Working Group Report No. 1. Prepared for the Ministry for the Environment. Sustainable Management Fund Project No. 5103. 57 p.

Stark, J., & Maxted, J. (2007). *A User Guide for the Macroinvertebrate Community Index*. Cawthron Report No. 1166. 57p. https://environment.govt.nz/assets/Publications/Files/mci-user-guide-may07.pdf

Stark, J. D. (1993). Performance of the Macroinvertebrate Community Index: Effects of sampling method, sample replication, water depth, current velocity, and substratum on index values. *New Zealand Journal of Marine and Freshwater Research*, *27*(4), 463-478. https://doi.org/10.1080/00288330.1993.9516588

StatsNZ. (2020). *Plenty of People moving to the Bay*. Stats NZ. https://www.stats.govt.nz/news/plenty-of-people-moving-to-the-bay/#:~:text=Bay%20of%20Plenty%20provisionally%20had,senior%20manager%20Brooke%20Theyers%20said.

StatsNZ. (2021). Agricultural and horticultural land use. Stats NZ https://www.stats.govt.nz/indicators/agricultural-and-horticultural-land-use

Statzner, B., & Bêche, L. A. (2010). Can biological invertebrate traits resolve effects of multiple stressors on running water ecosystems? *Freshwater Biology*, *55*(s1), 80-119. https://doi.org/https://doi.org/10.1111/j.1365-2427.2009.02369.x

Statzner, B., Dolédec, S., & Hugueny, B. (2004). Biological Trait Composition of European Stream Invertebrate Communities: Assessing the Effects of Various Trait Filter Types. *Ecography*, *27*(4), 470-488. http://www.jstor.org/stable/3683420

Statzner, B., Resh, V. H., & Roux, A. L. (1994). The synthesis of long-term ecological research in the context of concurrently developed ecological theory: design of a research strategy for the Upper Rhône River and its floodplain. *Freshwater Biology*, *31*(3), 253-263. https://doi.org/https://doi.org/10.1111/j.1365-2427.1994.tb01739.x

Sterk, M., Gort, G., Klimkowska, A., van Ruijven, J., van Teeffelen, A. J. A., & Wamelink, G. W. W. (2013). Assess ecosystem resilience: Linking response and effect traits to environmental variability. *Ecological Indicators*, *30*, 21-27. https://doi.org/https://doi.org/10.1016/j.ecolind.2013.02.001

Stevens, L. E., Schmidt, J. C., Ayers, T. J., & Brown, B. T. (1995). Flow Regulation, Geomorphology, and Colorado River Marsh Development in the Grand Canyon, Arizona. *Ecological Applications*, *5*(4), 1025-1039. https://doi.org/10.2307/2269352

Stone, W. W., Gilliom, R. J., & Ryberg, K. R. (2014). Pesticides in U.S. Streams and Rivers: Occurrence and Trends during 1992–2011. *Environmental Science & Technology*, *48*(19), 11025-11030. https://doi.org/10.1021/es5025367

Storey, R. G., & Quinn, J. M. (2011). Life histories and life history strategies of invertebrates inhabiting intermittent streams in Hawke's Bay, New Zealand. *New Zealand Journal of*

Marine and Freshwater Research, *45*(2), 213-230. https://doi.org/10.1080/00288330.2011.554988

Suedel, B. C., Boraczek, J. A., Peddicord, R. K., Clifford, P. A., & Dillon, T. M. (1994). Trophic Transfer and Biomagnification Potential of Contaminants in Aquatic Ecosystems. *Reviews of Environmental Contamination and Toxicology* (pp. 21-89). Springer. https://doi.org/10.1007/978-1-4612-2656-7_2

Sutherland, A., Culp, J., & Benoy, G. (2012). Evaluation of Deposited Sediment and Macroinvertebrate Metrics Used to Quantify Biological Response to Excessive Sedimentation in Agricultural Streams. *Environmental Management*, *50*, 50-63. https://doi.org/10.1007/s00267-012-9854-1

Syafrudin, M., Kristanti, R. A., Yuniarto, A., Hadibarata, T., Rhee, J., Al-Onazi, W. A., Algarni, T. S., Almarri, A. H., & Al-Mohaimeed, A. M. (2021). Pesticides in Drinking Water-A Review. *Int J Environ Res Public Health*, *18*(2). https://doi.org/10.3390/ijerph18020468

Szałkiewicz, E., Kałuża, T., & Grygoruk, M. (2022). Detailed analysis of habitat suitability curves for macroinvertebrates and functional feeding groups. *Scientific Reports*, *12*(1), 10757. https://doi.org/10.1038/s41598-022-15096-8

Taonui, R. (2015). *Whakapapa - genealogy*. The New Zealand Government. https://teara.govt.nz/en/whakapapa-genealogy

Thompson, R., & Townsend, C. R. (2000). New Zealand's stream invertebrate communities: an international perspective. In Collier, K., & Winterbourn, M., New Zealand's stream invertebrate communities: an international perspective (53-74). The University of Otago.

