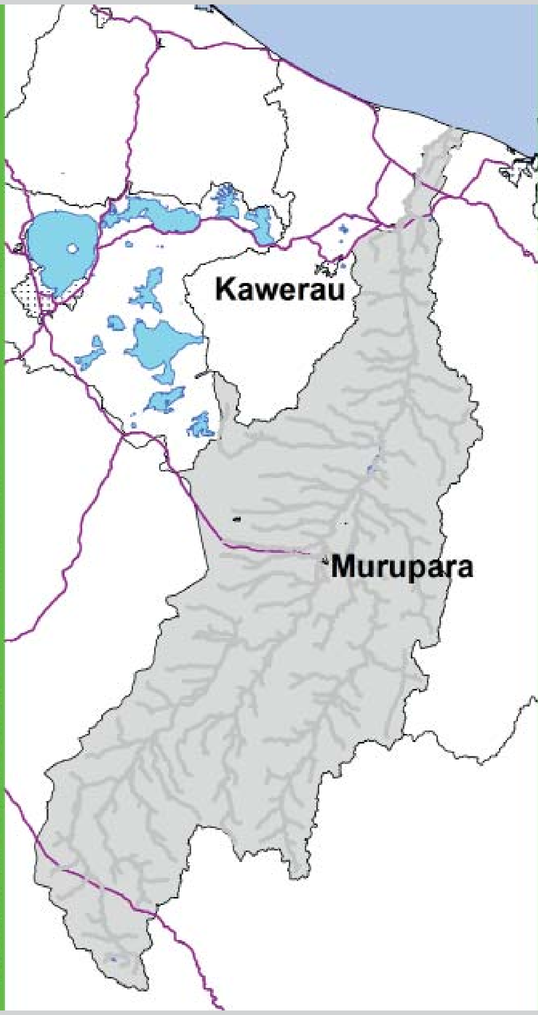
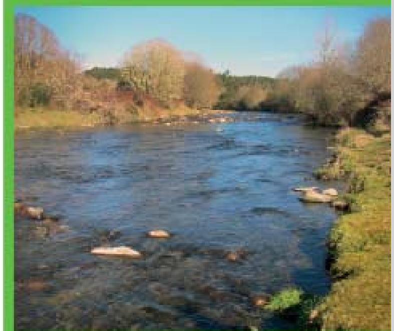
An ecological assessment of waterways throughout the Rangitāiki Catchment





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Cover Photo: Images of the Rangitāiki River taken during the large-scale field survey.

Non-technical summary

1 As part of work for the Rangitāiki River Forum, an ecological survey of the Rangitaiki Catchment was conducted to, amongst other things, characterise the water quality and ecological condition of waterways throughout the catchment and determine factors responsible for controlling this, detect any temporal changes in ecological health and provide science-based information to feed into vision documents such as Te Ara O Rangitāiki – Pathways of the Rangitāiki and any other future planning and policy work.

2 A total of 117 sites throughout the Rangitāiki Catchment were examined to provide a "snapshot" of their current ecological health. While most sites had not been surveyed before, others had been, some up to 30 years ago. This gave us the ability to see whether stream health had changed over time. At each site, assessments were made of habitat conditions, invertebrate communities and water quality. Invertebrates are used to assess stream health as they integrate a wide range of environmental factors over relatively long periods of time, unlike water quality, which can vary greatly over shorter time frames. Water samples were also collected from Lakes Matahina and Aniwaniwa, and aquatic plants surveyed at these lakes.

3 A wide range of streams were surveyed, ranging from small steep streams draining native bush, to the mainstem of the Rangitāiki below the Matahina Dam. This showed that a wide range of environmental gradients exist within the catchment. These gradients represented large-scale factors such as catchment area, distance inland and dominant land cover and small-scale factors such as water quality, catchment slope, streambed and bank condition.

4 Because of the large variability of environmental factors, simple classifications based on individual factors such as land cover, stream size or location alone would not adequately describe the environmental variability between streams. To better define management groups, streams were classified into eight groups based on all measured factors. Streams in each group differed in regard to factors such as stream size, location, slope, land cover and water quality.

5 Most of the streams supported invertebrate communities typical of streams in "good" or "excellent" health. Invertebrate composition differed greatly between streams, reflecting differences in river size, location, dominant land cover and water quality (particularly the plant nutrient nitrogen). Highest stream health was in streams draining native bush and exotic forest. Stream health was lowest in streams draining pasture and was especially low in the mainstem of the Rangitāiki River on the plains.

6 Invertebrate communities had not changed much over a 30-year period, even in streams draining pasture catchments. Lack of changes to stream health in highly modified pasture streams suggests that stream health changed before the earliest surveys (30 years ago) and that communities currently found have shifted to a new stable state.

7 Water chemistry differed between land use, with higher bacterial and nutrient loads in streams draining pasture than native bush. Water quality of streams draining pine plantations was intermediate. Average concentration of nitrogen in pasture streams was above levels where algal blooms can develop which may reduce overall stream health. However, algal blooms were not commonly observed in many soft-bottomed streams, suggesting that they are more resistant to effects of nutrients than gravel bed streams. Further observations of the degree to which algal blooms can develop in waterways throughout the catchment are required.

8 Water quality monitoring showed that Lake Aniwaniwa was enriched with nutrients, and Lake Matahina was highly enriched. Aquatic plants (mostly introduced) form extensive growths in Lake Aniwaniwa. Although weed growth may have negative effects on lake ecology and recreational values, results of water quality monitoring suggest that these plants may be removing nutrients from the lake and improving water quality in the lower river.

9 The results of this survey have relevance to a number of desired outcomes of the Rangitāiki River Forum. Two outcomes are to enhance Mauri and He Awa. If invertebrate communities act as a surrogate measure of Mauri, then we can assume that many of the sites sampled would have a high degree of Mauri. Other sites, particularly in the lower Rangitāiki had invertebrate communities indicative of only fair, or poor health, and would be expected to have low Mauri. This would reflect the extensive modifications to the river bank and the presence of a number of point source discharges.

10 Significant challenges exist for the Bay of Plenty Regional Council (BOPRC) to help the Rangitāiki River Forum to achieve some of their proposed objectives, such as "Restoring Water Quality". Although the forum and the community have expectations for the river water to be (amongst other things) swimmable, drinkable and abundant; the real challenge is to set clearly defined measurable objectives to help achieve their desired outcomes, and in identifying appropriate spatial scales for these objectives. For example, our survey showed that pasture streams have high concentrations of bacteria and nitrogen. However, our ecological monitoring suggests that this increase in nitrogen does not yet appear to have significant negative ecological impacts. The challenge for BOPRC, the community and relevant stakeholder groups is, then, to decide what terms such as "restoring the water quality" mean? Is it aimed at reducing the observed increases in nutrient levels in waterways in the catchment, even when the nitrogen concentrations may not be having ecological effects? Or, is it aimed at reducing the degree of bacterial contamination in the water, which has adverse effects on cultural and recreational values? The Forum has clearly expressed their vision and desired outcomes for the Rangitāiki River Catchment, so BOPRC needs to develop sufficient plans, methods and rules in their regional plans to help meet these outcomes.

11 In addition, the issue of setting appropriate spatial scales for management needs to be considered. For example, cyanobacterial blooms were observed in fast flowing riffle areas at discrete locations in the Rangitāiki River below Murupara, but most of these areas were presumably well away from areas of easy public access. These blooms may have been in response to elevated nitrogen enrichment in the river. However, a large portion of the river was naturally unsuitable for cyanobacteria as it was either too deep, or dominated by mobile pumice sands. Is it appropriate, therefore, for management activities to be focused on minimising nitrogen inputs to the river when the ecological effects of such inputs are highly spatially variable and may be causing cyanobacterial blooms in areas with only low public accessibility? Furthermore, should monitoring be done in remote sites if they are hard to get to, and if they represent only a relatively small spatial area within the catchment?

Recommendations for further work

12 Seven recommendations for further work are made:

(i) Continue with water quality monitoring;

(ii) Monitor algal cover and/or biomass to characterise the extent that algal blooms can form at sites throughout the catchment, and use this data to help develop nutrient limits;

(iii) Quantify the current extent of streams lacking riparian protection, and set objectives for implementation of riparian protection programmes;

(iv) Continue with monthly water quality sampling from Lakes Aniwaniwa and Matahina to monitor the Trophic Level Index (TLI);

(v) Work with communities to develop an action plan for Lake Aniwaniwa;

(vi) Set objectives to maintain an "optimum" macrophyte biomass in Lake Aniwaniwa;

(vii) Undertake cultural health investigations.

13 Implementation of these recommendations will further improve our state of knowledge about the ecological responses to pressures in the catchment, and allow clear and specific management goals to be developed. For example, data quantifying algal blooms will assist both the forum and the community to agree on desired states as to how much algal growth is acceptable. This information would also help with implementation of Central Government's recently released National Policy Statement for Freshwater Management (NPS-FW, 2014) in terms of setting nutrient limits. Finally, undertaking cultural stream health assessments throughout the catchment will be of great benefit to not only iwi, but other communities in the catchment so that the true recognition of iwi values can be recognised and better protected.

Technical summary

1 The Rangitāiki River, the largest in the Bay of Plenty, is faced with considerable pressures from land-use changes, reductions in water quality, alterations to the flow regime from hydro-electricity generation, and habitat change caused by flood protection works such as historic drainage of the old Rangitāiki swamps, creation of stopbanks, and use of rock reinforcing (riprap) to strengthen eroding banks. Because of these pressures, it is likely that the river's ability to support the cultural values at a level that iwi (Ngāti Awa, Ngāti Manawa, Ngāti Whare, and Tūwharetoa) would like, has declined.

2 The Rangitāiki River Forum (RRF) was formed as a statutory joint committee from the four local iwi and councillors from the Bay of Plenty Regional Council (BOPRC) and Whakatane District Council. Its aim is to protect and enhance the environmental, cultural, and spiritual health and wellbeing of the Rangitāiki River. It has prepared a draft vision document (Te Ara O Rangitāiki - Pathways of the Rangitāiki) that has identified four desired outcomes for the river, including:

- The desire to protect the Mauri of the water.
- He Taiao: to have a bountiful river… especially for eels and whitebait.
- He Tangata: to have a balanced, connected and spiritual relationship with the rivers and resources of the Rangitāiki.
- He Awa: to have a clean and healthy environment characterised by clean water, healthy ecosystems and the return of some threatened species.

Eight specific objectives have also been identified to help achieve these outcomes, including the protection and enhancement of habitats that support indigenous species, and the restoration of water quality in the catchment.

3 As part of the RRF's work, BOPRC commissioned an ecological survey of the Rangitāiki Catchment. This survey had five objectives:

(i) Characterise the ecological condition of waterways throughout the catchment;

(ii) Determine factors controlling ecological condition of waterways;

(iii) Detect temporal trends in ecological health over a 20–30 year period;

(iv) Characterise water quality of smaller tributaries draining different land uses, and of lakes Matahina and Aniwaniwa;

(v) To provide science-based information to feed into the Te Ara O Rangitāiki - Pathways of the Rangitāiki Document to help ensure sustainable development within the catchment.

4 A total of 117 representative sites in the mainstem Rangitāiki River, its large tributaries, and smaller headwater streams were examined to provide a "snapshot" of their current ecological state. At each site, assessments were made of habitat conditions and invertebrate communities. Invertebrates are used to assess stream health, as they are relatively long lived and respond to environmental stressors in a predictable way. They integrate a wide range of environmental conditions at a site over relatively long periods of time, unlike water quality sampling which can vary greatly over shorter time frames.

5 Most sites (97) had not been surveyed before, while 20 sites were selected as they had been surveyed previously either as part of council State of Environment Monitoring Programmes, or as part of historic consent investigations. This allowed us to assess any temporal changes to stream health. Water quality samples were also collected from each site on the day of sampling, and a subset of sites was randomly selected for monthly water quality monitoring over a six month period. Monthly water samples were also collected from lakes Matahina and Aniwaniwa and aquatic macrophyte communities of those lakes were also surveyed.

6 The results of the report are presented in five sections:

1 Spatial considerations;
2 Habitat data;
3 Invertebrate communities, including assessments of both state and trends;
4 Water quality;
5 Lake macrophytes.

It then makes recommendations for future monitoring, and discusses relevance of the results to the Te Ara O Rangitāiki - Pathways of the Rangitāiki Draft Discussion Document, and to the recently released National Policy Statement for Freshwater Management.

Spatial considerations

7 A spatial analysis of the location of historic samples showed that small waterways had been overlooked, as were waterways draining catchments dominated by native bush. This information was used to help select sites for the contemporary survey, which included small streams flowing through native bush. The contemporary survey thus filled a large information gap regarding the previously unknown ecology of these streams.

Environmental conditions

8 Twenty-six environmental factors were measured or derived at all sites. These factors were measured at three spatial scales. Firstly, the smallest scale was within the area of the stream being studied (the stream reach), where individual factors such as substrate size and stream width were measured. Secondly, medium-scale factors such as flow, modelled nitrogen load and stream order were assessed within a stream segment - the length of a stream between tributaries. Thirdly, large-scale factors such as climate and land cover that described overall conditions within the stream's catchment were described. This environmental data was used to firstly characterise environmental conditions throughout the catchment, and secondly to determine whether natural stream groups existed based on their overall environmental conditions.

9 Environmental conditions varied widely, with large differences in catchment area, distance to the sea and land cover. Small scale factors describing bank and riparian conditions also varied greatly, as did water chemistry (conductivity and predicted nitrogen loads). This variability highlights the very large environmental gradients within the Rangitāiki Catchment. Thus, simple classifications based only on single factors such as land cover, stream size or location within a catchment would not adequately describe the environmental variability between streams. Instead, any classification would need to be based on multiple environmental factors.

10 Classification of all environmental factors revealed the existence of eight groups that differed in regards to catchment, segment and reach scale factors. Sites within a particular class were located throughout the Rangitāiki Catchment. This meant that the classification was not spatially explicit, but based instead on multiple factors reflecting differences in land cover, stream size, and reach scale habitat factors. Specific management or restoration objectives could thus be set for each group, which recognises that not all waterways are the same.

Invertebrate communities

11 Most of the streams surveyed supported a diverse invertebrate community, dominated by animals indicative of streams in good ecological condition such as mayflies and caddisflies. The Macroinvertebrate Community Index (MCI) and its quantitative variant (QMCI) were calculated for each site to describe their overall ecological condition. Calculated scores were high, ranking many streams in either "excellent" or “good” condition. Community composition differed widely between sites, with some sites dominated by mayflies, stoneflies, caddisflies and toebiters, and other sites dominated by snails, blackflies and daphnia. Changes in community composition reflected differences in environmental factors such as river size, location, dominant land-cover, and predicted nitrogen yield. Other factors influencing invertebrate communities (and by inference stream health) included degree of bank under cutting and percentage riffles in a stream. Overall, streams draining native bush and pine plantations had the highest ecological health, while streams draining pasture had lower stream health – especially around the Rangitāiki Plains.

12 Only very minor changes were observed between invertebrate communities collected in contemporary or historic surveys - some of which were 30 years old. Lack of strong temporal changes suggests that a high degree of stability exists among invertebrate communities, even over 30 years. Reduced ecological health in pasture streams and the lower Rangitāiki River reflects the effects of past activities such as land use changes, construction of the hydro dams, and habitat modification that occurred prior to 1975; the date of the first surveys. These activities would have impacted on the ecology of the waterways at the time and reduced their ecological health to a new state. This new state, however, has not changed dramatically since.

13 Observed MCI scores were compared against those predicted in the absence of human activities (i.e., before any land use change had occurred). Large differences between observed and predicted scores imply a reduction in stream health. The greatest differences were in samples from the lower Rangitāiki below the Matahina Dam. This reduction in stream health reflected the combined effects of human disturbances associated with land use change, changes in water quality, and alterations in flow regime below the Matahina Dam. Pasture streams in the upper Rangitāiki above Murupara also displayed relatively large differences between contemporary and historic MCI scores and so were regarded as being the next most impacted. However, other pasture streams were in relatively good ecological condition, reflecting the fact that they often flowed from areas of native bush.

Water chemistry

14 Water chemistry differed between land use, with high *E. coli* counts and nutrient concentrations in streams draining pasture, and low concentrations in streams draining native bush. Water quality of streams draining pine plantations was intermediate between the two. Calculated nitrogen (N) yields were higher in catchments dominated by pasture and pine, and a significant positive relationship existed between the percentage of pasture in a catchment and calculated N catchment yield. Average concentrations of inorganic N in pasture streams were higher than 0.8 g/m^3 – a suggested upper level above which algal blooms can develop, with subsequent adverse effects on stream health. However, algal blooms were not observed in the soft-bottomed streams, suggesting that they are more resistant to the negative effects of nutrient enrichment than gravel-bed streams.

15 Lake Aniwaniwa had higher N concentrations than Lake Matahina, lower levels of phytoplankton (assessed as chlorophyll a), and higher water clarity. These water quality parameters were used to calculate the Trophic Level Index (TLI), to assess each lake's overall condition. Calculated TLI values suggested that Lake Aniwaniwa is an enriched lake, and Lake Matahina is a highly enriched lake with respect to nutrients. In comparison to the Te Arawa/Rotorua lakes, Lake Matahina was more similar to Lake Ōkaro, with high nutrient and phytoplankton levels, and low clarity. In contrast, Lake Aniwaniwa was more like lakes Rotoiti, Rotomahana and Rotoehu, and characterised by lower nutrient and phytoplankton levels and greater clarity. However, growth was much more prolific in Lake Aniwaniwa.

Aquatic plants

16 The ecological condition of each lake was assessed by examination of their aquatic plant communities and by calculating the Lake Submerged Plant Indicator (LakeSPI) score. A total of 12 species were recorded, of which six were native. Three introduced plants (hornwort, Canadian pondweed and curly oxygen weed) were dominant in both lakes. This extensive cover resulted in both lakes having a "poor" rating based on their LakeSPI score. Although weed growth in Lake Aniwaniwa has negative impacts on aesthetic, recreational and ecological values, results of water quality monitoring suggested that it was removing nutrients and as such may be improving water quality in the lower river.