Thompson, R. M., & Townsend, C. R. (2003). Impacts on Stream Food Webs of Native and Exotic Forest: An Intercontinental Comparison. *Ecology*, *84*(1), 145-161. http://www.jstor.org.ezproxy.waikato.ac.nz/stable/3108005

Tilman, D., Knops, J., Wedin, D., Reich, P., Ritchie, M., & Siemann, E. (1997). The Influence of Functional Diversity and Composition on Ecosystem Processes. *Science*, *277*(5330), 1300-1302. https://doi.org/10.1126/science.277.5330.1300

Tipa, G., & Teirney, L. (2006). *Using the Cultural Health Index: How to assess the health of streams and waterways*. Ministry for the Environment. https://environment.govt.nz/assets/Publications/Files/chi-for-streams-and-waterways-feb06-full-colour.pdf

Townsend, C., Dolédec, S., & Scarsbrook, M. (1997). Species traits in relation to temporal and spatial heterogeneity in streams: A test of habitat templet theory. *Freshwater Biology*, *37*(2), 367-387. https://doi.org/10.1046/j.1365-2427.1997.00166.x

Townsend, C. R., Uhlmann, S. S., & Matthaei, C. D. (2008). Individual and Combined Responses of Stream Ecosystems to Multiple Stressors. *Journal of Applied Ecology*, *45*(6), 1810-1819. http://www.jstor.org/stable/20144160

Tubić, B., Popović, N., Raković, M., Petrovic, A. S., Simic, V. M., & Paunović, M. (2017). Comparison of the effectiveness of kick and sweep hand net and Surber net sampling techniques used for collecting aquatic macroinvertebrate samples. *Archives of Biological Sciences*, *2017*(2). https://doi.org/10.2298/ABS160622087T

Tudi, M., Daniel Ruan, H., Wang, L., Lyu, J., Sadler, R., Connell, D., Chu, C., & Phung, D. T. (2021). Agriculture Development, Pesticide Application and Its Impact on the Environment. *International Journal of Environmental Research and Public Health*, *18*(3). https://doi.org/10.3390/ijerph18031112

Ulrich, W., Batáry, P., Baudry, J., Beaumelle, L., Bucher, R., Čerevková, A., de la Riva, E. G., Felipe-Lucia, M. R., Gallé, R., Kesse-Guyot, E., Rembiałkowska, E., Rusch, A., Stanley, D., & Birkhofer, K. (2023). From biodiversity to health: Quantifying the impact of diverse ecosystems on human well-being. *People and Nature*, *5*(1), 69-83. https://doi.org/https://doi.org/10.1002/pan3.10421

Usseglio-Polatera, P., Bournaud, M., Richoux, P., & Tachet, H. (2000a). Biological and ecological traits of benthic freshwater macroinvertebrates: relationships and definition of groups with similar traits. *Freshwater Biology*, *43*(2), 175-205. https://doi.org/https://doi.org/10.1046/j.1365-2427.2000.00535.x

Usseglio-Polatera, P., Bournaud, M., Richoux, P., & Tachet, H. (2000b). Biomonitoring through biological traits of benthic macroinvertebrates: how to use species trait databases? *Hydrobiologia*, *422*, 153-162. https://doi.org/10.1023/A:1017042921298

Van Meeuwen-Dijkgaar, A., Shaw, W., B.,, & Mazzien, F. (2010). *Ecosystem services of protected areas and ecological corridors within the Kaimai Tauranga Catchments: Technical report.* Technical Report Series 2. Department of Conservation. https://www.doc.govt.nz/globalassets/documents/science-and-technical/ecosystem-services-kaimai-tauranga-catchments-2010.pdf

Van Rossen, R., & Adam, B. (2015). *Land-use Intensification Risks in the Bay of Plenty Region: An assessment of current use, capability and drivers for change*. Rob Van Rossen Consulting Ltd. https://www.boprc.govt.nz/media/415891/land-use-intensification-risks-in-the-bay-of-plenty-region_-an-assessment-of-current-use-capability-and-drivers-for-change.pdf

Vannote, R. L., Minshall, G. W., Cummins, K. W., Sedell, J. R., & Cushing, C. E. (1980). The River Continuum Concept. *Canadian Journal of Fisheries and Aquatic Sciences*, *37*(1), 130-137.