Relevance to Rangitāiki Draft Discussion Document

17 The results of this survey have relevance to a number of desired outcomes of the RRF, especially to ensure that the Mauri of the water is protected, and that there is a celebrated, clean and healthy environment (He Awa). Strong links exist between western science measures of stream health such as the MCI and Maori assessments of cultural health, so the generally high MCI scores we observed at many of the sites suggest that a high degree of Mauri also exists throughout the catchment. However, sites in the Rangitāiki below the Matahina Dam had lower MCI scores, reflecting amongst other things extensive modifications to the river bank, presence of a number of point source discharges and being affected by the hydro-dams. These sites would be expected to have low Mauri.

18 Although the River Forum has clearly defined their overarching objectives, significant challenges still exist for achieving these objectives. A major challenge is to decide how terms such as "restoring the water quality" are properly defined and measured. Communities want water to be swimmable, drinkable, abundant and suitable for ceremonies and to sustain mahinga kai. The challenge we are now faced with is to identify the pressures that cause waterways in the catchment not to meet these values, identify where these pressures are strongest, and which values they affect. For example, our survey showed that pasture streams have high concentrations of bacteria and N. However, our ecological monitoring suggests that this increase in N does not appear to have significant negative ecological impacts to date at the sites we monitored. Are elevated N concentrations therefore contrary to the forum's expectations of restoring water quality if they are not having ecological effects? Should management be focussed primarily on reducing the degree of bacterial contamination in the water, or on increasing water clarity – both of which effect cultural and recreational values?

Furthermore, our survey work revealed the existence of discrete patches of cyanobacterial blooms growing in generally inaccessible areas of the Rangitaiki River. Is it appropriate, therefore, for management activities to be focused on minimising nitrogen inputs to the river when the ecological effects of such inputs are highly spatially variable and may be causing cyanobacterial blooms in areas with only low public accessibility? Such questions need to be addressed by BOPRC in consultation with the forum and the rest of the Rangitāiki community and relevant stakeholders, so that appropriate changes to plans, methods and rules in the Regional Plan can be made if necessary.

19 It is hoped that this report will help the forum (and BOPRC) address these issues by first identifying potential pressures, quantifying the ecological response to these and help set ecological bottom lines. It is the maintenance of these bottom lines that will help the forum achieve their overall objectives and vision for the Rangitaiki Catchment.

Recommendations

20 Seven recommendations have been made based on interpretation of the results of this study:

(i) Continue with water quality monitoring;

(ii) Monitor algal cover and/or biomass to characterise this, and help develop nutrient limits if nutrients are causing algal blooms;

(iii) Quantify the current extent of streams lacking riparian protection, and set objectives for implementation of riparian protection programmes;

(iv) Continue with monthly TLI sampling from Lakes Aniwaniwa and Matahina;

(v) Work with communities to develop an action plan for Lake Aniwaniwa;

(vi) Set objectives to maintain an "optimum" macrophyte biomass in Lake Aniwaniwa;

(vii) Undertake cultural health investigations.

21 Implementation of these recommendations will further improve our state of knowledge about the ecological responses to pressures in the catchment and allow clear and specific management goals to be developed. For example, improved information concerning algal blooms will assist both the RRF and the community to agree on desired states of how much algae they want in streams throughout the catchment. This information would also help with implementation of Central Government's recently released National Policy Statement for Freshwater Management (NPS-FW: 2014) in terms of setting nutrient limits. Finally, undertaking cultural stream health assessments throughout the catchment will be of great benefit to not only iwi, but other communities in the catchment so that the true recognition of iwi values can be recognised and better protected.

Contents

Part 1: Introduction

1.1 The Rangitāiki Catchment: an overview

The Rangitāiki River is the longest river in the Bay of Plenty region, and at 2,947 km^2 is also the largest catchment. It originates in small headwater streams arising on the northern flanks of the Kaimanawa Ranges, which coalesce and flow north for about 155 km to the coast. A number of large tributaries such as the Wheao and Whirinaki Rivers join the Rangitāiki in the upper half of its catchment.

Like many rivers throughout New Zealand, the Rangitāiki has undergone considerable modification to its catchment since European settlement some 150 years ago. For example, the river flows for much of its length through the 1186 km^2 Kāingaroa Forest, the largest exotic plantation forest in the southern hemisphere. This forest was first planted in *Pinus radiata* the late 1920s in what used to be scrub and tussock land and many areas are now into their third harvest rotation. Forestry covers approximately 52% of the catchment, making it the dominant land use (Figure 1). Other land use in the catchment include native bush, (28%) and pasture (18%) which comprises a mix of dairy farming and beef. Areas of native bush occur along the eastern side of the catchment in the Urewera and Whirinaki State Forests, while intensive dairy farming occurs mainly in the Galatea and Rangitāiki Plains. Based on our understanding of the effects of land-use change on stream ecology (Collier and Winterbourn, 2000; Townsend et al., 1997; Winterbourn, 1986; Young and Huryn, 1999), it is highly likely that changes in land use such as converting tussock and scrub to pine plantations, or conversion of scrub and native bush to pasture would have had large historic impacts on the ecology of the many waterways throughout the Rangitāiki. Such changes would reflect alterations to the flow regime, riparian conditions, and water quality (e.g. nutrients and sedimentation). However, most of these dramatic land use changes have occurred over 50 years or more ago, and recent analysis of land use within the catchment has shown that it has been relatively stable for the past 20 years (Boubee et al 2009).

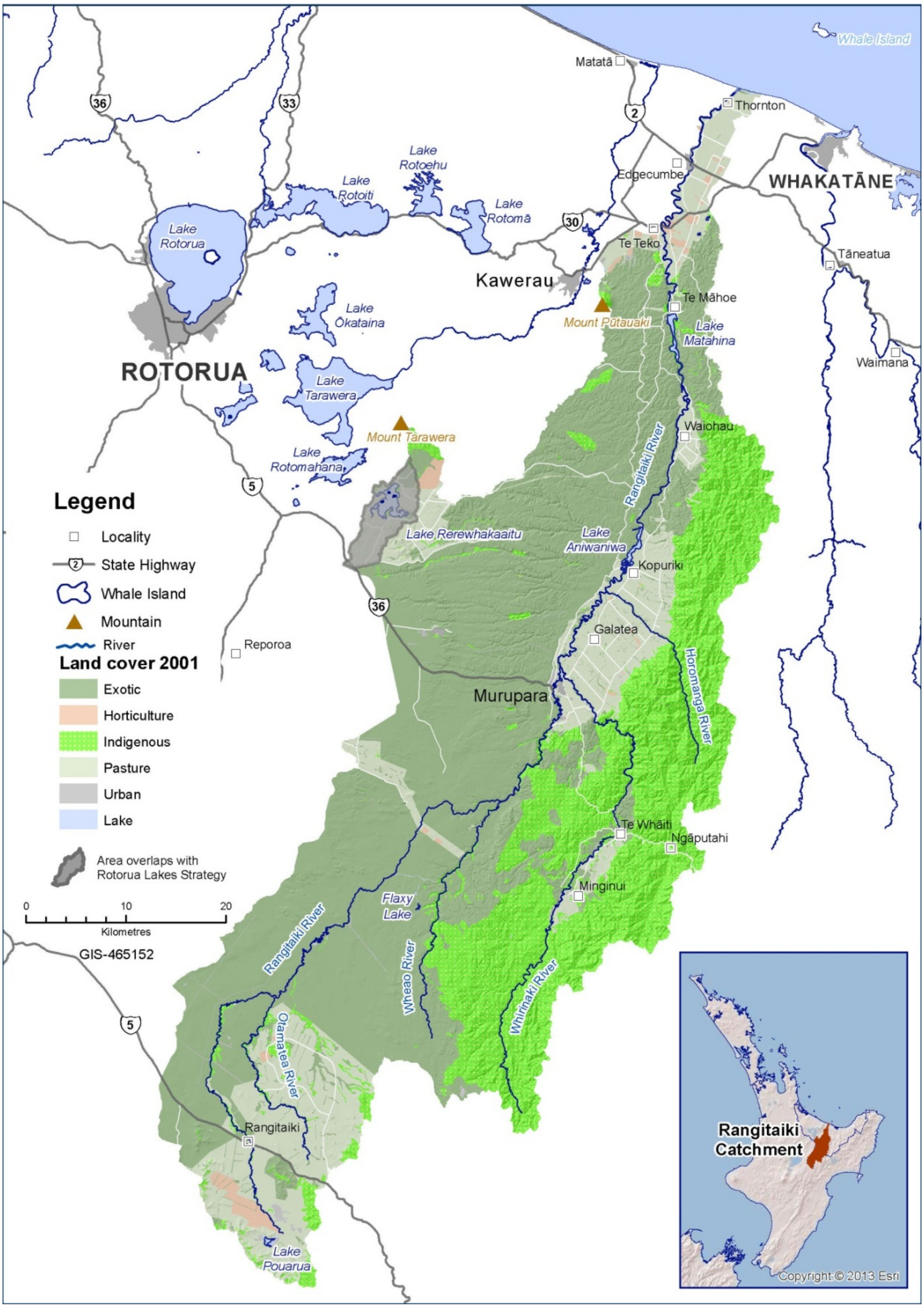


Figure 1 Map of the Rangitāiki Catchment, showing major place names, and the dominant land uses.

Land-use change is not the only stressor potentially affecting the ecological condition of the Rangitāiki River. Three hydroelectric power schemes occur in the river, producing a combined power output of approximately 530 GWh. In the upper catchment, the Wheao Power Scheme (111 GWh) diverts water from the Rangitāiki River, the Wheao River and Flaxy Creek through a series of constructed canals leading to the power house, where it discharges back into the Wheao. This scheme was constructed in 1982. Midway down the catchment is the second hydro scheme, where the Rangitāiki River was dammed above the Aniwhenua Falls. The resultant Lake Aniwaniwa is a major recreational resource supporting a significant trout fishery, duck shooting opportunities and is a popular waterskiing area. The Aniwhenua Scheme produces 130 GWh, and was completed in 1979. The Matahina Dam is the lowermost hydro-scheme on the river, approximately 20 km from the coast, behind which is Lake Matahina. This scheme, commissioned in 1967, produces 290 GWh of energy, and at 86 m, is the highest earth dam in the North Island.

The ecological effects of hydro schemes on aquatic ecosystems are well-known (Henriques, 1987; Young et al., 2004). Dams alter natural longitudinal processes in rivers, including downstream sediment and nutrient transport. They also affect downstream flow regimes, often with large effects on downstream ecosystems (Lessard et al., 2012). Dams also interrupt the ability of many of New Zealand's native fish to freely migrate between the ocean and the headwaters. Within the Rangitāiki River, the natural migration behaviour of longfin and shortfin eels, koaro, giant and banded kokopu have all been affected by the Aniwhenua and Matahina Dams. Although systems of manual trap and transfer have been implemented to minimise these effects, considerable challenges still exist to ensure that natural migration pathways are maintained. The hydro schemes have also greatly altered flow regimes in the rivers. For example, the Wheao Hydro Electric Scheme diverts most of the flow from the Wheao River into Flaxy Lake, before discharging this back into the lower reaches of the Wheao, downstream of the dam (Boubee et al., 2009). This has led to the loss of about a 7 km stretch of river with natural flows. Similarly, a significant proportion of flow from the Rangitāiki River is diverted to the Wheao River via the Rangitāiki Canal, leaving a minimum flow of only 0.5 m^3/s downstream from the diversion. The Rangitāiki River then naturally rejoins the Wheao River, some 18 km away. Thus this stretch of river has a highly modified flow regime, characterised by increased periods of "flat-lining" in the affected parts of the river, and reduced numbers of smaller flushing flows that are important for maintaining ecological functions such as "cleansing" the river of excess algal blooms. Flows below Lake Matahina are also regulated, with a new operating regime (consented in 2014) characterised by a twin peaking regime in place, a minimum flow of 35 m^3/s at Te Teko during periods of "normal" inflows into the lake, and a new minimum flow of 30 m^3/s when inflows fall below this. This latter flow regime was the subject of Environment Court mediation between the dam's operator (TrustPower) and the regulatory authority in the region (Bay of Plenty Regional Council (BOPRC)).

Other pressures facing the Rangitāiki River include an increased demand for further abstraction of groundwater. There is only a very small amount of surface water available for abstraction (Boubee et al., 2009), and so increasing intensification of farming in the Galatea Plains and upper catchment around SH35 may place more pressure on groundwater resources for further abstraction. Water quality issues are also as important as nutrient concentrations, particularly phosphorus and nitrogen, appear to be increasing in the Rangitāiki River at Murupara since 2000 (Boubee et al., 2009; Scholes et al., 2011). These increases most likely reflect effects of activities such as conversion of land to more intensive use such as dairy farming in the upper catchment around the Napier/Taupo Highway.

Such intensification has significant effects on water quality (Larned et al., 2004), and the highly permeable pumice soils typical of many areas of the catchment would only exacerbate this effect. Other water quality issues potentially arise from point source discharges associated with oxidation pond discharges from the small town of Murupara midway in the catchment, or from point source discharges from the Fonterra Dairy Factory at Edgecumbe, less than 10 km from the coast.

Finally, habitat conditions within the lower river have been significantly altered over the past 150 years. The Rangitāiki River used to flow unconstrained over the Rangitāiki Plains, forming a large flax dominated wetland between the Rangitāiki and Tarawera rivers. Excavation work during the 1920s and 1930s saw much of these wetlands drained, and an extensive network of canals and farm drains were created to lower the water table, and turn this wetland into now productive farmland, dominated by dairy farming. Much of the original wetland vegetation that grew alongside the waterways has gone, and today many of the small remaining streams and drains in the plains lack any form of overhanging riparian vegetation. Instead, the dominant vegetation around these straightened waterways is short, grazed pasture. A large proportion of the Rangitāiki River, particularly below the Matahina Dam, is also constrained by stop banks, which are often protected with riprap to minimise flood damage. These works are maintained by BOPRC as part of their statutory obligations for flood control throughout the region. Although rip-rap lined banks may provide a degree of shelter for some fish species in the spaces between the boulders (Johnston, 2011), they are unlikely to provide the same ecological role as riparian vegetation in terms of providing cover and shade. For example, rip-rap banks at the salt-water/freshwater interface are highly unsuitable areas for whitebait spawning (Mitchell, 1991). Despite their possible adverse ecological effects, activities in the lower part of the river have allowed the development of farms and establishment of small towns throughout the Rangitāiki Plains, which have allowed the region to prosper.

The ecological condition of the Rangitāiki Catchment (upstream of Aniwhenua Dam) was investigated in an extensive report by Boubee et al (2009) that described parameters such as surface and groundwater hydrology, soils, land use, rainfall, water quality and ecology. Flows in the catchment were generally stable and groundwater dominated, and significantly affected by Hydro schemes. Flows from tributaries such as the Whirinaki River were described as being more variable and rainfall dominated than the flow in the Rangitāiki, reflecting the different geologies. The report also showed that land use in the catchment appeared stable between 1990 and 2009, but that there was some evidence of conversion from forestry to pasture in areas with low to moderate erosion potential. In particular, the report suggested that increasing dairying, especially on the Galatea Plains, had the potential to increase pressure on instream habitat conditions. It also highlighted potential adverse impacts from land use intensification on groundwater quality and quantity.

The Boubee et al report also highlighted that total N concentrations were higher in the Rangitāiki than the Whirinaki and that these concentrations were increasing. There was an especially sharp increase in nutrients in the Rangitāiki River at Murupara from about 2000–2004, and this was suggested to reflect land use consents granted to farms in the upper catchment to discharge untreated dairy effluent to pasture. The report also highlighted the generally low bacterial contamination of waterways throughout the catchment, and showed how this bacterial load increased during rainfall events. It concluded by saying that although surface waters throughout the catchment were regarded as being unsafe for human consumption, they represented only a low risk to recreational users.

Finally, the Boubee report examined the catchment's ecological values, including aquatic plants, invertebrates and fish. However, it concentrated mainly on aquatic communities at selected sites in the mainstem of the Rangitāiki River and (with the exception of fish communities), did not specifically examine the ecology of the many smaller tributary streams. For example, comments on the invertebrate community were restricted mainly to the long term monitoring sites run by either NIWA or BOPRC as part of their State of Environment monitoring in larger rivers such as the Whirinaki, and the Rangitāiki at Murupara and Te Teko. The report assessed algal communities at selected sites and found that algal biomass occasionally exceeded MfE recommended guideline values for trout and angling (Biggs 2000). It suggested that these algal growths were possibly in response to increasing nutrient concentrations in the Rangitāiki River. It also showed that Lake Aniwaniwa was characterised by dense growths of exotic macrophytes, and in particular hornwort (*Ceratophyllum demersum*). The report also showed that phytoplankton blooms often occurred in Lakes Aniwaniwa and Matahina, reducing their clarity.

1.2 Catchment management legislation

Land use change and catchment management within the Rangitāiki River needs to be seen in a historical context, especially post European settlement where arguably most of the changes occurred. For example, following the passing of legislation in 1910, 40,000 ha of land in the Rangitāiki Plains were drained and converted into farmland. During the 1920s and 1930s the New Zealand Forest Service was formed to plant out the Kāingaroa Plains, (which was dominated by tussock and scrub) with *Pinus radiata* as a work-creation project. Catchment management in the 1930's was also driven primarily around the need to control flooding and erosion. In 1941 the Soil Conservation and Rivers Control Act was passed, and the Soil Conservation and Rivers Control Council was established. The aims of the council included promotion of soil conservation and prevention of erosion and flood damage. Large flooding in 1964 caused considerable damage to farmland throughout the Rangitāiki Plains, and this was an impetus for improved flood defences there.

By 1967, catchment boards were found throughout the country, whose main focus was to minimise and prevent damage by floods and erosion. After a review of New Zealand's soil conservation in 1964, the Water and Soil Conservation Act was passed in 1967, and a Water and Soil Division set up within the Ministry of Works and Development. Under its control were the Soil Conservation and Rivers Control Council, the Water Pollution Control Council and the Water Allocation Council. Importantly, at local level the control lay with each catchment authority, via the regional catchment boards. Emphasis was placed on measuring the extent and types of erosion through land inventory surveys and land capability assessments.

Further change to catchment management occurred with the passing of the Resource Management Act (RMA) in 1991, and the creation of regional councils (essentially the former catchment boards) and territorial authorities (district and city councils). The RMA (1991) is based on the sustainable management of resources (including soils) and encourages long-term planning. Under the RMA, councils' main responsibilities are to manage environmental, resource and transport planning, including the sustainable use of land, air and water, and to protect communities against the effects of flooding.

Catchment management of the Rangitāiki River has therefore been the responsibility of the BOPRC (and its predecessors) for over 60 years, as part of its statutory responsibilities under the RMA, and prior to this under legislation focussed on minimising soil erosion and flooding.

Today, BOPRC is responsible for deciding all consent applications for discharges to, and abstractions from any waterway throughout the catchment, and for regulating other activities such as the damming of rivers and draining of wetlands. They are also responsible for the maintenance of the drainage network in the lower Rangitāiki Plains (much of which was constructed in the early – mid 1900's), as well as the construction and maintenance of the many kilometres of stopbanks that characterise the lower reaches and which provide vital flood protection to the many farms, towns and infrastructure on the Rangitāiki Plains.

The Rangitāiki River has also played an important role in the lives of generations of Māori, with iwi such as Ngati Whare, Ngati Manawa, Ngati Awa and Ngāti Tūwharetoa traditionally having close connections to the river. Indeed, amongst these iwi, the Rangitāiki is considered taonga: a significant cultural treasure to be shared and protected by all. Iwi have a concept called Kaitiaki - a special obligation to ensure the health and wellbeing of the Rangitāiki River and its resources are managed for the benefit of present and future generations. Given the large degree that land cover within the catchment has been altered, combined with changes in the flow regime of the river, changes in water chemistry, reduced instream habitat from land drainage and alterations to riparian conditions, and the effects of the hydro dams on fish migration, it is clear that the river today is very different to what it was 100 years ago. It is therefore arguable whether the river is able to provide iwi with the same services and values that it used to.

In an effort to restore the Mauri (life-giving capacity) of the Rangitāiki River and its tributaries, the Rangitāiki River Forum (RRF) was established under the Ngāti Whare Claims Settlement Act 2012 and the Ngati Manawa Claims Settlement Act 2012. The RRF was formed in May 2012 as a statutory joint committee with the aim of protecting and enhancing the environmental, cultural, and spiritual health and wellbeing of the Rangitāiki River. The RRF is a partnership made up of representatives from the four iwi, and councillors from BOPRC and Whakatāne District Council (WDC). These partners now work together to set the direction of how the river is looked after for future generations.

It is clear that the Rangitāiki River represents a catchment with only a finite amount of resources that are available for continued resource use. Consequently, there is a definite requirement to balance the needs of the community in terms of allowing sustainable development while safeguarding ecological, recreational, and cultural values of the river. In this way, the future use of the river by the next generation will be ensured. To help achieve this balance between development and the need to preserve values, the RRF has prepared Te Ara O Rangitāiki – Pathways of the Rangitāiki, a Draft Rangitāiki River Consultation Document (Bay of Plenty Regional Council, 2014) that sets out its vision, goals and strategies. The River Document has identified four desired outcomes for the Rangitāiki River:

- Mauri - Mauri of the water is protected;
- He Tangata - We will have a balanced, connected and respectful relationship with the rivers and resources of the Rangitāiki;
- He Taiao - We want to see a bountiful river– where native habitats and customary harvesting practices sustain people and where whitebait and tuna (eels) abound;
- He Awa - We want to see a celebrated, clean and healthy environment – characterised by clean water, healthy ecosystems and the return of some threatened species. People use and enjoy this environment for their spiritual, cultural and recreational needs and celebrate its heritage with pride.

1.3 Aims of this study

As part of the RRF's work, BOPRC commissioned an ecological survey of the Rangitāiki Catchment to provide information for the Draft Rangitāiki River Consultation Document. The aims of the study were to:

1 Characterise the ecological condition of waterways in the Rangitāiki Catchment;

2 Determine factors affecting the ecological condition of waterways;

3 Detect any temporal trends in changes in ecological health over time;

4 Characterise water quality conditions of the smaller tributaries streams draining different land uses in the Rangitāiki catchment, and of Lakes Matahina and Aniwaniwa;

5 Provide science-based information that can feed into the Draft Rangitāiki River Document for consultation and any future documents prepared by the either the RRF or BOPRC to ensure sustainable development within the catchment.

Prior to the survey work commencing, a literature review was conducted on all ecological work that had been undertaken previously in the catchment (Suren, 2013). This review provided a stock-take of our current knowledge of the catchment's ecology. One of the findings of the review was that considerable discrepancies existed between the types of waterways historically sampled, and the types of waterways found throughout the catchment. For example, most historic samples (39%) were collected the mainstem of the Rangitāiki River, while another 35% of samples were from large rivers such as the Whirinaki or Wheao (orders 5 and 6). Only 23% of samples (i.e., 7) had been collected from small-medium sized rivers (order 3 and 4), and only one sample (the Mangapapa) had been collected from a small stream (order 1 or 2). A large discrepancy thus existed between where sampling had occurred and the numerical abundance of smaller waterways in the catchment. The majority of samples (65%) had also been collected from streams flowing through plantation forestry, over-representing this land use type. In contrast, only 6% of streams were collected from streams draining native forest, despite the fact that 25% of the catchment drains native forest. On the basis of these results, it was deemed necessary to undertake a more thorough ecological survey of the many smaller streams draining under-represented land uses such as native bush. The literature review also identified sites to be re-sampled as part of a contemporary survey to allow historical ecological data to be compared with the contemporary data to determine whether any changes in ecological condition had occurred.

This report describes the results of the ecological study in five sections:

1 Spatial considerations for the study;

2 Description of habitat data;

3 Description of invertebrate communities, including assessments of both current state and temporal trends;

4 Assessment of water quality;

5 Description of lake macrophytes.

Each section contains the specific methods, results and discussion that summarises key findings. A final section of the report makes a number of recommendations for further work, and considers the relevance of the findings to the RRF, and the Draft Rangitāiki River Consultation Document, as well as the NPS-FW.

Part 2: Spatial considerations

2.1 Introduction

There are approximately 4400 km of waterways in the Rangitāiki Catchment, ranging from very small streams that could easily be stepped across right through to the large mainstem of the Rangitāiki. Waterways in the catchment also flow through a variety of contrasting land uses, ranging from relatively undisturbed native bush, pine plantations (at various stages of their growing and harvesting cycle) and pasture. Previous ecological assessments have concentrated mostly on large rivers, with small waterways being generally under represented. It is important to sample smaller rivers as river size can have profound effects on ecological communities. For example, small streams are in more intimate contact with the surrounding landscape, and are affected more by riparian conditions than larger rivers (Ministry for the Environment, 2001; Quinn et al., 2001). Hydraulic and hydrological processes are also likely to play a bigger role in structuring ecological communities in larger rivers, as factors such as the shear stress are much greater in fast flowing deep rivers than shallow ones (Davis, 1986; Dittrich and Schmedtje, 1995). Any plants or animals adhering to rocks are thus more likely controlled by hydraulic forces in larger rivers. As such, effects of riparian vegetation and catchment land use changes may not be as apparent in larger rivers as in the small streams.

Furthermore, most historic samples have been collected from streams flowing through plantation forestry, with fewer samples coming from streams draining native bush or pasture. Given the importance of land use in influencing ecological processes (Quinn et al., 1994; Quinn et al., 1993), it is clear that more samples needed to be collected from streams draining these other land uses to better characterise them. Furthermore, under the recently released National Policy Statement for Freshwater Management (NPS-FM, 2014), regional councils have a legislative requirement to maintain or improve overall water quality within a region. This means that adequate monitoring of the full range of waterways found within each region is essential to ascertain if councils are meeting their statutory obligations.

A key part of this ecological survey was to classify waterways into groups that shared specific characteristics, as biological communities and ecological processes are likely to differ between stream types – even within a single catchment such as the Rangitāiki. Before any between-stream comparisons could be made, streams needed to be assigned to similar classes. Classification also allows comparison of rivers draining different land uses to be made: e.g., pasture with native bush or plantation forest, but a series of specific “rules” are needed to assign a stream to a specific land use class. A spatial framework is therefore needed that allocates streams throughout the Rangitāiki Catchment to specific and clearly defined classes. Utilisation of such a framework will ensure that any sites selected for the ecological survey would be representative of waterways throughout the catchment.

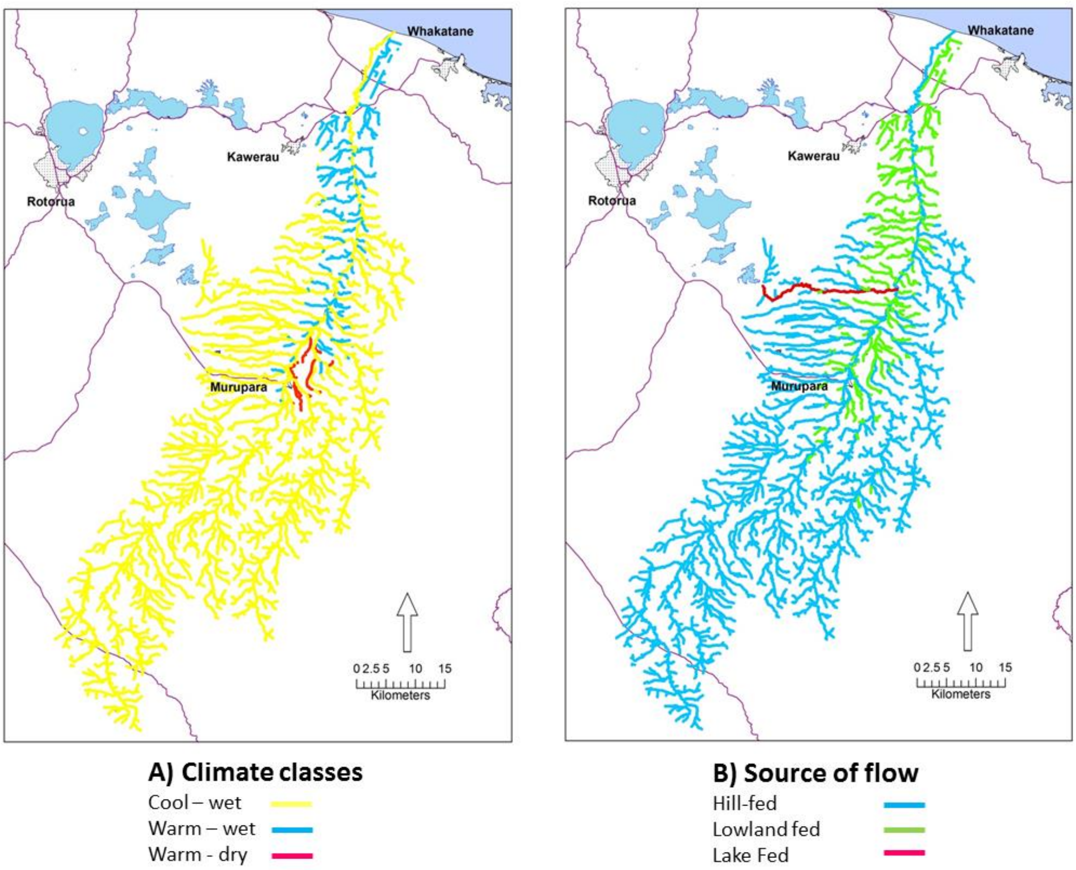
2.2 Methods

The River Environment Classification (REC) was used to classify all waterways in the Rangitāiki Catchment. The REC is based on a digital elevation model (DEM) with a 20 m contour that shows the location of waterways running through valleys defined by this DEM. A network of waterways is thus built up along valleys, with individual segments (called an NZReach) of waterway being defined as segments between in-flowing tributaries.

This network of waterways is essentially very similar to the blue lines that show streams on a standard 1:50,000 topographic map. The DEM is also used to delineate a catchment boundary around each identified waterway. This catchment boundary is used to interrogate other databases to obtain information such as climatic conditions within the catchment, or information about its underlying geology and land cover. The DEM also represents a flow path along a waterway, enabling calculations of catchment conditions to be made at locations both upstream and downstream from a particular point by combining all catchments either above or below this point.

The REC classifies rivers according to parameters that influence ecological communities such as climate, source of flow, geology and land cover (Snelder et al., 1998; Snelder and Biggs, 2002). Every waterway within the Rangitāiki Catchment thus has its own NZReach identification, of which the REC database identified 6244 reaches. Each river segment was classified according to climate, source of flow, geology, landcover and stream size (small, medium, large). A total of 30 different REC stream types were identified, of which 16 represented 94% of the total accessible waterway length throughout the catchment. These stream types included the following classes (Figure 2):

- Three climate classes (cool wet, warm wet, warm dry);
- Three source of flow classes (Hill fed, Lake fed, and lowland fed);
- Two geology classes (volcanic and hard sedimentary);
- Three land-use classes (pasture, pine plantation and native bush); and
- Three stream sizes (small, medium, large).



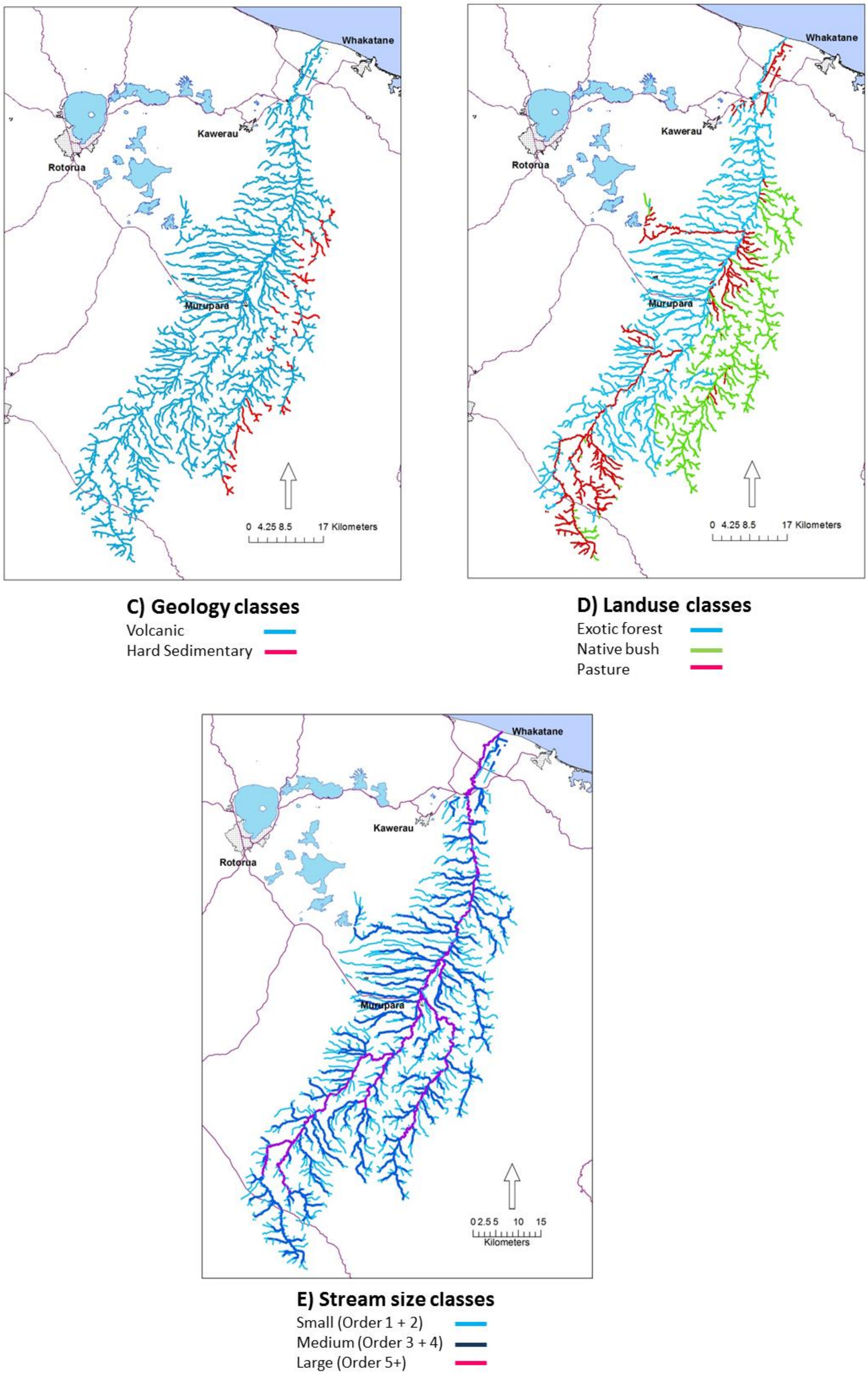


Figure 2 *Map of the Rangitāiki Catchment showing the location of waterways in different REC A) climate, B) source of flow, C) geology, D) land use classes and E) stream size.*

Many of these reaches occurred in inaccessible areas in the middle of native bush or pine plantations, and were thus not possible to sample. GIS was used to select a subset of reaches within 200 m of a road. This spatial filter was chosen as it seemed an acceptable distance to walk to sample a site. This spatial filter reduced the number of waterway reaches down to 948. Individual reaches in each of the 16 dominant stream types were subsequently randomly selected as sites to sample. If during the field work a particular pre-identified site was dry or inaccessible (e.g. surrounded by impenetrable gorse or blackberry, or down a steep gorge), the next randomly selected site belonging to the same class was sampled. Using this method, samples were collected from the 15 most common REC classes, with at least three samples being collected in each stream class.

Although 82 sites were selected using the above process, twenty sites were chosen where surveys had previously been conducted. Most of these sites (ten) had been surveyed in the upper Rangitāiki, Wheo and Flaxy Creek areas in the 80s and 90s as part of investigations for the Wheo Hydroelectric Scheme. Five more sites were part of BOPRC's annual state of environment monitoring, and one site was part of NIWA's National Water Quality Monitoring Network sites. An additional 15 sites were sampled in the mainstem of the Rangitāiki River between Murupara and the Thornton Bridge to better characterise the river in this area, and to document any changes to the ecology arising from both land use changes through both the Galatea and Rangitāiki Plains, and from any effects of the hydro dams. Samples from these areas were selected using a stratified random approach where sites that were suitable for sampling (shallow, and with cobble or gravel streambeds) were randomly selected. A total of 117 sites were thus selected for sampling. The % of sites sampled in the different REC classes was then compared to the % of sites in the same REC classes throughout the catchment.