Villéger, S., Mason, N. W. H., & Mouillot, D. (2008). New multidimensional functional diversity indices for a multifaceted framework in functional ecology. *Ecology*, *89*(8), 2290-2301. https://doi.org/https://doi.org/10.1890/07-1206.1

Villeneuve, B., Piffady, J., Valette, L., Souchon, Y., & Usseglio-Polatera, P. (2018). Direct and indirect effects of multiple stressors on stream invertebrates across watershed, reach and site scales: A structural equation modelling better informing on hydromorphological impacts. The Science of the *Total Environment*, *612*, 660-671. https://doi.org/10.1016/j.scitotenv.2017.08.197

Violle, C., Navas, M.-L., Vile, D., Kazakou, E., Fortunel, C., Hummel, I., & Garnier, E. (2007). Let the Concept of Trait Be Functional. *Oikos*, *116*(5), 882-892. http://www.jstor.org/stable/40235131

Virens, A. (2023). From the Mountains to the Sea—Ki Uta ki Tai: Ecological Enclosure, Interconnection, and Subjectivity in the Commons. *Antipode*, *55*(4), 1275-1293. https://doi.org/https://doi.org/10.1111/anti.12931

Vries, J., Kraak, M., Verdonschot, R., & Verdonschot, P. (2019). Quantifying cumulative stress acting on macroinvertebrate assemblages in lowland streams. *Science of the total environment*, *694*, 133630. https://doi.org/https://doi.org/10.1016/j.scitotenv.2019.133630

Wagenhoff, A., Townsend, C. R., & Matthaei, C. D. (2012). Macroinvertebrate responses along broad stressor gradients of deposited fine sediment and dissolved nutrients: a stream mesocosm experiment. *Journal of Applied Ecology*, *49*(4), 892-902. https://doi.org/https://doi.org/10.1111/j.1365-2664.2012.02162.x

Wagenhoff, A., Townsend, C. R., Phillips, N., & Matthaei, C. D. (2011). Subsidy-stress and multiple-stressor effects along gradients of deposited fine sediment and dissolved nutrients in a regional set of streams and rivers. *Freshwater Biology*, *56*(9), 1916-1936. https://doi.org/https://doi.org/10.1111/j.1365-2427.2011.02619.x

Waite, I. (2013). Development and application of an agricultural intensity index to invertebrate and algal metrics from streams at two scales. Journal of the American Water Resources Association *49*(2), 431-448. https://onlinelibrary.wiley.com/doi/abs/10.1111/jawr.12032

Waite, I. R., Kennen, J. G., May, J. T., Brown, L. R., Cuffney, T. F., Jones, K. A., & Orlando, J. L. (2014). Stream macroinvertebrate response models for bioassessment metrics: addressing the issue of spatial scale. *PLoS One*, *9*(3). Pub Med. https://doi.org/10.1371/journal.pone.0090944

Wallace, J. B., & Webster, J. R. (1996). The Role of Macroinvertebrates in Stream Ecosystem Function. *Annual Review of Entomology*, *41*(1), 115-139. https://doi.org/10.1146/annurev.en.41.010196.000555

Wang, Y., Jia, Y., Li, Z., Tao, J., Lin, L., Chen, K., Liu, Z., Tan, X., & Zhang, Q. (2021). Trophic structure in response to land use in subtropical streams. *Ecological Indicators*, *127*, 107746. https://doi.org/https://doi.org/10.1016/j.ecolind.2021.107746

Weather and Climate. (2023). *Climate: Bay of Plenty*. https://en.climate-data.org/oceania/new-zealand/bay-of-plenty-1983/

Werner, I., & Moran, K. (2008). Effects of Pyrethroid Insecticides on Aquatic Organisms. *ACS Symposium Series*, *991*, 310-334. https://doi.org/10.1021/bk-2008-0991.ch014

Whitehead, A. L., Fraser, C., Snelder, T. H., Walter, K., Woodward, S., & C., Z. (2022). *Water quality state and trends in New Zealand rivers: analyses of national data ending in 2020.* NIWA Client Report No: 2021 (NIWA Client Report 2021296CH). NIWA.https://environment.govt.nz/assets/publications/river-water-quality-state-and-trends.pdf

Wilcock, R. J., & Nagels, J. W. (2001). Effects of aquatic macrophytes on physico-chemical conditions of three contrasting lowland streams: a consequence of diffuse pollution from agriculture? *Water Science and Technology*, *43*(5), 163-168. https://doi.org/10.2166/wst.2001.0277

Williams, G. P., & Wolman, M. G. (1984). *Downstream effects of dams on alluvial rivers*. Geological Survey Professional Paper *1286*. Department of the Interior. https://pubs.usgs.gov/publication/pp1286

Winterbourn, M., S.,, & Harding, J. S. (1997). *New Zealand eco-regions: a classification for use in stream conservation and management* (11). Department of Conservation Technical Series No. 11. Department of Conservation. https://docs.niwa.co.nz/library/public/DOCTS11.pdf

Winterbourn, M. J. (1974). The life histories, trophic relations and production of *Stenoperla prasina* (Plecoptera) and *Deleatidium sp*. (Ephemeroptera) in a New Zealand river. *Freshwater Biology*, *4*(6), 507-524. https://doi.org/https://doi.org/10.1111/j.1365-2427.1974.tb00111.x