2.3 Results

Results of the REC analysis showed that a total of 4410 km of waterways exist within the Rangitāiki Catchment. Of this combined distance, most (73%) were small first and second-order streams, while the larger fifth order or greater streams comprised only 6.1% of total waterway length (Table 1). The dominance of small streams throughout the catchment was not, however, reflected by the survey results. Here, of the 117 streams sampled (equating to 165 km), the combined stream length of small first and second order streams was only 28% of total stream length sampled, much less than their combined length throughout the catchment based on the REC data. This disparity reflected the fact that many of the first and second order streams that were identified in the REC river network and selected for sampling were in fact dry. It must be remembered that the REC database does not contain any information about whether a stream is permanently flowing or not, and that as with any 1:50,000 topographic map, the presence of a blue line on a map does not necessarily indicate permanently flowing water. Dry streams were particularly notable throughout the pumice dominated western parts of the catchment, and through the Kāingaroa forest. Most of the small streams in the Greywacke dominated eastern parts of the catchment appeared permanent. This meant that the actual length of flowing waterways in the Rangitāiki is somewhat less than the 4,410 km as originally calculated, based on the REC layer.

Our survey also overrepresented the number of medium and large size rivers sampled (Table 1), reflecting the fact that many of the smaller waterways were ephemeral and so could not be sampled. It also reflected the 15 additional samples collected from the main stem of the Rangitāiki River between Murupara and the Thornton Bridge, which were collected to better describe the ecological condition of the mainstem river.

The REC analysis of all waterways throughout the catchment showed that the vast majority (86%) of them were in the cool-wet climate class, while a further 12% were in the warm-wet climate class (Table 1). Only 1.5% of streams were in the warm-dry climate class. The dominant source of flow consisted of hill fed streams (78%) and lowland fed streams (21%). The REC identified only 1% of total waterway length sampled as being lake-fed: the main stem of the Rangitāiki River below Lake Aniwaniwa and Lake Matahina. Examination of the REC classes of the surveyed streams showed that they had similar patterns in the percentage of stream length for both climate class and source of flow, with the exception that no streams in the warm-dry climate class had been sampled.

The dominant geology consisted of volcanic (93%) and hard sedimentary (6%) material in the catchment, with only small areas (0.4%) of alluvium (Table 1). The surveyed streams showed a similar breakdown of dominant geology. Just over half (51%) of the waterway length drained catchments of exotic plantation forest, while natural vegetation (native forest, scrub, and tussock) was the dominant vegetation draining 28% of waterway length in the catchment. The other dominant land-use was pasture, which drained 21% of waterway length. Similar differences in land cover were found in all streams surveyed for the study (Table 1), except that no urban streams were sampled.

Table 1 Calculated percentage stream length in different REC classes of climate, source of flow, geology and land cover from both the entire Rangitāiki Catchment, and from the sub sample of sites surveyed. Also shown is the calculated length of streams of different size when grouped according to stream order.

Variable	Value	Catchment stream length (%)	Surveyed stream length (%)
Stream size	Small (Order 1 and 2)	73.0	27.9
	Medium (Order 3 and 4)	20.9	42.8
	Large (Order 5+)	6.1	29.3
Climate class	CW	86.4	88.3
	WW	12.0	11.7
	WD	1.5	0
Source of flow	H	77.9	77.5
	L	20.9	20.6
	Lk	1.0	1.9
Geology	Al	0.4	0.7
	HS	6.2	2.4
	VA	93.4	96.9
Land cover	EF	50.7	48.1
	IF	28.2	28.5
	P	21.0	23.4
	U	0.1	0

2.4 Discussion

This analysis was done to ensure that selected sites were representative of waterways throughout the Rangitāiki Catchment. The REC was used to classify streams according to climate, source of flow, geology and land cover, and stream size. Streams were randomly selected from a subset of all waterways to ensure a similar range of REC classes and stream size were sampled. However, during our fieldwork, we found that many of these randomly selected streams were dry, and so new sites were selected. An additional 20 sites were not randomly selected on the basis of their REC classification, but were instead sampled to compare their contemporary condition with that found previously. From the spatial analysis, it was clear that, with the exception of waterway size, the sites selected for the survey were a good representation of the different waterways found throughout the catchment.

Although the number of small waterways sampled was considerably smaller than what was predicted from the REC, this disparity simply reflects the fact that many of the small streams were dry, and the fact that the REC could not model this with any certainty. However, 43% of streams surveyed were medium sized (orders 3 and 4), and undoubtedly small enough to still have close links to the surrounding catchment. Approximately 70% of stream length sampled in the contemporary survey was from small to medium-size streams, much greater than the 24% of stream length sampled in historic surveys. Given the fact that historic samples previously overlooked the small waterways, it is clear that the contemporary survey has helped fill a large information gap concerning our knowledge as to the ecology of these smaller waterways.

Part 3: Environmental conditions

3.1 Introduction

Assessments of stream habitat are important, as habitat represents the living space or all aquatic plants and animals. Habitat consists of the water and physical, chemical and biological environment, both instream and of the immediate streamside (or riparian) areas (Harding et al., 2009). Instream habitat includes the nature of the stream bed (e.g. mud, cobbles or boulders), the type of flow (e.g. riffles, runs or pools), the presence of fine sediment, and the degree of bank erosion and undercut banks. Riparian vegetation is also measured as this is important in providing both shade to the stream, stability to the banks, and overhanging vegetation which provides important habitat for fish spawning, and debris for instream habitat and food for invertebrates.

It is important to realise that the habitat conditions of a stream are a product of the interaction of many different factors operating at different spatial and temporal scales, and in a distinct hierarchy (Frissell et al., 1986). Factors such as climate, geology, topography and land use directly control the shape and flow dynamics of a river, and operate at the scale of the catchment, and act over long term timescales (Figure 3). For example, streams in areas of high rainfall will have very different flow dynamics than streams in areas of low rainfall, and streams in steep catchments are fundamentally different to streams in flat catchments. Streams draining catchments dominated by easily eroded material such as pumice or mudstone will also be very different to streams draining more erosion resistant rock such as granite. This combination of climate, topography and geology all affect both habitat conditions and the biological communities at sites within a catchment (Biggs and Gerbeaux, 1993; Snelder and Biggs, 2002).

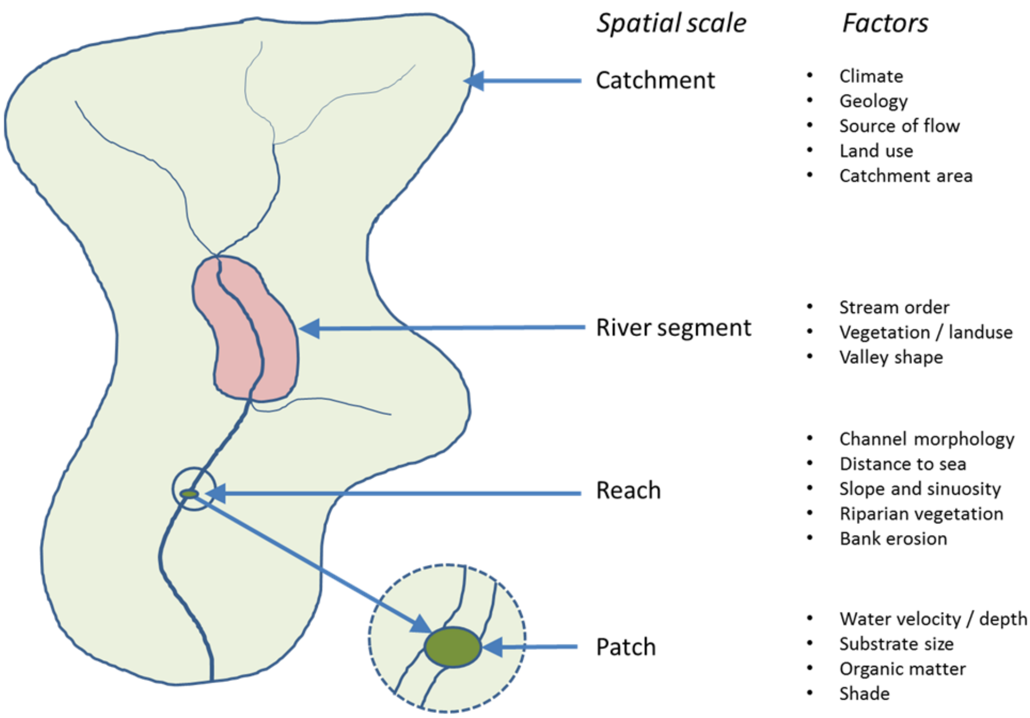


Figure 3 Illustration of the different environmental factors known to influence stream habitat conditions showing how they operate at different spatial hierarchies (modified from Harding et al., (2009)).

Other factors such as stream size, stream topography (including the valley shape, channel morphology, altitude and stream sinuosity), distance from the sea and riparian vegetation operate at a spatial scale of individual river segments (Figure 3), as these factors can change within the catchment. Thus small, relatively straight streams flowing through steep sided forested hill catchments would coalesce into larger, more sinuous streams flowing over a flat river plain, often dominated by pasture grasses. Other habitat factors such as water velocity, substrate size and shade operate at even smaller spatial scales (Figure 3). Within an individual river reach water velocity and depth can vary greatly between riffles and runs and pools (Jowett, 1993), and these small-scale hydraulic differences can have profound effects on stream ecology (Jowett and Richardson, 1990; Pridmore and Roper, 1985). Substrate size is also highly variable within a river reach, responding to differences in slope and velocity. Substrate size is of fundamental importance in influencing invertebrate and plant communities within a stream reach, as this is directly related to factors such as water velocity and substrate stability - both of which exert huge effects on ecological communities (Biggs et al., 2001; Gurtz and Wallace, 1984; Minshall, 1984).

A wide range of factors at different spatial scales was measured, ranging from small scale factors such as substrate size, stream width and depth, reach scale factors such as bank erosion, distance to sea and riparian vegetation, and large scale factors such as climate, land use and catchment area. This was done to firstly characterise the habitat conditions of the sites within the catchment, and secondly to determine whether natural groupings existed based on habitat conditions. Identification of such natural groupings may help with planning further ecological work throughout the catchment to ensure that future sites are sampled from these representative groups.

3.2 Methods

3.2.1 Field assessment

At each site, habitat assessments were made using a mixture of quantitative and categorical methods. For quantitative measurements, five transects were selected at equally spaced distances along the study reach (which was defined as 20 times the stream width). At each transect, measurements were made of stream width, and of water depth and depth of the fine bed sediment at ¼, ½ and ¾ across the width of the channel. Measurements were also made of the degree of bank undercutting, and of the distance into the stream of overhanging vegetation. Substrate size was assessed using the Wolman (1954) technique, and the resultant percentage cover of the different substrate classes was converted to a substrate index (Jowett, 1993), which ranged from 0.1 (sand or silt dominated) to 0.8 (bedrock dominated). Instream flow diversity was assessed by calculating the number and percentage of riffles, runs or pools along the study reach.

Finally, assessments were made of eleven categorical habitat parameters including: stream shade; bank stability; the width, intactness and vegetation composition of bankside and riparian buffers; stock access and stock damage, and overall stream habitat diversity (Table 2). Most of these assessments were based on assigning either a 1-4 or a 1-5 categorical score to a particular parameter. For example, “Buffer intactness” was assessed as being: 1) Completely intact; 2) Occasional breaks i.e. 1-20% gaps in reach; 3) Breaks common i.e. 20 - 50% gaps in reach; 4) Breaks frequent i.e. 50-99% gaps in reach; 5) Buffer absent, while “Stock damage” was assessed as: 1) None; 2) Low; 3) Modest; 4) High.

Where relevant (e.g., for bank stability and bankside vegetation), assessments were made of these parameters on both left and right banks. All categorical parameters were divided into four or five classes, each of which were assigned a specific score (1, 5, 10, and 20) or (1, 5, 10, 15 and 20). These scores were summed to create an overall stream habitat score (HABSCORE), with a theoretical range of 19 to 380.

Both quantitative and qualitative measures were possible in shallow, streams (<0.5 m deep) where it was possible to safely wade across. In larger rivers where deep water (> 0.5 m deep) and fast flows made it impractical to measure any of the quantitative factors, only categorical measures were made.

Table 2 List of all habitat factors measured at each of the 117 streams. Quantitative factors were measured at five transects placed across the stream, or were an assessment of the whole stream, while categorical factors were measured along the whole length of the stream, or its riparian area along the stream's left or right banks.

Variable type	Measured factor	Measured where
Quantitative	Stream width	5 transects
	Stream depth	5 transects (at 3 locations per transect)
	Degree of bank undercutting	Left and right banks at 5 transects
	Overhanging vegetation	Left and right banks at 5 transects
	Fine sediment depth	5 transects (at 3 locations per transect)
	Substrate index	Whole stream
	Flow diversity	Whole stream
	% backwaters, riffles, runs, pools	Whole stream
Categorical	Groundcover of buffer vegetation	Left and right banks
	Width of bankside buffer vegetation	Left and right banks
	Buffer intactness	Left and right banks
	Composition of streamside vegetation	Left and right banks
	Stream shading	Left and right banks
	Bank stability	Left and right banks
	Stock access	Left and right banks
	Stock damage	Left and right banks
	Instream diversity	Whole stream
	Land slope 0 to 30 m from the stream bank	Left and right banks

3.2.2 **Large scale factors**

Each sampling site had its unique NZReach code. This was used to link with the Freshwater Environments of New Zealand (FWENZ) database and the land cover database (LCDB3) to provide data on environmental factors such as climate, physical attributes such as catchment slope, elevation, distance to sea and catchment area, the percentage of major land cover classes above each sampling location and water quality information such as modelled nitrogen load (based on the CLUES data model (Semadeni-Davies et al., 2007)). This information was used to see how such large-scale factors such as climate, elevation and distance to sea, as well as more site specific factors such as nitrogen loadings were influencing the ecological condition at each site.

Prior to this analysis, the physical location of each site was checked against the REC river layer to ensure spatial accuracy of the data. This was important as some sites were located just above the confluence of two waterways, so it was necessary to ensure that the correct NZReach had been selected. This large-scale information was combined with the more site specific habitat data collected at each site, giving environmental data for both landscape factors (e.g. catchment details and land cover), and local scale factors (e.g. substrate size, riparian vegetation, and flow type). A total of 26 environmental factors were thus collected or measured in the field, or obtained from databases such as FWENZ and LCDB3 (Table 3).

3.2.3 **Statistical analyses**

As can be seen from Table 3, a total of 26 environmental factors were measured or derived from all 117 sites. These data described components of the overall environmental condition at each site which may have influenced invertebrate community composition, and therefore ecological health. In order to reduce the inherent complexity of this data (26 factors collected from 117 sites), a Principal Components Analysis (PCA) was used to reduce the dimensionality of the environmental data set to reveal any hidden structure in it. In this way we could identify what the major environmental differences were between sites that we had sampled. Prior to the PCA, all factors were standardised so that measures with different units could be analysed together. The PCA also identified what environmental parameters were responsible for any observed gradients in the data. This was done by examining correlation coefficients between the environmental factors and the PCA axis 1 and 2 scores.

All habitat data was then classified to see whether discrete groupings based on all measured and derived environmental data existed. The aim of a classification is to assign different sites to particular groupings such that sites within a group are more similar to each other and sites between groups. There are two major types of classification, agglomerative (i.e. lumping) and divisive (i.e. splitting) approaches. Agglomerative methods start with individual items (in this case sites) and group them together based on similarities between the individual sites. These combined site groups are then further recombined with other sites groups to form a larger group, and the process repeated. The initial between-site groupings are therefore based on relatively small differences between sites. Divisive classifications start with the whole set of data and progressively splits them up to form the groups at lower levels of the dendrogram. This latter strategy of classification is often regarded as the preferred approach as it uses more of the information in the data set to make the initial divisions in the data.

A TWINSPAN classification was chosen for this analysis, as it is divisive technique. Another feature of TWINSPAN is that it identifies which particular factors were most important in dividing the samples into different groups. The TWINSPAN classification was taken to the second and third divisions which created either four or eight sample groupings respectively. Classification did not go below the third level, as six of the eight sample groups at this stage contained ten or fewer sites, below which further subdivision would have resulted in a group with too few sites.

Following the classification, we used a technique called Analysis of Similarities (ANOSIM) to measure the degree to which the TWINSPAN classification explained the observed variability of the combined data set of environmental parameters. ANOSIM produces a statistic (called Global R) that is a measure of the degree to which a predictor variable (in this case the TWINSPAN groups) explains any observed variability in the data. A high global R means that a particular grouping variable explains a lot of the observed variation in the data.

Four additional categories were also created. The first was based on the REC land use classification, while the second category was based on stream size (small, medium and large). The third category called "Location" was based on spatially explicit geographical areas from where the samples had been collected. These locations represented areas of different land use, different human induced pressures (such as discharges, dams or land use), as well as natural longitudinal effects associated with increased distance to sea.

We identified seven *a priori* regions in the catchment:

1 The lower catchment below Edgecumbe, reflecting the increased tidal influence, and potential discharges from either the Fonterra Dairy Plant, or the Edgecumbe township;

2 The Te Teko reach, from the Matahina Dam to below Te Teko, reflecting the impact of the Matahina Dam;

3 The Matahina reach, from the Aniwhenua Dam to the Matahina Dam, reflecting the impact of the Aniwhenua Dam and Lake Matahina;

4 The Aniwaniwa Reach, from Lake Aniwaniwa to Murupara, reflecting potential impacts from the Murupara oxidation ponds, and farming activities on the Galatea Plains;

5 The upper Rangitāiki Catchment above the confluence with the Whirinaki River, where plantation forestry or pasture were the dominant land use;

6 The Whirinaki River, where many streams flowed through native bush;

7 The Wheao River, where many streams flowed through either native bush or plantation forestry.

The fourth category came from the BOPRC Water Quality Classification, based loosely on the ‘Water quality classes’ given in Schedule 3 of the RMA. This classification has divided all waterways in the region into nine classes, with each class having different criteria on the basis of upstream land cover, importance for fish communities, used for contact recreation or municipal water supply, or modified watercourses providing drainage functions in agricultural land.