Winterbourn, M. J. (1978). The macroinvertebrate fauna of a New Zealand forest stream. *New Zealand Journal of Zoology*, *5*(1), 157-169. https://doi.org/10.1080/03014223.1978.10423746

Winterbourn, M. J., Rounick, J. S., & Cowie, B. (1981). Are New Zealand stream ecosystems really different? *New Zealand Journal of Marine and Freshwater Research*, *15*(3), 321-328. https://doi.org/10.1080/00288330.1981.9515927

Wohl, E., Bledsoe, B. P., Jacobson, R. B., Poff, N. L., Rathburn, S. L., Walters, D. M., & Wilcox, A. C. (2015). The Natural Sediment Regime in Rivers: Broadening the Foundation for Ecosystem Management. *BioScience*, *65*(4), 358-371. https://doi.org/10.1093/biosci/biv002

Wood, P. J., & Armitage, P. D. (1997). Biological Effects of Fine Sediment in the Lotic Environment. *Environmental Management*, *21*(2), 203-217. https://doi.org/10.1007/s002679900019

Yadamsuren, O., Morse, J. C., Hayford, B., Gelhaus, J. K., & Adler, P. H. (2020). Macroinvertebrate community responses to land use: a trait-based approach for freshwater biomonitoring in Mongolia. *Hydrobiologia*, *847*(8), 1887-1902. https://doi.org/10.1007/s10750-020-04220-2

Young, R. G., Quarterman, A. J., Eyles, R. F., Smith, R. A., & Bowden, W. B. (2005). Water quality and thermal regime of the Motueka River: Influences of land cover, geology and position in the catchment. *New Zealand Journal of Marine and Freshwater Research*, *39*(4), 803-825. https://doi.org/10.1080/00288330.2005.9517354

Yuan, L. L. (2010). Estimating the effects of excess nutrients on stream invertebrates from observational data. *Ecological Applications*, *20*(1), 110-125. https://doi.org/https://doi.org/10.1890/08-1750.1

Zermeño-Hernández, I., Benítez-Malvido, J., Suazo-Ortuño, I., & Méndez-Toribio, M. (2020). Impact of adjacent land use on the ecological condition of riparian habitats: The relation between condition and vegetation properties. *Applied Vegetation Science*, *23*(4), 610-621. https://doi.org/https://doi.org/10.1111/avsc.12508

Zespri. (2021). *Kiwiflier* 429). https://www.zespri.com/content/dam/zespri/nz/publications/Kiwiflier/kiwiflier-2021/KF-430.pdf

7 Appendices

7.1 Appendix 1

Table 6.*Post-hoc results from (Log) total abundance and (Log) Simpsons Diversity of macroinvertebrate communities across three local land-use types (native forest, pastoral and horticultural) across 24 sites in the Te Awanui/Tauranga Harbour catchment.*

Response	**Contrast**	**Estimate**	**SE**	**df**	**t-ratio**	**P-value**
(Log) Total count	Native - Pastoral	0.35	0.58	20	0.6	0.82
(Log) Total count	Native - Horticultural	0.65	0.46	18	1.42	0.35
(Log) Total count	Pastoral - Horticultural	0.31	0.51	19	0.61	0.82
Taxa-richness	Native - Pastoral	-0.01	0.14	Inf	-0.06	1
Taxa-richness	Native - Horticultural	0.28	0.11	Inf	2.47	**0.04**
Taxa-richness	Pastoral - Horticultural	0.29	0.13	Inf	2.17	0.08
(Log) Simpsons index	Native - Pastoral	0.02	0.08	19	0.22	0.97
(Log) Simpsons index	Native - Horticultural	0.1	0.06	16	1.63	0.26
(Log) Simpsons index	Pastoral - Horticultural	0.08	0.07	18	1.19	0.47

Table 9. *Post-hoc ANOVA results from mixed-models testing changes in standardized macroinvertebrate indices across streams in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Contrast**	**Estimate**	**SE**	**df**	**t-ratio**	**P-value**
QMCI	Horticultural - Native	-1.531	0.31	13.7	-4.95	<**0.001**
QMCI	Horticultural - Pastoral	-0.66	0.37	15.5	-1.81	0.20
QMCI	Native - Pastoral	0.87	0.44	16.6	1.98	0.15
MCI	Horticultural - Native	-1.53	0.31	13.7	-4.95	<**0.001**
MCI	Horticultural - Pastoral	-0.66	0.37	15.5	-1.81	0.20
MCI	Native - Pastoral	0.87	0.44	16.6	1.98	0.15
ASPM	Horticultural - Native	-0.15	0.04	13.7	-3.78	**0.01**
ASPM	Horticultural - Pastoral	-0.09	0.05	15.5	-1.98	0.15
ASPM	Native - Pastoral	0.06	0.06	16.5	1.02	0.58
%EPT-abundance	Horticultural - Native	-0.21	0.10	14.8	-2.21	0.10
%EPT-abundance	Horticultural - Pastoral	-0.11	0.11	16.8	-0.98	0.60
%EPT-abundance	Native - Pastoral	0.10	0.13	18.1	0.79	0.71
EPT-richness	Horticultural - Native	-0.45	0.14	Inf	-3.17	**0.001**
EPT-richness	Horticultural - Pastoral	-0.19	0.18	Inf	-1.04	0.55
EPT-richness	Native - Pastoral	0.26	0.19	Inf	1.38	0.35
%Decreaser-abundance	Horticultural - Native	-0.10	0.11	16.0	-0.91	0.64
%Decreaser-abundance	Horticultural - Pastoral	-0.08	0.12	17.8	-0.65	0.80
%Decreaser-abundance	Native - Pastoral	0.02	0.14	19.0	0.14	0.99
Decreaser richness	Horticultural - Native	-0.11	0.04	15.3	-3.15	**0.02**
Decreaser richness	Horticultural - Pastoral	-0.07	0.04	17.2	-1.72	0.23
Decreaser richness	Native - Pastoral	0.04	0.05	18.5	0.89	0.65
%Increaser-abundance	Native - Pastoral	-0.06	0.12	19.1	-0.48	0.88
%Increaser-abundance	Native - Horticultural	-0.19	0.09	16.1	-2.15	0.11
%Increaser-abundance	Pastoral - Horticultural	-0.14	0.10	17.9	-1.36	0.38
%Increaser-richness	Native - Pastoral	-0.05	0.05	18.8	-0.93	0.63
%Increaser-richness	Native - Horticultural	-0.08	0.04	15.6	-2.04	0.14
%Increaser-richness	Pastoral - Horticultural	-0.03	0.05	17.5	-0.72	0.76

The relationship between the two Sediment Assessment Method protocols (SAM 2 & 3)

A linear regression and a logit-transformed linear regression model were used to investigate the relationship between the SAM 2 (Sediment Assessment Method) and the SAM 3 (Wolman walk) protocols (Figure 52). A logit transformation was performed in addition to account for the proportional nature of the data, as the aim of this analysis was to identify if the percentage of fine sediment detected using the SAM 2 method was correlated with the number of fine sediment (substrate belonging to the sand or mud size class) detected using the SAM 2 method.

The results of the linear regression between the SAM 2 and SAM 3 protocols were shown to be statistically significant ($R^2 = 0.91 \pm 16.5$, (F(2, 24) =[232.8], $P < 0.001$); (Table 49) and were highly correlated (91%) (Figure 49). Therefore, it would not be unreasonable to use both methods interchangeably.

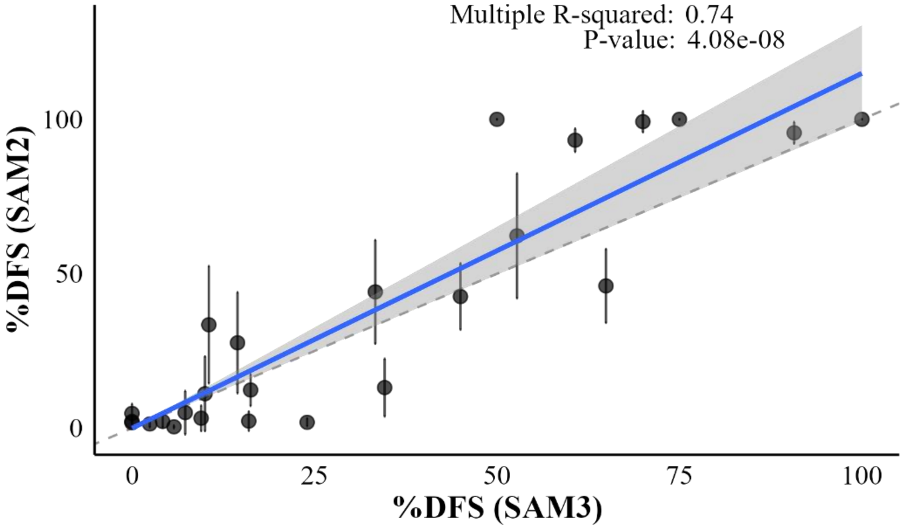

Figure 52. *The relationship between SAM2 and SAM3. The plot was logistically transformed.*

Table 49. *Results from the logistic regression of SAM 2 and SAM 3 relationship. Regression statistics (left), coefficients (middle), and residuals (right).*

Regression statistics	
Multiple R-squared	0.74
Adjusted R-squared	0.73
Residual standard error	0.97
d.f.	23
F-statistic	64.4
P- value	4.08E-08