Under the Water and Land Plan, waterways are assigned to a particular classification depending on their predominant use or value. For example, streams and rivers classified under 'natural state' have a level of protection such that no discharge is allowed to "alter the natural quality of the water" while others allow slightly lower standards for some parameters (e.g. 'contact recreation' has a lower standard for water clarity). Four water quality classifications were encountered in the 117 sites sampled, including aquatic ecosystem (70 sites), regional baseline (30 sites), natural state (nine sites), and contact recreation and stream water quality (four sites each).

ANOSIM was used to see which of these classifications best described variation to the environmental data. Finally, analysis of variance (ANOVA) was used to see which of the measured environmental parameters differed most between the classification group that best classified the samples according to their environmental parameters.

Table 3 List of the 26 environmental parameters obtained from either the REC, FWENZ, or the LCDB3, or measured in the field.

Spatial hierarchy	Source	Variable	Abbreviation
Catchment	REC	Catchment area	area_sqkm
	LCDB3	% Native bush	Us_Native
		% Exotic	Us_Exotic
		% Pasture	Us_Pastoral
	FWENZ	Mean January Air temp	SegJanAirT
		Total rain days	UsDaysRain
Segment	FWENZ	Mean reach flow	SegFlow
		Mean annual low flow	SegLowFlow
		Local reach slope	SegSlope
		Modelled N load	SegCluesN
		Water Conductivity	Conduct
		Stream Order	Order
Reach	FWENZ	Distance to coast	DsDist2Coa
	Measured	Reach sediment	Sub_Index
		Bank Undercutting	Bank_Under
		Overhanging vegetation	Veg_Over
		Water depth	Depth
		Sediment depth	Sed_Depth
		Channel width	Chan_Width
		% Backwaters	%_Back
		% Pools	%_Pool
		% Rapids	%_Rapid
		% Riffles	%_Riffle
		% Run	%_Run
		Flow diversity	Flow_Divers
		HABITAT SCORE	HABSCORE

3.3 Results

A wide range of different waterways were sampled, from very small catchments (0.3 km²) up to almost the entire Rangitāiki Catchment (area = 2,924 km^2). Distance inland also varied greatly, highlighting the fact that we sampled both small headwater streams some 200 km from the coast, to the main Rangitāiki 3.5 km below Edgecumbe. A diverse range of catchment land use was encountered, with some catchments being totally dominated by native bush, plantation forestry, or pasture (Table 4). The average percentage of native bush and plantation forestry in the catchments of all sites surveyed was similar, and double that of the average percentage of pasture (Table 4). Bank and riparian conditions were also highly variable, with some sites having no undercut banks and no bank vegetation, while other sites had heavily undercut banks and a lot of overhanging vegetation.

A wide range of stream depths, widths and flow diversity was also observed (Table 4). Conductivity was also highly variable, and ranged almost six-fold between sites. Derived CLUES nitrogen loading was also highly variable between sites, ranging almost 60 fold. This wide range in variability of stream type highlights the many different types of waterways found throughout the catchment (Figure 4), and suggests that there are very large environmental gradients that may be influencing the resultant ecological condition of each stream. Such gradients may reflect considerable differences in ecological health between the waterways.

RES_002 RES_045

RES_065 RES_026

RES_082 RES_121

Figure 4 Examples of the wide range of stream types surveyed that flowed through different land uses, and differed greatly in size. The range of streams included: small heavily modified drains on Galatea Plains (Res_002); relatively under modified streams flowing through pasture land uses (RES-045); small streams flowing through either exotic plantation (RES_065) or native bush (RES_026); and larger rivers such as the Whirinaki near Minginui (RES_082) and the Rangitāiki below Murupara (RES_121). The location of each of these sites is shown in Appendix 1 (in green).

Table 4 Calculated minimum, maximum, and mean (plus or minus one standard deviation) of the 26 environmental factors measured or derived at each of the 117 sites.

Variable (Abbreviation)	Minimum	Mean + SD	Maximum
area_sqkm	0.3	524.9 ± 990.1	2924.2
Us_Native	0.0	37.5 ± 36.7	100
Us_Exotic	0.0	37.9 ± 36.7	100
Us_Pasture	0.0	18.8 ± 27.5	100
SegJanAirT	14.40	17.13 ± 1.29	18.90
UsDaysRain	8.98	11.73 ± 2.2	17.81
SegFlow	0.01	12.44 ± 23.3	68.26
SegLowFlow	0.00	4.46 ± 8.23	23.65
SegSlope	0.00	0.85 ± 1.23	5.15
SegCluesLogN	0.19	0.79 ± 1.61	12.52
Conduct	38.1	87.7 ± 31.7	236
Order	1	3.7 ± 2.0	8
DsDist2Coa	2.6	97 ± 48.5	197.1
Sub_Index	0.007	0.35 ± 0.14	0.68
Bank_Under	0.0	0.2 ± 0.3	2.0
Veg_Over	0.0	0.6 ± 0.8	5.0
Depth	0.1	0.5 ± 0.4	3.1
Sed_Depth	0.0	0.4 ± 0.5	3.4
Chan_Width	0.9	13.5 ± 18.1	85.0
%_Back	0.0	0.6 ± 4.1	42
%_Pool	0.0	7.6 ± 17.9	100
%_Rapid	0.0	0.7 ± 4.3	40
%_Riffle	0.0	24.8 ± 33.5	100
%_Run	0.0	65.8 ± 35.3	100
Flow_Divers	1.0	1.8 ± 0.8	4
HABSCORE	71.0	242.5 ± 80.2	381

A PCA was used to identify any major gradients in the environmental data, and to determine if any natural groupings could be made according to environmental factors. The first two axes of the PCA explained a total of 35% of total variability in the data. A major gradient along the PCA axis 1 was related to catchment area, distance to sea, and river size (in terms of flow statistics, channel width and measured water depth), as well as land cover between native bush and exotic forest. PCA axis 2 appeared to represent gradients in both land cover (and resultant predicted CLUES nitrogen yield), catchment and local stream topography, and the reach scale HABSCORE assessment (Figure 5).

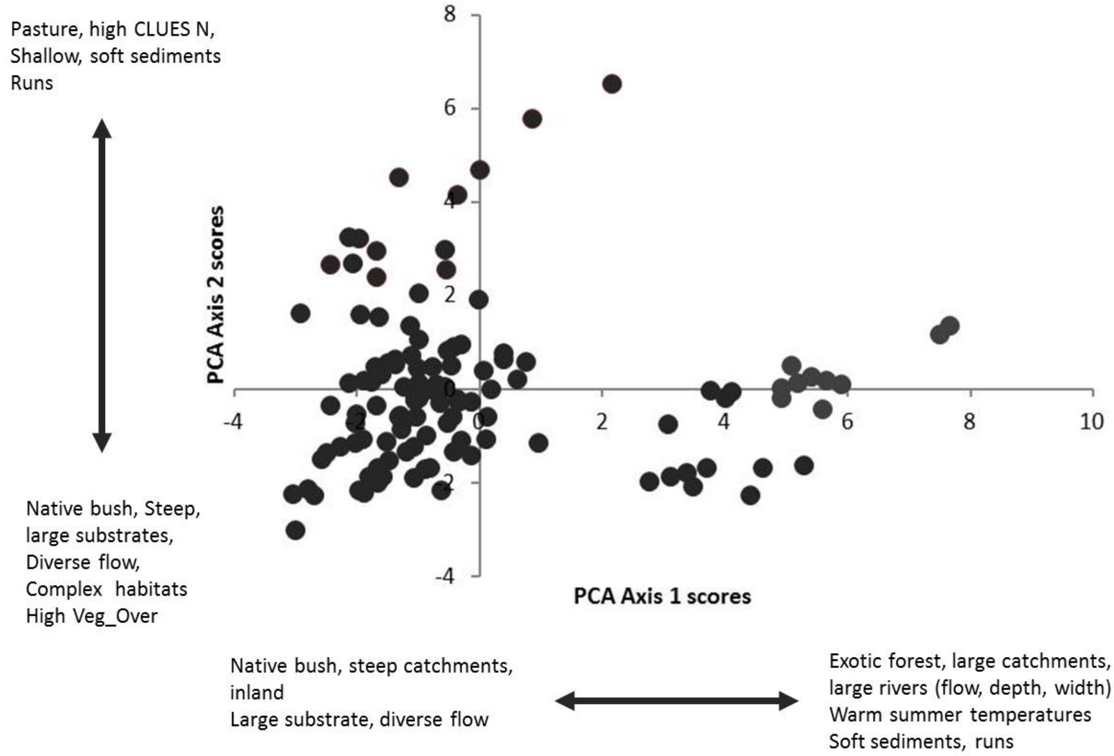


Figure 5 Results of a PCA of the standardised environmental data collected from the 117 sites throughout the Rangitāiki. Also shown are the environmental factors which were significantly correlated to the PCA axes 1 or 2 scores.

Results of the ANOSIM showed that the TWINSPAN classification at the third level explained most of the variability to the environmental data, followed by the second level TWINSPAN classification. (Table 5). These explained more of the variation than the a priori “Location” variable, or factors describing either land cover or stream size. The BOPRC Water Quality Classification explained the least variation to the environmental data. This result suggests that differences in environmental factors between streams were affected by many factors including river size, location within a catchment, land cover, and reach specific factors. This is important as it emphasises that a simple classification system based on single factors such as land cover, stream size or location within a catchment do not adequately describe the environmental variability between streams. It also suggests that that the Water Quality Classification scheme used by the council classifies waterways into broad categories that do not adequately reflect natural environmental differences between waterways.

Table 5 Results of ANOSIM showing how grouping of sites according to either the TWINSPAN classification (at the second and third divisions), the a priori location group, or groupings based on dominant land use in the catchment influenced environmental conditions in the 117 sites surveyed. Bold = highly significant values (P < 0.05).

Categorical group	Global R	P value
TWINSPAN group 3	0.74	**<.001**
TWINSPAN group 2	0.665	**<.001**
Location	0.475	**<.001**
Stream size class	0.356	**<.001**
Land cover	0.32	**<.001**
WQ_Classification	0.261	**<.001**

Examination of the two-dimensional PCA ordination when coded to the TWINSPAN classification results again emphasised the gradients present within the environmental data (Figure 6). Sites collected from the mainstem of the Rangitāiki mostly had high axis 1 scores (>2), while sites collected from smaller, more inland waterways had low axis one scores. Axis 2 represented amongst other things a land-use gradient, from native bush to exotic forestry to pasture streams. ANOVA highlighted the biggest differences between the TWINSPAN groups was based on factors to do with stream size (flow, stream order, and width), land use factors, predicted clues nitrogen, and reach scale factors such as overhanging vegetation, bank condition, flow type and sediment depth.

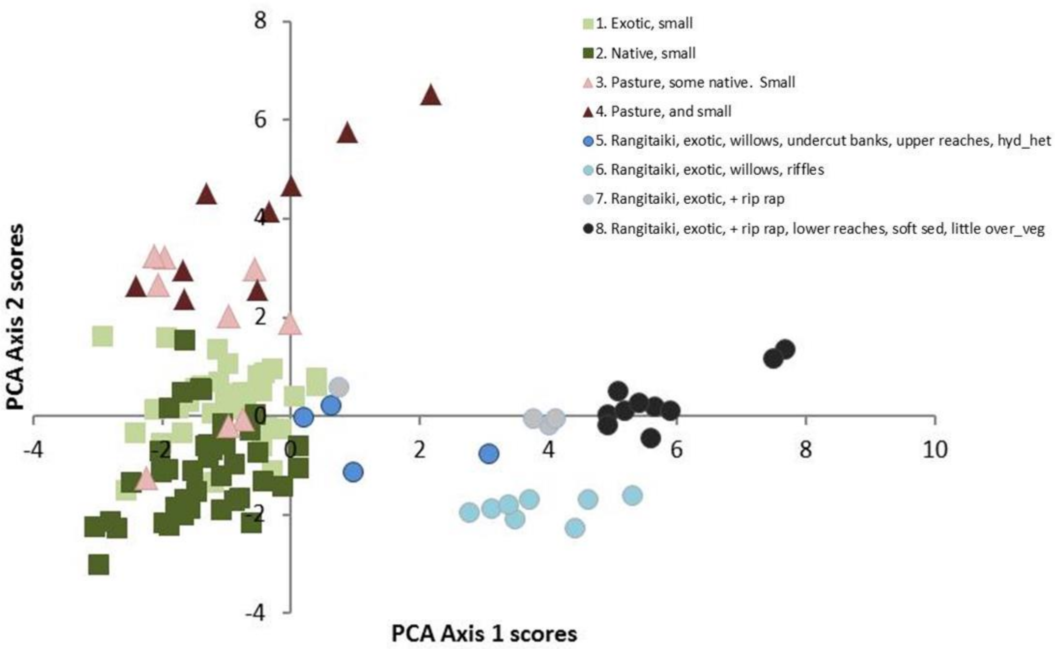


Figure 6 The PCA of standardised environmental data collected from the 117 sites throughout the Rangitāiki as coded by the allocation of sites according to groups derived from the TWINSPAN classification. Also shown are the main factors that differed between the TWINSPAN groups (hyd_het = hydraulic heterogeneity; soft sed = soft, deep sediments; over_veg = overhanging vegetation).

3.4 Discussion

Our habitat assessments highlighted the great variety of waterways throughout the Rangitāiki Catchment, ranging from small, shaded streams draining forested areas (both native and exotic), through to the larger mainstem of the Rangitāiki River with no overhead shade, and flowing through intensely farmed pasture areas in the Rangitāiki and Galatea plains, with an upstream catchment dominated plantation forestry. This variety has undoubtedly resulted in large environmental gradients that would influence the ecological condition of each stream.

Classification of the environmental factors revealed the existence of eight discrete groups that differed in regards to catchment, segment and reach scale factors. Catchment scale factors included catchment area and land cover, segment scale factors included distance to sea and river size (in terms of flow statistics, channel width and measured water depth) and predicted CLUES nitrogen yield. Reach scale factors included sediment size, presence of overhanging vegetation, bank condition, hydraulic heterogeneity and percentage of riffles and runs. All these environmental factors contributed to dissimilarities between the derived TWINSPAN classification groups.

The derived TWINSPAN classification groups explained more of the variation of environmental data than the other classifications examined, including those based on location in the catchment, land cover and stream size. The location classification was based on a spatially explicit set of sites arranged in discrete parts of the catchment that were thought to be subject to different land use practices, and other human induced pressures (such as discharges, dams or land use). Despite being spatially-based, this classification did not adequately consider factors such as land cover and stream size, which varied within each group. For example, the Aniwaniwa and upper reaches of the catchment contained sites that drained all three land-use categories, as well is containing both large and small waterways. As such, the location category alone would not be a particularly strong classification to base future sampling protocols on.

Examination of sites when coded to the TWINSPAN classification clearly showed that sites within a particular class were found throughout the catchment (Figure 7). As such this classification was not spatially explicit, but based instead on environmental factors reflecting differences in land use, stream size, and patch scale habitat factors. These differences occurred throughout the catchment, and were generally not restricted to specific areas. For example, sites belonging to TWINSPAN group 1 (small streams draining exotic forest) were found throughout the Rangitāiki Catchment, emphasising the extent of plantation forestry. Sites in TWINSPAN group 2 (small streams draining native forest) were found predominantly in streams draining the Whirinaki State Forest, Urewera National Park, and Ikawhenua Ranges. These streams had the lowest predicted CLUES N loading, and were generally very small in terms of estimated flow and measured width. Streams in TWINSPAN group 4 (small streams draining pasture catchments) had the greatest spatial distribution throughout the catchment (Figure 7), being found both in the uppermost reaches in Lochinvar Station, as well as in farmland on the coast. These streams were characterised by high predicted CLUES N loading, a low amount of overhanging vegetation, and a dominance of fine soft sediments. Land-use changes affect a wide range of environmental factors such as shade, habitat for fish, soil erosion, bed siltation, nutrient levels (e.g., (Parkyn and Wilcock, 2004; Quinn and Cooper, 1997), so it was not surprising to see a major gradient in the ordination and classification that represented land-use changes.

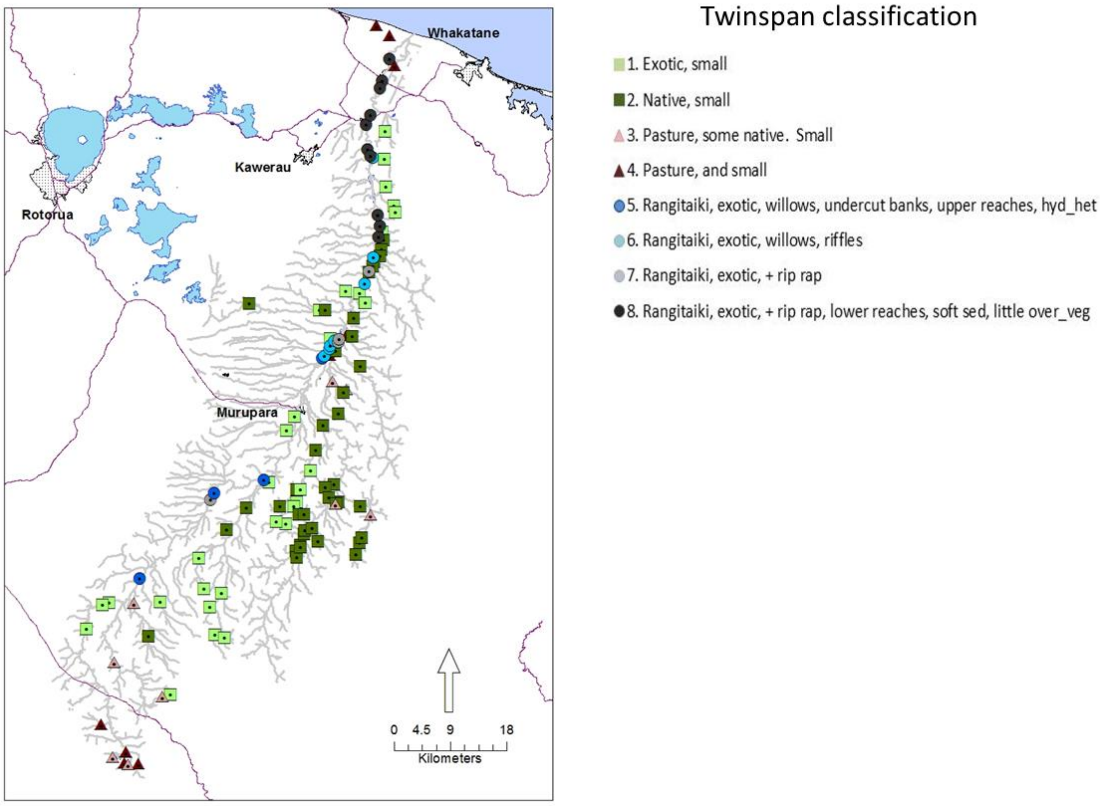


Figure 7 Map showing the location of each sampling site when coded to its TWINSPAN classification. Abbreviations as in Figure 6.

The other major gradient observed in the data was one which reflected river size. Thus, sites from the mainstem of the Rangitāiki (even in the upper reaches near the confluence with the Wheao River) formed discrete clusters from the smaller tributary streams, irrespective of land-use. The major gradients observed in the mainstem sites in the Rangitāiki River reflected a combination of flow variability, bank morphology and presence of overhanging vegetation. Bank undercutting was lowest in TWINSPAN groups 7 and 8 (most likely reflecting the presence of riprap), and highest in TWINSPAN group 5 in the upper Rangitāiki. Here, banks were mostly natural, and riprap was absent. TWINSPAN groups 5 and 6 had high cover of overhanging vegetation, reflecting the dominance of willows in this part of the Rangitāiki. In contrast, sites in the lower Rangitāiki (TWINSPAN group 8) had the lowest cover of overhanging vegetation, reflecting the highly modified stop banks generally covered with unmanaged grass in this area. Finally, sites in TWINSPAN groups 7 and 8 had the lowest flow diversity, being dominated by deep, slow flowing runs. This contrasts greatly with sites in the upper Rangitāiki Catchment (TWINSPAN groups 5 and 6) where flow diversity was very high, with a mixture of riffles, runs and pools.

Identification of these eight groups based on measured environmental factors could be used to set specific management objectives of relevance to each group. For example, the effects of riparian vegetation diminish with increasing river size (Quinn et al., 2001), particularly for functions such as shade. It thus makes little sense in setting management objectives to plant riparian vegetation along the larger rivers in an attempt to provide shade. The different groups would also be useful to help set realistic restoration goals for different waterways. Thus, there is little point in setting restoration objectives for pasture streams in TWINSPAN group 4 that do not consider the fact that streams in this group are typically characterised by slow flows, and deep soft sediments.

Such conditions could inherently limit the effectiveness of restoration activities if they were focused on activities that would not address the potential limiting environmental pressure. Such a situation was observed by Suren et al (2005) where riparian planting along urban streams did little to affect habitat factors such as substrate size and flow. Because of this, responses of the biological community to this restoration work was also limited (Suren and McMurtrie, 2005), especially as urban stormwater was still flowing into the streams and constraining the resultant communities.

It is recommended that any future management of waterways throughout the Rangitāiki Catchment use some form of classification to help set realistic management and restoration objectives of streams within a specific group. Such a classification system could be based on the one presented here, which classified sites into natural groupings based on a mixture of factors such as land cover, river size, bank and riparian conditions, flow diversity and substrate nature. Specific objectives for managing objectives such as minimising algal blooms, setting upper limits on nutrient or bacterial loadings could thus be applied to streams in the different groups.

Finally, the results of this analysis have implications for the implementation of the NPS-FW throughout the Bay of Plenty. A key part of the implementation of the NPS-FW by BOPRC is the division of the region into nine Freshwater Management Areas (FMAs). However, the results of this analysis clearly emphasise that waterways within an individual FMA are highly variable, and as such cannot be managed at the spatial scale of an individual FMA. Therefore, some classification is needed for waterways within each FMA to set realistic management goals. Our analysis showed that the current BOPRC Water Quality Classification may not be the optimal classification to use. Consequently, consideration needs to be given by the Policy and Planning teams into investigating different spatial classifications such as the one presented here by which to help implement the requirements of the NPS-FW.

Part 4: Invertebrate communities

4.1 Introduction

Traditional assessments of stream “health” were focused on analysing water quality, with particular emphasis on parameters such as nutrients, levels of dissolved oxygen, stream pH, and bacterial loadings. These parameters are important as they can influence the resultant ecological communities in the stream. For example, high nutrients can often lead to excess algal blooms, and low oxygen levels can result in fish deaths. Although these water quality parameters provide detailed information about the water quality conditions and a site at a particular time, water quality is highly variable. Suspended sediment and bacterial loads are often much higher during rainfall events than at base flow, and parameters such as pH and dissolved oxygen can vary greatly as plants produce oxygen during the day, but respire at night. Because of this variability, water quality monitoring needs to be conducted over long periods to ensure that fluctuations due to either rainfall or normal seasonal patterns are considered. An additional challenge with relying solely on water quality data is the need to clearly link it to an observed ecological effect.

In contrast, aquatic invertebrates (Figure 8) such as larval and adult stages of aquatic insects, snails, worms, and crustaceans (such as shrimp, or koura) are relatively long-lived (months to years), relatively sedentary, and display a wide range of ecological tolerances for different physical, chemical and hydrological conditions. Because of this, individual animals are exposed to a wide range of stressors and changes that occur in rivers and streams over the course of their life, irrespective of when these occur and how long they occur for. Some invertebrates can tolerate different stressors better than others, and so sensitive animals will be absent from streams where these stressors are found. Freshwater invertebrates are therefore ideal organisms to use as indicators of stream ecological condition, as they act as biological integrators of a stream’s antecedent physical, chemical, and hydrological characteristics. Thus, the health of a waterway can be determined by the types of invertebrates found in it.

The aims of this section of the study were to:

1. Characterise the ecological condition of waterways in the Rangitāiki Catchment;
2. Determine factors affecting the ecological condition of waterways;
3. Detect any temporal trends in changes in ecological health over time;
4. Quantify ecological pressures throughout the catchment.

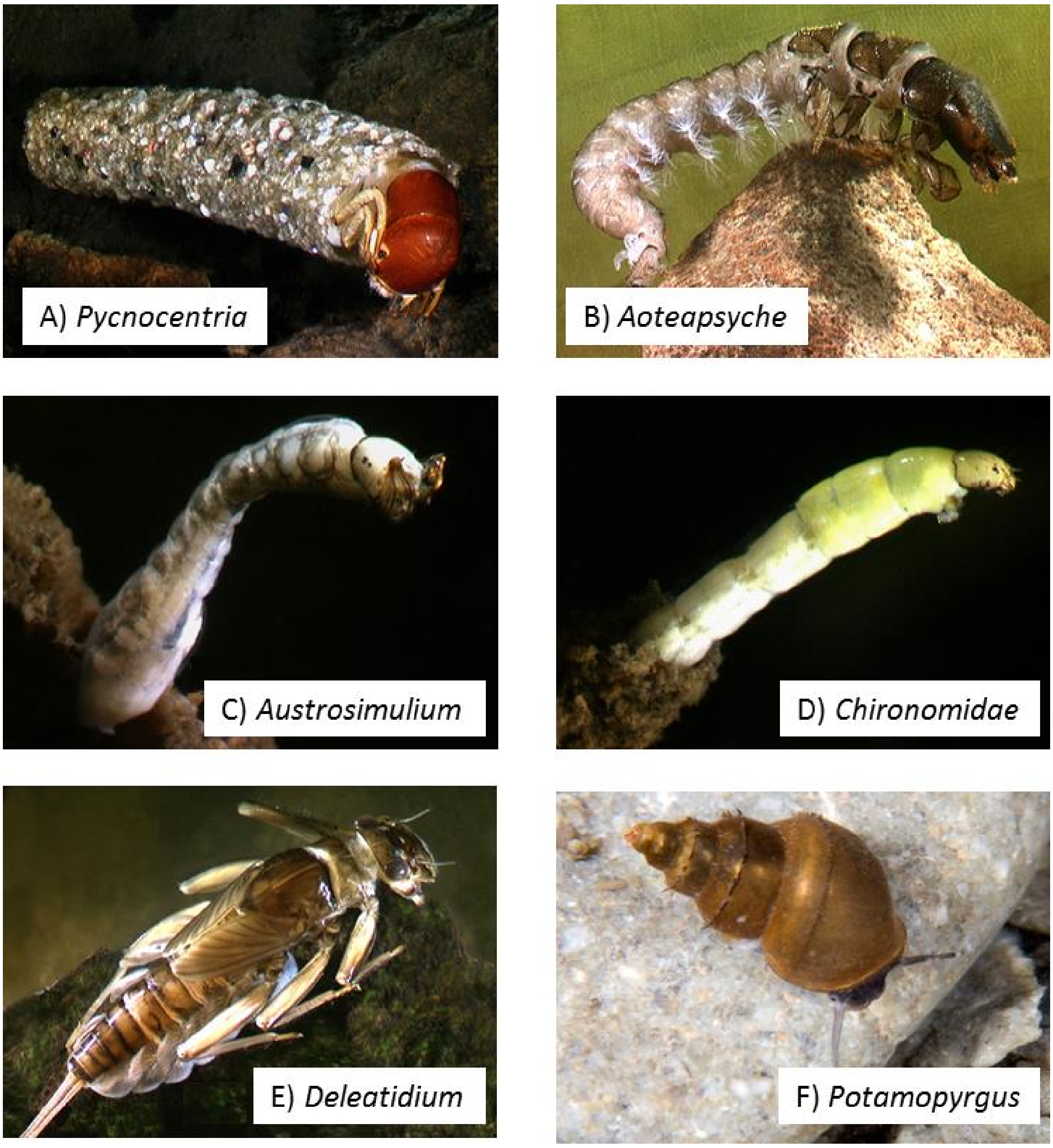


Figure 8 *Examples of aquatic invertebrates commonly collected from streams throughout the Rangitāiki Catchment, showing the wide range of aquatic insects including caddisflies (A and B), true-flies (C and D) a mayfly (E) and snail (F). Photos courtesy of Stephen Moore, Landcare Research Ltd.*

4.2 Field methods

4.2.1 Assessment of current state

Invertebrate samples were collected from each site once during an eight month period from June 2013 to January 2014 to provide a "snapshot" of the current ecological state of mainstem rivers, their large tributaries and smaller headwater streams. A total of 117 sites were sampled throughout the catchment (Figure 9). Most of the samples (95) were collected between June and August from small tributaries streams and larger rivers that flowed into the Rangitāiki. It was not possible to collect invertebrate samples from the mainstem of the Rangitāiki during this time due to logistical reasons. Instead, an additional 22 samples were collected from the mainstem river between Murupara and the Thornton Bridge during a two week period in late January and early February 2014.

The fact that sampling was undertaken over an eight month period is unlikely to have hindered our ability to adequately assess the ecological condition of each stream, as many New Zealand aquatic invertebrates have non-seasonal life cycles (Towns, 1981, 1983; Winterbourn and Harding, 1993), meaning that they could be collected with equal ability at any time of the year. Furthermore, Scarsbrook (2002) showed that invertebrate communities fluctuate around a relatively stable state, with little evidence of trajectories or sudden shifts. A similar finding was also highlighted by Winterbourn (1997) in a five-year study of invertebrate communities in three mountain streams. Other studies (Armitage and Gunn, 1996; Weatherley and Ormerod, 1990) have reported only slight changes in community composition in streams where habitat conditions remain relatively constant, and confirm observations that communities undergo significant changes in composition only when habitat conditions change significantly.

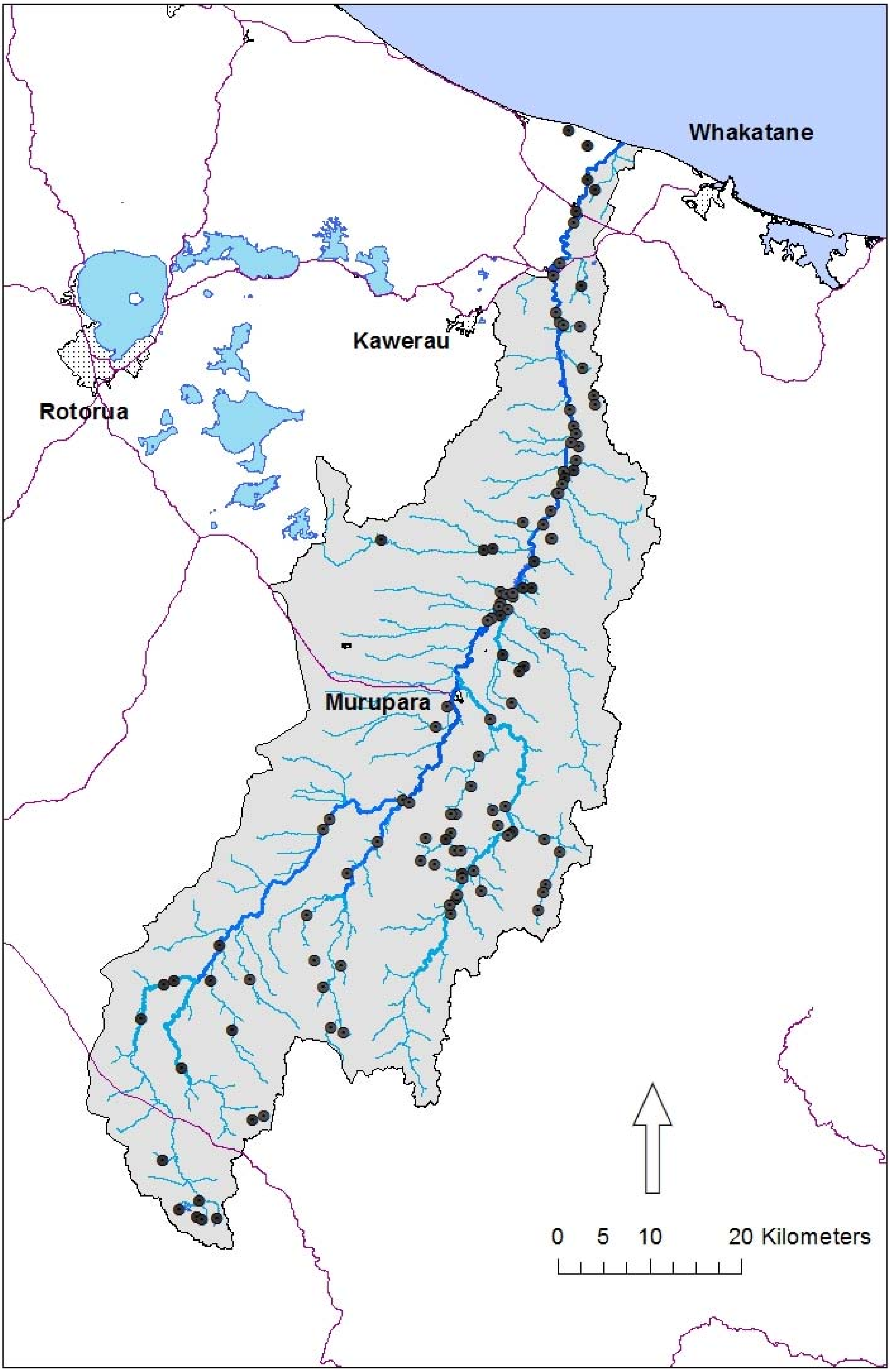


Figure 9 *Map of the Rangitāiki Catchment showing its spatial extent within the Bay of Plenty, and the location of all invertebrate sampling sites collected in the contemporary survey.*

The only caveat with using invertebrates as indicators of ecological condition is that samples not be collected after significant flood events. This is because high flows can often wash away many of the different animals present, so that absence from a particular site is a reflection more of the past flow regime rather than the physical conditions at a site. A common definition of a “flood event” is when flows exceed three times the river’s long term median flow (FRE3). Floods this size or greater have been shown to have consistent effects on invertebrate and algal communities in streams throughout New Zealand (Clausen and Biggs, 1997). Examination of river flows in the Whirinaki Catchment, midway in the Rangitāiki, showed that flows during the study were generally low, and apart from a relatively small flood on 19 June 2013 (Figure 10), flows appeared to be generally low and stable during the winter-spring collection phase. This suggests that the three month period between sample collection for this phase of the study would have done little to affect the overall results. A larger flood (47.9 m^3/s) occurred 13 September 2013 (Figure 10), which was likely to have disrupted at least some components of the invertebrate community. The next set of samples from the mainstem of the Rangitāiki were collected in late January or early February, over four months since this last flood. Based on observations of the recovery of invertebrate communities from high flows (Sagar, 1986; Scrimgeour et al., 1988; Suren and Jowett, 2006), the invertebrate communities in the river would have recovered from any effects of the large September flood.

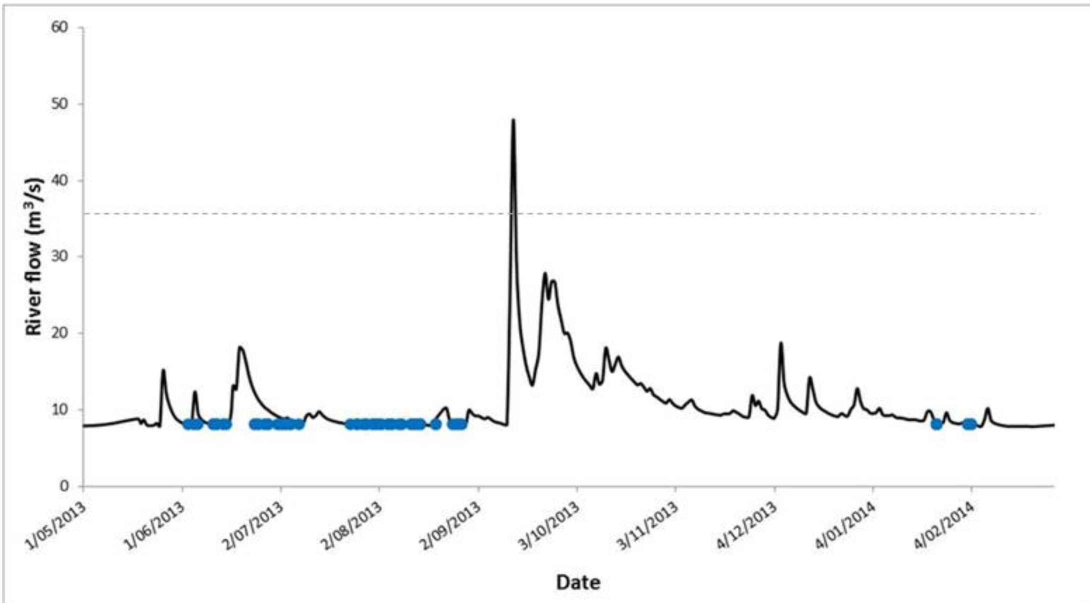


Figure 10 *Flow hydrograph of the Whirinaki River, midway up the Rangitāiki Catchment showing mean daily flows, and individual sampling dates (blue circles). Also shown is the magnitude of the FRE3 flood (dotted line), above which significant disturbance to ecological communities is possible.*

Invertebrate sampling can either be quantitative, where a known area of stream bed is sampled using specialised devices such as Surber Samplers, or semi-quantitative, where different habitats in the stream are sampled but where the area of stream bed is not quantified. Collection of semi-quantitative samples allows a much wider range of habitats to be sampled from a stream than would be possible if quantitative sampling were done. This is because quantitative samples collected with Surber Samplers can only be taken from large streams with relatively uniform substrates. Quantitative samples allow estimates of invertebrate density to be made, as well as describing the different types of invertebrates living in a waterway.