Coefficients	**SAM 3**	**SAM 2**
Estimate	-0.59	0.63
Standard error	0.20	0.08
t value	2.93	8.02
Pr(>|t|)	0.01	4.08E-08

Residuals	**Values**
Min	-1.70
1Q	-0.67
Median	-0.08
3Q	0.58
Max	1.96

7.2 Appendix 2

Table 19. *Post-hoc results from mixed-models testing changes in metrics of macroinvertebrate functional diversity across streams in three local land uses (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	Contrast	Estimate	SE	df	t. ratio	P- Value
Functional divergence	Horticultural - Native	2.5	2	16.4	0.11	1
Functional divergence	Horticultural - Pastoral	0.29	1	19.6	0.6	0.84
Functional divergence	Native Pastoral	0.15	2	17.4	0.87	0.87

Table 22. *Post-hoc results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait Body flexibility' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	Contrast	Estimate	SE	df	t-ratio	P-value
Noflex	Native - Pastoral	-0.05	0.17	18	-0.29	0.96
Noflex	Native - Horticultural	-0.33	0.13	15	-2.58	0.05
Noflex	Pastoral - Horticultural	-0.28	0.14	17	-1.98	0.15
Lowflex	Native - Pastoral	-0.05	0.12	19	-0.42	0.91
Lowflex	Native - Horticultural	0.14	0.09	16	1.5	0.32
Lowflex	Pastoral - Horticultural	0.19	0.1	18	1.85	0.18
Highflex	Native - Pastoral	-0.02	0.1	15	-0.21	0.98
Highflex	Native - Horticultural	0.01	0.07	13	0.07	1
Highflex	Pastoral - Horticultural	0.03	0.08	14	0.31	0.95

Table 24. *Post-hoc results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'Body form' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	Contrast	Estimate	SE	df	t-ratio	P-value
Streamlined	Native - Pastoral	-0.11	0.10	16.63	-1.11	0.52
Streamlined	Native - Horticultural	-0.02	0.07	13.88	-0.23	0.97
Streamlined	Pastoral - Horticultural	0.09	0.08	15.46	1.12	0.52
Flattened	Native - Pastoral	0.14	0.16	18.10	0.88	0.66
Flattened	Native - Horticultural	0.32	0.12	15.14	2.65	0.05
Flattened	Pastoral - Horticultural	0.18	0.13	16.81	1.35	0.39
Spherical	Native - Pastoral	0.03	0.05	18.77	0.54	0.85
Spherical	Native - Horticultural	0.00	0.04	16.17	0.05	1.00
Spherical	Pastoral - Horticultural	-0.03	0.04	17.65	-0.58	0.83

Table 27. *Post-hoc results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'Maximum number of descendants per reproductive cycle' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	Contrast	Estimate	SE	df	t-ratio	P-value
Desc1	Native - Pastoral	-0.18	0.12	16	-1.49	0.32

Desc1	Native - Horticultural	-0.32	0.09	14	-3.60	**0.01**
Desc1	Pastoral - Horticultural	-0.14	0.10	15	-1.38	0.38
Desc2	Native - Pastoral	-0.04	0.12	18	-0.34	0.94
Desc2	Native - Horticultural	0.08	0.09	15	0.85	0.68
Desc2	Pastoral - Horticultural	0.12	0.10	17	1.16	0.49
Desc3	Native - Pastoral	0.15	0.09	17	1.71	0.23
Desc3	Native - Horticultural	0.13	0.06	14	2.02	0.15
Desc3	Pastoral - Horticultural	-0.02	0.07	16	-0.26	0.96
Desc4	Native - Pastoral	0.17	0.09	19	2.01	0.14
Desc4	Native - Horticultural	0.24	0.07	17	3.50	**0.01**
Desc4	Pastoral - Horticultural	0.07	0.07	18	0.90	0.65

Table 29. *Post-hoc results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'Maximum number of reproductive cycles per year' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Contrast**	**Estimate**	**SE**	**df**	**t-ratio**	**P-value**
Semi	Native - Pastoral	0.15	0.07	18	2.28	0.08
Semi	Native - Horticultural	0.14	0.05	15	2.72	**0.04**
Semi	Pastoral - Horticultural	-0.013	0.06	17	-0.23	0.97
Univ	Native - Pastoral	-0.07	0.09	18	-0.84	0.68
Univ	Native - Horticultural	0.04	0.07	15	0.53	0.86
Univ	Pastoral - Horticultural	0.11	0.08	17	1.46	0.33
Pluriv	Native - Pastoral	0.03	0.16	17	0.19	0.98
Pluriv	Native - Horticultural	-0.15	0.12	15	-1.23	0.46
Pluriv	Pastoral - Horticultural	-0.18	0.14	16	-1.31	0.41