Semi-quantitative sampling is used to characterise the different types of invertebrates living in a waterway and their relative abundance to each other, rather than their actual density. However, relative abundance information is more than sufficient to be able to detect changes to invertebrate community composition as a result of pressures associated with activities such as land use change. Because of this, semi-quantitative sampling is most commonly done by regional councils as part of their state of environment monitoring programmes (Stark et al., 2001).

Semi-quantitative samples were collected using a triangular kick net (mesh size – 0.5 mm) placed into the stream, and disturbing the substrate immediately above the net (Figure 11). All dislodged material (both invertebrates and organic matter) was subsequently collected into appropriately labelled plastic bottles and preserved with isopropyl alcohol. Care was taken to sample as many different habitat types in each waterway, including the stream bed, macrophytes, overhanging vegetation, and any submerged logs or debris jams. These habitat types were sampled in proportion to their occurrence. Only a single pooled sample was collected from each site, so that approximately 1 m² of stream bed or organic material was sampled.

Figure 11 Invertebrates were collected from the streambed by disturbing the substrate immediately above a triangular net, which captured all dislodged material (both invertebrates and organic matter).

All samples were processed by a modification of Protocol P2 (Stark et al., 2001), which involved counting and identifying the first 200 invertebrates in a sample. The modification to this method was to first sieve the contents of each sample through a 0.5 mm sieve, and then examine the contents of each sieve separately. This was done to minimise any bias towards only collecting and counting larger specimens. All invertebrates were identified down to genera, or levels of taxonomic resolution consistent with that of Stark (1996).

4.2.2 Assessment of temporal changes

A number of studies have surveyed invertebrate communities at sites throughout the catchment as part of consent investigations, impact assessments, or state of environment monitoring (Table 6). A total of 20 sites which had previously been surveyed were subsequently resurveyed so that changes in ecological condition over time could be assessed. Comparison of the dates of the previous survey to the contemporary survey showed that between 12 and 38 years had passed between the surveys.

These historic sites were resampled to help assess temporal variation to their ecological condition. Where possible, sites were sampled at the same physical location, however some sites were inaccessible due to forestry roads being heavily overgrown, or bridges removed. Under such conditions, samples were collected from the same waterway as close as access would allow - usually within less than 500 m. Such a small spatial discrepancy between sample locations was assumed to have little or no effect as to the overall invertebrate community composition in that reach of the river, thus making it possible to still compare contemporary communities with those collected previously.

Table 6 List of historic studies undertaken in the Rangitāiki Catchment and the number of sites that had been sampled which were resurveyed as part of the contemporary survey. Also shown is the number of years separating the two data sources.

Organisation	Study	Reason	Years between surveys	Number of sites
Bioresearchers	Upper Rangitāiki and Wheao Hydroelectric Ecological investigations	Consent investigations	30 and 38	10
Bioresearchers	Effects of the discharge of dairy effluent at Edgecumbe	Impact assessment	35	1
Kingett-Mitchell	Re-consenting process for the Matahina Dam	Consent investigations	12	3
BOPRC	Natural Environment Resources Monitoring Network (NERMN) Programme	State of Environment monitoring	12	5
NIWA	National Water Quality Monitoring Network	State of Environment monitoring	23 (on-going)	1

Three other sites (the Haumea River, and two sites in the Rangitāiki River near Murupara) were not sampled in the latest study, but provided long term data (up to 20 years at the Rangitāiki at Murupara site) which could be used to detect changes to the invertebrate communities over time.

4.3 Statistical analysis

Achieving the aims of this study relied on data generated from the contemporary invertebrate survey and obtained from historic surveys. The contemporary data consisted of a large data matrix of all the different invertebrates found at the 117 sites throughout the catchment, while the historic data consisted of a similar data matrix collected from 20 sites where samples had been collected over times spanning 12–38 years ago. Because we collected invertebrates using semi-quantitative methods, all data from both the contemporary and historic surveys were converted into percentage abundance at each site.

When performing statistical tests on data, it is important to ensure that the data is normally distributed, or follows the shape of a bell curve. This means that observed data on, for example, the % abundance of caddisflies in a stream are likely to fall around the middle value, known as the mean or median of the normal distribution, but are also as equally likely to fall to the left or right of that middle value. However, much ecological data is highly skewed, and many values might fall to either the left or right of the mean value. Under such circumstances, mathematical transformations of the data can be used to ensure it is normally distributed. Because many statistical tests rely on data to be normally distributed, all data was examined for normality and transformed where necessary.

4.3.1 Assessment of current state

A fundamental requirement of this work rested on the ability to quantify the ecological condition of streams. A major challenge in using invertebrates to indicate ecological condition is the need to convey often complex information about the relative abundance of many different invertebrates found in each stream into simple numerical indices (or metrics) that summarise certain attributes of the invertebrate community. Such metrics include the macroinvertebrate community index (MCI) and its quantitative variant (QMCI (Stark, 1985; Stark and Maxted, 2007a)), as well as the number and percentage of insect taxa such as Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies). This is referred to as the number (or percentage) of EPT taxa. These insect taxa are especially sensitive to effects of land use changes, and are often reduced in number in streams draining pasture or urban catchments (Hall et al., 2001; Quinn and Cooper, 1997; Collier and Winterbourn, 2000; Scarsbrook and Halliday, 1999). Once these metrics were calculated for each site, we used them to act as surrogates for overall stream health using classes provided by guidelines or protocols for the use of those methods (see Stark and Maxted, 2007b). We were thus able to measure the current ecological condition of waterways throughout the catchment, and to see how this changed throughout the catchment, or in streams draining different land uses.

4.3.2 Interactions with environmental factors

Following assessment of stream ecological condition, we wanted to determine which of the measured environmental factors were influencing the invertebrate communities. This analysis was done in five steps.

Firstly, the statistical technique of ordination was used to simplify the large invertebrate data matrix (104 taxa x 117 sites) into a two dimensional space, so that any patterns in the invertebrate community data could be visualised. In this way, sites with similar species composition would be plotted close together, and sites with dissimilar species plotted far apart. Ordination is a useful technique for detecting environmental gradients in ecological data, and for assessing how ecological communities respond to these environmental gradients. For this analysis, Detrended Correspondence Analysis (DCA) ordination was used (McCune and Mefford, 1997). A useful output of the ordination is a statistic called the "gradient length" for each axis. This is a surrogate measure for the degree to which the taxonomic composition changes along each ordination axis. A gradient length of three or greater suggests a complete species turnover along a particular axis. Thus, samples at one end of an axis will have no species in common with samples at the other end of the axis.

Secondly, the relative abundance of each taxa was regressed against the ordination axis 1 and 2 scores to determine which taxa were responding to environmental gradients. For example, a positive correlation for a particular taxa (such as the mayfly *Deleatidium*) for the axis 1 ordination scores means that *Deleatidium* densities increase in samples with high axis 1 scores.

Thirdly, measured environmental factors were regressed against the axis 1 and 2 scores to see which factors were responsible for the observed gradients in the ecological data. As with the invertebrate data, a positive correlation for a particular environmental factor (for example, substrate size) for axis 1 ordination scores means that substrate size increases in samples with high axis 1 scores. By inference, any environmental factors that were correlated to the ordination scores represent environmental gradients that have shaped the structure and composition of the invertebrate communities.

Fourthly, stepwise multiple regression (SMR) analysis was used to determine which measured environmental factors were significantly related to the calculated biotic metrics (MCI, EPT, QMCI and % EPT) and the DCA axis 1 and 2 ordination scores. SMR is a statistical technique to determine which of many individual predictor factors are significantly correlated to a single dependent variable. The goal of SMR is to choose a small subset of predictor factors from a larger set so that the resulting regression model is simple, yet has good predictive ability. There are two forms of SMR:

1 Forward SMR looks at the correlation of each predictive variable in turn against the dependent variable, and selects the variable with the strongest predictive power to enter into the model. It then repeats this process looking at the correlation of the remaining predictive factors, and adds the next strongest predictive variable to the model.

2 Backwards SMR looks at the correlation of all predictive factors in a combined model against the dependent variable, and removes the variable with the weakest predictive power from the model. It then repeats this process looking at the correlation of the remaining predictive factors, and removes the next weakest variable in the model.

Both forward and backwards SMR usually give the same results, but there are times when the different methods yield different predictive models. Under such situations, the model with the strongest predictive model, as assessed by the model F-ratio, was chosen. SMR was only done for quantitative environmental factors, which were all standardised to their means prior to the analysis. This standardisation made it possible to analyse data with different units of measurement (e.g. percentage data, flow data, climatic data).

Lastly, ANOSIM was used to determine how variability to the invertebrate communities (expressed as ordination scores) differed between the different REC classifications for climate, source of flow, geology and land cover, and stream size (small, medium and large). The a priori "Location" classification was also tested, as was the TWINSPAN classification of environmental data and the BOPRC water quality classification classes.

4.3.3 Assessment of temporal changes

Twenty of the 117 sites sampled throughout the Rangitāiki River Catchment had been sampled previously as part of consent investigations, impact assessments, the council's SOE monitoring programme, or as one of NIWA's national water quality monitoring network sampling sites. The earliest samples from the other historic studies had been collected from some sites (e.g. the Wheao and upper Rangitāiki River) in 1976, 38 years ago. It was thus possible to determine whether stream health had changed at these sites over time. Three other sites (e.g. Rangitāiki at Murupara) were not sampled in the latest study, but provided long term data (up to 20 years at the Rangitāiki at Murupara site) which could be used to detect changes to the invertebrate communities over time.

The taxonomic resolution of each study was examined and all taxa reassigned to consistent levels of taxonomic resolution. For example, early ecological surveys simply grouped all midges under the taxa "Chironomidae", whereas contemporary surveys often identify midges down to either tribe or genus. Following the creation of a combined dataset with similar levels of taxonomic resolution, changes in invertebrate communities were assessed by comparing the invertebrate communities found in the contemporary surveys with those collected from previous surveys. Prior to analysis, all sites were allocated to their appropriate REC land use class, giving a mixture of sites in plantation forestry, pasture and native bush. Grouping by land use was done as streams draining catchments dominated by pasture may have changed more as a result of land use activities than streams in catchments dominated by native bush, where land use activities would be minor.

Detection of temporal trends was done in two ways. Firstly, data for biotic metrics such as MCI and EPT, and their quantitative variants (QMCI and %-EPT) from sites where less than eight historic samples had been collected were averaged over time, and compared with the data collected in 2013 using a two sample t-test. Eleven sites were analysed in this way, with eight sites in plantation forestry, and three in pasture. This analysis addressed the hypothesis that "*there was no difference in calculated biotic metrics over time between samples in streams draining different land use classes*". Secondly, data from the other nine sites had been collected more than eight times between 1984 and 2013 were examined for any trends in calculated biotic metrics using nonparametric Spearman's rank correlations, as recommended by Stark and Maxted (2007b). This method is useful when undertaking analyses to see whether or not biotic metrics have deteriorated, improved or stay the same. It has the advantage in that it can be used when factors are not normally distributed. It is also not very sensitive to outliers, which are observations within your data that do not follow the usual pattern. Significant trends for individual sites are reported at the 95% level of confidence ($p < 0.05$). This addressed the hypothesis that "*there were no significant trends in biotic metrics over time at individual sites*".

4.3.4 Quantifying ecological pressure

Invertebrate communities throughout the Rangitāiki Catchment, and indeed throughout New Zealand, are influenced by a mixture of natural environmental variability, and by human pressure (e.g., Collier and Winterbourn, 2000; Quinn and Cooper, 1997). Obviously, the more human pressures within a catchment, the more the invertebrate community would change from that which could be expected in the absence of this pressure. In the Rangitāiki survey, most of the sites draining native bush could be regarded as reference sites, and as such their observed ecological condition (e.g. MCI scores) would represent the best attainable condition. There is, however, a lack of information on what the ecological health of streams draining pasture and plantation pine forests would have been in the absence of these land use changes, so it is potentially difficult to assess the degree to which they have changed. To overcome this, Clapcott et al (2011) have developed models to predict invertebrate communities, and in particular MCI scores, throughout New Zealand in the absence of human activities. These models have been based on statistically established relationships between land uses, environmental variability and ecological responses for individual stream types.

By examining differences in MCI scores observed in this survey (MCI_{obs}) with predicted MCI scores in natural streams without human disturbance (MCI_{nat}), it was possible to assess the magnitude of change to instream health. This was done by assessing the difference between MCI_{obs} and MCI_{nat} in each stream, and calculating the average difference of all streams within each of the 8 TWINSPAN classification groups that best explained variability in the invertebrate community (See Section 4.4.2).

4.4 Results

4.4.1 Assessment of current state

A total of 104 invertebrate taxa were found in the 117 sites surveyed. On average, 18 taxa were found in each stream, with the fewest taxa (six) being found in a small stream in Kopuriki flowing into the head of Lake Aniwaniwa, and the most taxa (34) found in the Wharetiki Stream, flowing from the Ikawhenua Ranges through native bush. The ten most common taxa included four mayflies (*Deleatidium, Coloburiscus, Zephlebia*, and *Austroclima*), three caddisflies (*Pycnocentria*, *Aoteapsyche* and *Orthopsyche*), the blackfly *Austrosimulium*, the freshwater snail *Potamopyrgus*, and Orthoclad midges (Table 7). All these taxa were regarded as being common or abundant, being found with a relative abundance of 3% or more. The majority of other taxa were found with a relative abundance of less than 3%, and so were regarded as being found only occasionally. The most widespread invertebrate was the caddisfly *Pycnocentria*, found in 104 of the sites that were sampled, followed by the mayflies *Coloburiscus, Deleatidium, Austroclima* and *Zephlebia* which were found in 86 or more sites (Table 7). Five other taxa were widespread and found in at least ½ of the sites sampled. The widespread distribution and high relative abundance of invertebrates such as mayflies and caddisflies suggests that many of the streams sampled during the survey were in relatively good ecological condition.

Table 7 List of the ten most common invertebrate taxa collected from the 117 sites sampled throughout the Rangitāiki Catchment. The tables shows the most common taxa ranked according to average percentage abundance, or the number of sites they were found in.

Taxa	Average percentage abundance	Taxa	Number of sites
Pycnocentria	14.4	*Pycnocentria*	104
Potamopyrgus	9.7	*Coloburiscus*	92
Deleatidium	8.9	*Austroclima*	90
Coloburiscus	6.7	*Deleatidium*	86
Zephlebia	6.6	*Zephlebia*	86
Austroclima	6.0	Elmidae	76
Austrosimulium	4.5	Orthocladiinae	75
Aoteapsyche	4.1	*Hydrobiosis*	67
Orthocladiinae	3.5	*Potamopyrgus*	64
Orthopsyche	3.1	*Archichauliodes*	61

This contention is supported by examination of the calculated biotic metrics. The average MCI score was 126, and the average QMCI score was 6.2 (Table 8) - both equivalent to streams in "excellent" condition using the water quality classification limits of Stark and Maxted (2007b). Furthermore, the average number and percentage of EPT taxa at each site was also very high (Table 8), suggesting that many of these sensitive insect taxa were found at many sites throughout the catchment. A total of 88 sites had MCI scores greater than 120, representing "excellent" quality classes, while 68 sites had QMCI scores in the "excellent" category (Figure 12). A further 14 or 19 sites had MCI and QMCI scores in the "good" quality class respectively. Only four sites had MCI scores indicative of "poor" quality, while 12 sites had similarly low QMCI scores. The differences in the number of sites allocated to the four water quality classes on the basis of the MCI or QMCI scores reflects the fact that the QMCI takes into consideration relative abundance of taxa, whereas the MCI is based only on presence-absence.

Table 8 Summary statistics of calculated biotic metrics from the 117 sites sampled throughout the Rangitāiki Catchment.

	Richness	MCI score	EPT	QMCI score	P_EPT
Minimum	6	68.5	0	2.9	0
Average	18.8	126.3	11	6.2	63.1
Maximum	34	154.7	24	8.8	99.5

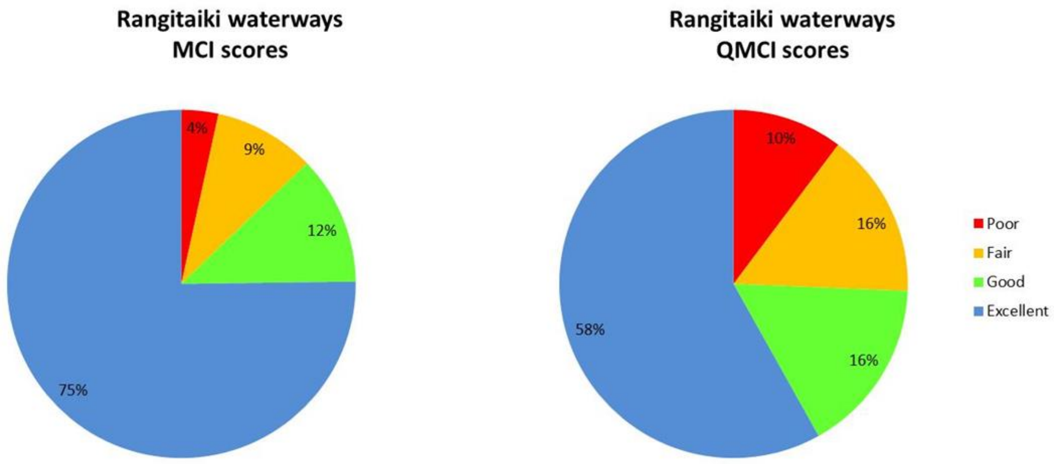


Figure 12 Pie chart representation of the percentage of sites sampled in the Rangitāiki Catchment allocated to one of the four water quality classes of Stark and Maxted (2007).