Table 31. *Post-hoc results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'Maximum number of reproductive cycles per individual' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Contrast**	**Estimate**	**SE**	**df**	**t-ratio**	**P-value**
Cpi1	Native - Pastoral	-0.05	0.05	19	-0.93	0.63
Cpi1	Native - Horticultural	0.08	0.04	16	2.19	0.10
Cpi1	Pastoral - Horticultural	0.13	0.04	18	3.07	**0.02**
Cpi2	Native - Pastoral	0.04	0.14	19	0.30	0.95
Cpi2	Native - Horticultural	-0.21	0.11	16	-1.91	0.17
Cpi2	Pastoral - Horticultural	-0.25	0.12	18	-2.09	0.12

Table 33. *Post-hoc results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'Life duration of adults' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Contrast**	**Estimate**	**SE**	**df**	**t-ratio**	**P-value**
Lda2	Native - Pastoral	0.12	0.10	16	1.20	0.47

Lda2	Native - Horticultural	0.22	0.07	14	2.98	**0.03**
Lda2	Pastoral - Horticultural	0.10	0.08	15	1.17	0.49
Lda3	Native - Pastoral	-0.03	0.12	18	-0.22	0.98
Lda3	Native - Horticultural	0.00	0.09	15	0.00	1.00
Lda3	Pastoral - Horticultural	0.03	0.10	16	0.26	0.96
Lda4	Native - Pastoral	-0.16	0.14	17	-1.17	0.49
Lda4	Native - Horticultural	-0.27	0.10	14	-2.61	0.051
Lda4	Pastoral - Horticultural	-0.11	0.12	16	-0.92	0.63
Lda5	Native - Pastoral	-0.23	0.12	16	-1.84	0.19
Lda5	Native - Horticultural	-0.21	0.09	14	-2.28	0.09
Lda5	Pastoral - Horticultural	0.02	0.10	15	0.21	0.98

Table 35. *Posthoc results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'Reproductive techniques' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Contrast**	**Estimate**	**SE**	**df**	**t-ratio**	**P-value**
Single	Native - Pastoral	0.04	0.09	19	0.48	0.88
Single	Native - Horticultural	-0.16	0.07	17	-2.25	0.09
Single	Pastoral - Horticultural	-0.2	0.08	18	-2.61	**0.05**
Herma	Native - Pastoral	0.04	0.05	19	0.81	0.7
Herma	Native - Horticultural	0.01	0.04	16	0.17	0.99
Herma	Pastoral - Horticultural	-0.03	0.04	18	-0.78	0.72
Two	Native - Pastoral	-0.02	0.02	19	-1.09	0.53
Two	Native - Horticultural	0.03	0.02	16	1.76	0.21
Two	Pastoral - Horticultural	0.05	0.02	17	2.87	**0.03**

Table 37. *Posthoc results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'Oviposition sites' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Contrast**	**Estimate**	**SE**	**df**	**t-ratio**	**P-value**
Surface	Native - Pastoral	0.13	0.14	16	0.95	0.62
Surface	Native - Horticultural	0.24	0.10	14	2.42	0.07
Surface	Pastoral - Horticultural	0.11	0.11	15	0.99	0.59
Submerged	Native - Pastoral	-0.19	0.14	17	-1.40	0.36
Submerged	Native - Horticultural	-0.31	0.10	14	-3.07	**0.02**
Submerged	Pastoral - Horticultural	-0.12	0.11	16	-1.05	0.56
Terrestrial	Native - Pastoral	-0.18	0.11	17	-1.71	0.23
Terrestrial	Native - Horticultural	-0.14	0.08	14	-1.78	0.21
Terrestrial	Pastoral - Horticultural	0.04	0.09	16	0.47	0.89
Eggendo	Native - Pastoral	-0.03	0.02	18	-1.54	0.30
Eggendo	Native - Horticultural	-0.01	0.01	15	-0.40	0.92
Eggendo	Pastoral - Horticultural	0.02	0.01	17	1.44	0.34

Table 39. *Post-hoc results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'Egg/egg mass' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Contrast**	**Estimate**	**SE**	**df**	**t-ratio**	**P-value**
Eggfree	Native - Pastoral	0.10	0.10	17	1.01	0.58
Eggfree	Native - Horticultural	0.19	0.07	14	2.56	0.06
Eggfree	Pastoral - Horticultural	0.09	0.08	16	1.06	0.55
Eggcement	Native - Pastoral	-0.29	0.13	17	-2.16	0.11
Eggcement	Native - Horticultural	-0.21	0.10	14	-2.15	0.11
Eggcement	Pastoral - Horticultural	0.07	0.11	16	0.66	0.79
Eggprotected	Native - Pastoral	0.04	0.12	19	0.33	0.94
Eggprotected	Native - Horticultural	-0.21	0.09	16	-2.25	0.09
Eggprotected	Pastoral - Horticultural	-0.25	0.10	18	-2.44	0.06

Table 41. *Post-hoc results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'Aquatic stages' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Contrast**	**Estimate**	**SE**	**df**	**t-ratio**	**P-value**
Aduandlar	Horticultural-Native	0.36	0.09	13.4	4.23	**0.003**
Aduandlar	Horticultural-pastoral	0.18	0.10	14.8	1.85	0.19
Aduandlar	Native-pastoral	-0.82	0.12	15.8	-1.53	0.30
Aduorlar	Horticultural-Native	-0.45	0.10	13.7	-4.54	**0.001**
Aduorlar	Horticultural-pastoral	-0.17	0.11	15.2	-1.53	0.30
Aduorlar	Native-pastoral	0.28	0.13	16.3	2.04	0.13

Table 44. *Post-hoc results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'Attachment to substrate of aquatic stages (excluding eggs)' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	**Contrast**	**Estimate**	**SE**	**df**	**t-ratio**	**P-value**
Swimmer	Native - Pastoral	0.01	0.04	17	0.38	0.92
Swimmer	Native - Horticultural	0.07	0.03	14	2.42	0.07
Swimmer	Pastoral - Horticultural	0.05	0.03	15	1.67	0.25
Crawler	Native - Pastoral	-0.03	0.05	16	-0.65	0.80
Crawler	Native - Horticultural	-0.02	0.03	14	-0.70	0.77
Crawler	Pastoral - Horticultural	0.006	0.04	15	0.17	0.99
Burrower	Native - Pastoral	0.05	0.09	19	0.53	0.86
Burrower	Native - Horticultural	0.12	0.07	16	1.76	0.22
Burrower	Pastoral - Horticultural	0.07	0.07	17	0.98	0.60
Attached	Native - Pastoral	0.17	0.12	17	1.48	0.32
Attached	Native - Horticultural	0.15	0.09	14	1.76	0.22
Attached	Pastoral - Horticultural	-0.02	0.10	16	-0.20	0.98

Table 46. *Post-hoc results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'Feeding habits' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	Contrast	Estimate	SE	df	t-ratio	P-value
Shredder	Native - Pastoral	0.08	0.10	17	0.79	0.71
Shredder	Native - Horticultural	0.06	0.08	14	0.75	0.74
Shredder	Pastoral - Horticultural	-0.02	0.09	16	-0.28	0.96
Scraper	Native - Pastoral	-0.04	0.09	19	-0.47	0.89
Scraper	Native - Horticultural	-0.10	0.07	16	-1.47	0.33
Scraper	Pastoral - Horticultural	-0.06	0.08	17	-0.78	0.72
Deposit	Native - Pastoral	0.03	0.06	19	0.40	0.92
Deposit	Native - Horticultural	0.04	0.05	16	0.82	0.69
Deposit	Pastoral - Horticultural	0.02	0.06	18	0.29	0.95
Filterfeeder	Native - Pastoral	0.05	0.13	19	0.40	0.92
Filterfeeder	Native - Horticultural	0.10	0.10	16	0.93	0.63
Filterfeeder	Pastoral - Horticultural	0.04	0.11	17	0.37	0.93
Predator	Native - Pastoral	-0.12	0.08	18	-1.56	0.29
Predator	Native - Horticultural	-0.01	0.06	15	-0.11	0.99
Predator	Pastoral - Horticultural	0.12	0.07	17	1.73	0.22
Algalp	Native - Pastoral	-0.02	0.04	18	-0.52	0.86
Algalp	Native - Horticultural	-0.02	0.03	15	-0.58	0.84
Algalp	Pastoral - Horticultural	0.004	0.04	16	0.10	1.00

Table 48. *Post-hoc results from mixed-models testing changes in Community Weighted Mean trait abundances of the modalities of the trait 'Dietary preferences' in three local land-use types (native forest, pastoral, horticultural) in the Te Awanui/Tauranga Harbour basin. Logit-transformed % deposited fine sediment (DFS) was included in the models as a covariate.*

Response	Contrast	Estimate	SE	df	t-ratio	P-value
Specialist	Native - Pastoral	-0.02	0.08	17	-0.26	0.96
Specialist	Native - Horticultural	0.04	0.06	14	0.74	0.74
Specialist	Pastoral - Horticultural	0.07	0.07	16	0.97	0.60
Moderatespe	Native - Pastoral	0.06	0.11	16	0.53	0.86
Moderatespe	Native - Horticultural	0.16	0.08	13	2.10	0.13
Moderatespe	Pastoral - Horticultural	0.11	0.09	15	1.20	0.47
Generalist	Native - Pastoral	-0.03	0.06	17	-0.52	0.86
Generalist	Native - Horticultural	-0.07	0.04	14	-1.58	0.29
Generalist	Pastoral - Horticultural	-0.04	0.05	16	-0.77	0.73