Two of the four sites with the lowest MCI scores were from streams draining pasture catchments, while the other two sites were from streams draining exotic forest. Seven of the 12 sites with the lowest QMCI scores were from streams draining exotic forest, and five from streams draining pasture. None of the 34 streams draining native bush were ranked in either the “fair” or “poor” water quality classes for the MCI, and only three streams were ranked as “fair” on the basis of their QMCI scores. Examination of the spatial distribution of MCI water quality classes generally showed that sites above Lake Matahina were ranked as being in either “good” or “excellent” condition (Figure 13), with the exception of two sites in Lochinvar Station which were assessed as being in “poor” condition. Thirteen sites above Lake Matahina were regarded as in “fair” condition. Examination of the QMCI water quality classes showed a greater spatial distribution of sites throughout the catchment of either fair or poor quality, with 25 sites above Lake Matahina being ranked as either fair or poor. Most of the samples collected from the lower Rangitāiki below the Matahina Dam were rated as being in either “fair” or “poor” condition using both the MCI and QMCI (Figure 13).

Results of the ordination showed large differences in invertebrate community composition from the 117 sites (Figure 14). Resultant gradient scores for axis 1 and 2 were both greater than three, suggesting complete species turnover in samples at the extreme ends of these axes. This would only have occurred if there were large environmental gradients present in the data. Examination of correlations between individual taxa and ordination scores (Table 9) showed that samples with low axis 1 and 2 scores contained invertebrates dominated by mayflies, stoneflies, caddisflies and toebiters. These samples were from sites dominated by steep streams flowing through native bush, which had large substrates and were generally far inland (Figure 14). Samples with high axis 1 scores were characterised by snails, blackflies, and water fleas (Cladocera). Two genera of mayflies (*Austroclima* and *Zephlebia*) were also characteristic of these sites. These samples generally came from sites flowing through pine plantation forest. Samples with high axis 2 scores were also characterised by snails, midges, micro crustacea (water fleas and seed shrimp (ostracods), worms, leeches and algal piercing caddisflies (*Oxyethira* and *Paroxyethira*). These were from large rivers in large catchments dominated by pasture land use, and were typified by high predicted nitrogen loads.

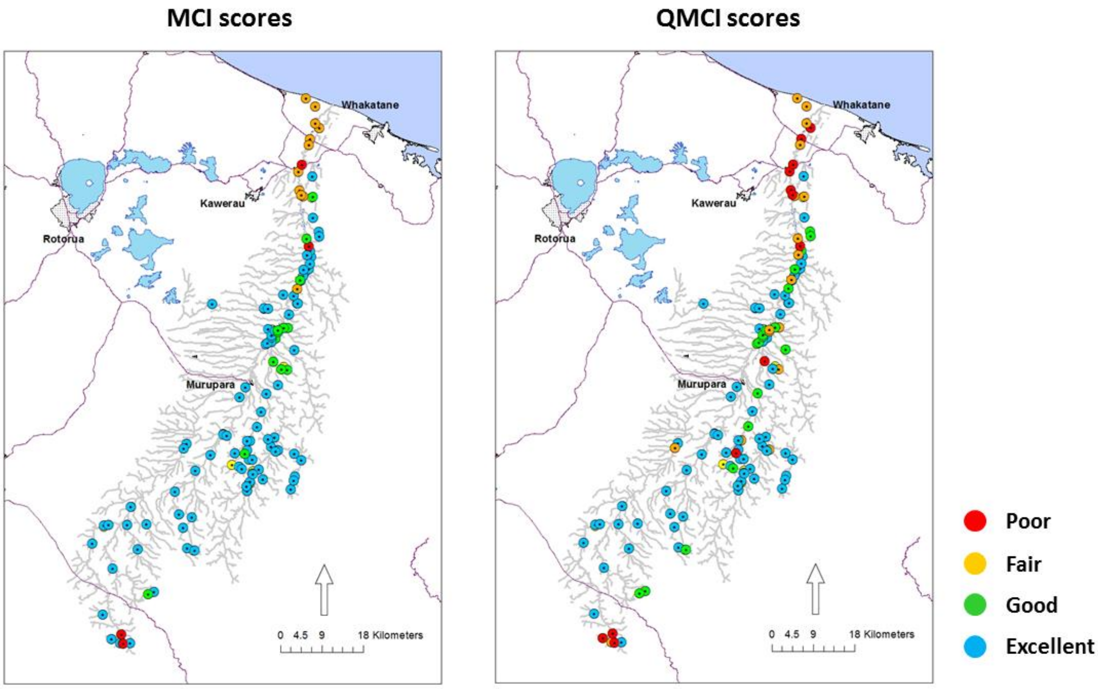


Figure 13 *Map of the Rangitāiki Catchment showing samples allocated to one of the four water quality classes of Stark and Maxted (2007) for either the MCI or QMCI scores.*

Table 9 *List of invertebrate taxa that showed significant (< 0.01) correlations to the DCA axis 1 or 2 scores, and whether each correlation was positive or negative.*

Axis 1 scores				Axis 2 scores			
Negative correlations		Positive correlations		Negative correlations		Positive correlations	
Taxa	r-value	Taxa	r-value	Taxa	r-value	Taxa	r-value
Deleatidium	-0.543	*Zephlebia*	0.489	Orthopsyche	-0.469	*Potamopyrgus*	0.602
Olinga	-0.427	*Austrosimulium*	0.467	*Austroclima*	-0.413	*Xanthocnemis*	0.531
Coloburiscus	-0.403	*Physa*	0.33	*Coloburiscus*	-0.404	*Hygraula*	0.514
Archichauliodes	-0.395	*Gyraulus*	0.327	*Pycnocentria*	-0.372	Lymnaea	0.444
Eriopterini	-0.344	*Cladocera*	0.28	*Zephlebia*	-0.338	Ostracoda	0.426
Ichthybotus	-0.3	*Austroclima*	0.272	*Austroperla*	-0.329	*Chironomus*	0.381
Zelandoperla	-0.3	*Hygraula*	0.263	Ptilodactylidae	-0.307	Copepoda	0.373
Tanytarsini	-0.287			*Hydrobiosella*	-0.279	Cladocera	0.362
Aoteapsyche	-0.28			*Archichauliodes*	-0.277	*Physa*	0.36
Aphrophila	-0.267			*Zelolessica*	-0.275	*Gyraulus*	0.358
Beraeoptera	-0.252			*Megaleptoperla*	-0.239	Oligochaeta	0.357
Psilochorema	-0.238					Sphaeriidae	0.346
						Orthocladiinae	0.344
						Sigara	0.32
						Hirudinea	0.306
						Paroxyethira	0.278
						Oxyethira	0.274
						NEMATODES	0.248
						Hexatomini	0.24
						HYDROIDS	0.237

4.4.2 Interactions with environmental factors

Regression analysis of environmental factors against the DCA axis 1 and 2 scores revealed which factors were responsible for the observed gradients in the data (Table 10). A gradient of land-use and of flow type was evident along axis 1 (Figure 14). Thus, samples with low axis 1 scores were from streams flowing through native bush, which were dominated by riffles, but which also had a high degree of flow variability. In contrast, samples with high axis 1 scores were from catchments dominated by pine plantations, and were deep rivers, dominated by runs and which had thick layers of fine bed sediments (Figure 14). These samples were typified by those in the lower Rangitāiki River, where up to 52% of the total catchment was dominated by plantation forestry. Axis 2 appeared to represent gradients of stream size (as a function of catchment area, channel width and depth, stream order, and flow statistics), stream location (as a function of distance to sea, slope, and stream order) and water chemistry (including CLUES nitrogen load and conductivity). Other important parameters along this axis also included pasture land cover, and locally measured factors such as HABSCORE, degree of bank under cutting and flow heterogeneity (Figure 14).

Table 10 List of environmental factors that showed significant (< 0.01) correlations to the DCA axis 1 or 2 scores, and whether each variable was either positively or negatively correlated to these scores.

Axis 1 scores				Axis 2 scores			
Positive correlations		Negative correlations		Positive correlations		Negative correlations	
Variable	**r-value**	**Variable**	**r-value**	**Taxa**	**r-value**	**Taxa**	**r-value**
%_Run	0.409	Hydrol_Hetero	-0.301	Area_sqkm	0.529	HABSCORE	-0.27
Exotic_Bush	0.302	%_Riff	-0.377	SegFlow	0.526	SegSlope	-0.275
Depth	0.284	Native_Bush	-0.47	SegLowFlow	0.516	ReachSed	-0.308
Sed-Dept	0.242			SegCluesLogN	0.45	Bank_Under	-0.318
				Strm_Width	0.427	SegDist2Coa	-0.321
				Depth	0.408	Hydro_Hetero	-0.403
				Sed-Dept	0.349		
				Order	0.342		
				SegJanAirT	0.323		
				Pasture	0.31		
				COND	0.305		

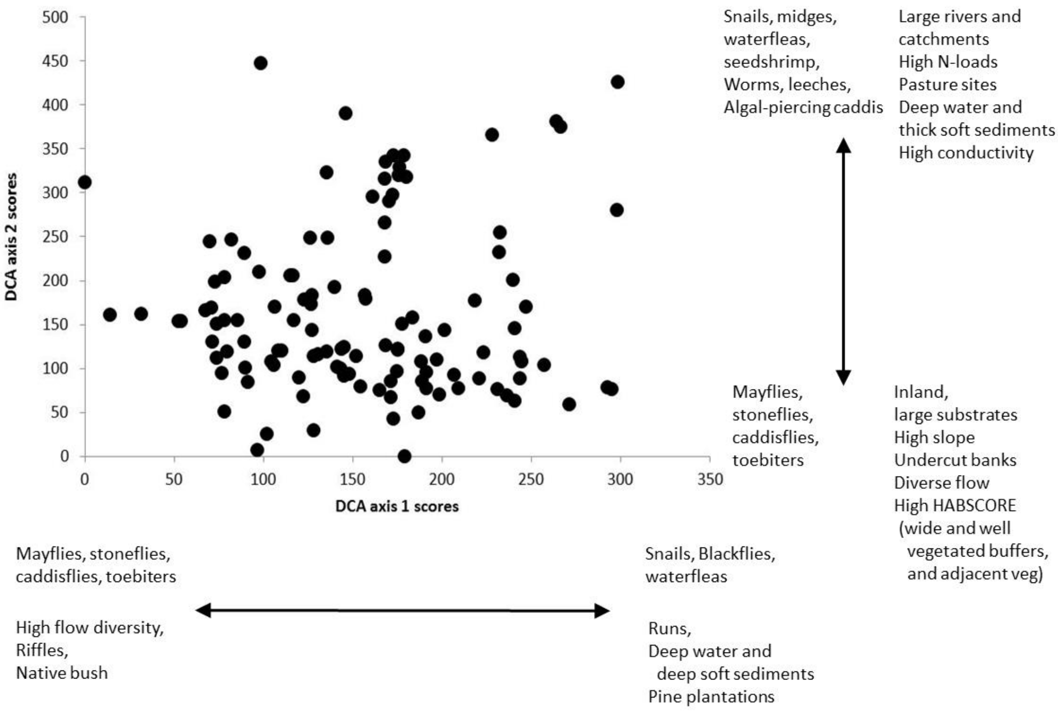


Figure 14 Results of the DCA ordination showing the sample spread on axes 1 and 2, reflecting underlying gradients in the ecological data. Also shown are the invertebrate taxa and environmental factors that showed significant correlations to these axes.

All stepwise multiple regression analyses produced significant predictive models for the biotic metrics and ordination scores (Table 11). The strength of these models was relatively high, explaining from between 43% and 65% of the variation in axis 1 scores and MCI scores respectively. All models significantly reduced the number of predictive factors from the original 26 to between five and eight factors (Table 11). The most commonly selected environmental factors included CLUES nitrogen load, degree of bank under-cutting and percentage riffles in the stream, which were found in 5 of the 6 SMR models. Predicted CLUES nitrogen load was negatively correlated to the MCI score, and the number and percentage of EPT taxa, suggesting that where predicted nitrogen loadings were high, then these metrics would be low. This variable was also positively correlated to the axis 1 and 2 scores. Samples with high scores were characterised by taxa such as snails, blackflies, midges, worms and algal piercing caddisflies.

The next most commonly selected factors were the calculated HABSCORE and predicted reach sediment size, which were found in four of the models (Table 11). Both REACHSED and HABSCORE showed strong positive relationships to the four biotic metrics, emphasising the importance of instream habitat conditions and substrate size in determining the overall ecological condition of a stream. HABSCORE values were also negatively associated with DCA axis 1 and 2 scores, meaning that sites with low axis one and two scores had high HABSCORES. Sites here were dominated by mayflies, caddisflies and stoneflies. Eight environmental factors were found in only one SMR model, suggesting that their overall contribution in structuring invertebrate communities was not particularly strong.

Table 11 Summary results of the SMR analysis showing relationships between invertebrate communities (expressed as the four biotic metrics and DCA axis 1 and 2 scores) and measured environmental factors selected in each SMR model. Also shown is the nature of the relationship between each environmental variable and invertebrate communities, as well as the predictive strength (shown as r^2) of the resultant SMR model.

MCI		**EPT**		**QMCI**		**P_EPT**		**DAC Axis 1 scores**		**DCA Axis 2 scores**	
CluesLogN	-	CluesLogN	-	ReachSed	+	USDaysRain	+	SegJanAirT	-	USDaysRain	-
ReachSed	+	ReachSed	+	HABSCORE	+	DSDist2Coa	+	CluesLogN	+	CluesLogN	+
HABSCORE	+	Bank_Under	+	Pasture	-	CluesLogN	-	Hydrol_Hetero	-	HABSCORE	-
SegFlow	-	Sed_Depth	-	Bank_Under	-	ReachSed	+	HABSCORE	-	Area_sqkm	+
SegLowFlow	+	% Riff(1)	+	Depth	-	Exotic_Bush	+	Exotic_Bush	+	Exotic_Bush	-
Bank_Under	+			% Riff(1)	+	Bank_Under	+	Depth	+	SegLowFlow	-
% Riff(1)	+					Sed_Depth	-	Strm_Width	-	Bank_Under	-
						% Riff(1)	+	% Run(1)	+	% Riff(1)	-
r^2 = 0.65		r^2 =0.562		r^2 =0.584		r^2 =0.581		r^2 =0.435		r^2 =0.633	

Results of the ANOSIM showed that the classification based on the eight groups derived from the TWINSPAN of habitat data explained most of the variability observed between the invertebrate communities (Table 12). The TWINSPAN grouping was based on a classification of the environmental factors at each site, which reflected gradients in stream size, land cover, predicted CLUES nitrogen yields, presence of overhanging vegetation, and hydraulic heterogeneity. Similar factors were also highlighted by the DCA analysis as being important in controlling invertebrate communities throughout the Rangitaiki Catchment.

Table 12 Results of ANOSIM showing how the a priori location group, REC groupings, and stream size classes influenced invertebrate communities in the 117 sites surveyed. Bold = highly significant values (P < 0.05).

Categorical group	Global R	P value
TWINSPAN Group3	0.359	**0.001**
Location	0.230	**0.001**
Climate	0.142	0.07
Source of flow	0.058	0.165
Geology	0.203	0.97
Land cover	0.120	**0.001**
Stream size class	0.106	**0.001**
Water Quality Classification	0.102	**0.02**

4.4.3 Assessment of temporal changes

Temporal differences in 11 streams were examined by t-tests to see whether there were any difference in calculated metrics between contemporary data and the average of all historic data. No differences were found between contemporary and historic values for any of the four biotic metrics from the eight sites draining plantation forestry (Table 13). Sites draining pasture streams had significantly more EPT taxa in the contemporary surveys than the historic surveys, with ten taxa being recorded only in the contemporary surveys (Table 13). Whether this reflects an actual increase in the number of EPT taxa found in these streams or just improvements in identification is unknown. However, most of the taxa recorded only in the contemporary surveys are well known and easily identified, so it is unlikely that their absence from the historic surveys reflects inaccuracies with their earlier identification. Examination of other EPT taxa found in both surveys showed no significant differences in relative abundance between the two sampling periods, suggesting that the condition of these pasture streams had not changed. Lack of consistent changes to any of the biotic metrics in both plantation forestry and pasture streams suggest that that ecological health in these streams waterways had not changed over time.

Table 13 *Differences in calculated biotic metrics from streams draining plantation forestry or pasture between contemporary survey data and historic data (mean + 1 standard deviation). Metrics which significantly differed (P < 0.05) between time periods are shown in bold and shaded.*

Land use	Time period	MCI	QMCI	EPT	% EPT
Plantation forestry	Historic	106 ± 25	5.5 ± 1.4	7.3 ± 4.5	48.5 ± 43
(n = 8)	Contemporary	113 ± 22	5.7 ± 1.6	7.8 ± 3.2	52.89 ± 37
Pasture	Historic	117 ± 7	5.2 ± 0.2	**6.3 ± 2.1**	48.1 ± 1.4
(n = 3)	Contemporary	128 ± 6	5.8 ± 0.8	**12.3 ± 0.6**	75 ± 18

Nine of the streams had been surveyed for more than eight years, meaning that it was possible to undertake trend analysis to see whether metrics have deteriorated, improved, or remained the same. Trend analysis of the four calculated biotic metrics showed significant changes in only four of the 11 sites examined, and only for a few metrics. Significant increases in the number of EPT taxa were found at one site (the Pahekeheke, draining plantation forestry), and significant increases in the percentage of EPT were found at two sites - the Whirinaki, draining a catchment dominated by native bush, and the Rangitāiki River just above its confluence with the Wheao (Figure 15). Catchment land use at this site was dominated by plantation forestry (approximately 60%), although pasture land use was also a significant proportion of the catchment (20%). The Rangitāiki River at Murupara (part of NIWA's National Water Quality Monitoring Network) also showed a significant increase in calculated QMCI scores over time (Figure 15). Catchment land-use here was also dominated by a mixture of plantation forestry (70%) and pasture (20%).

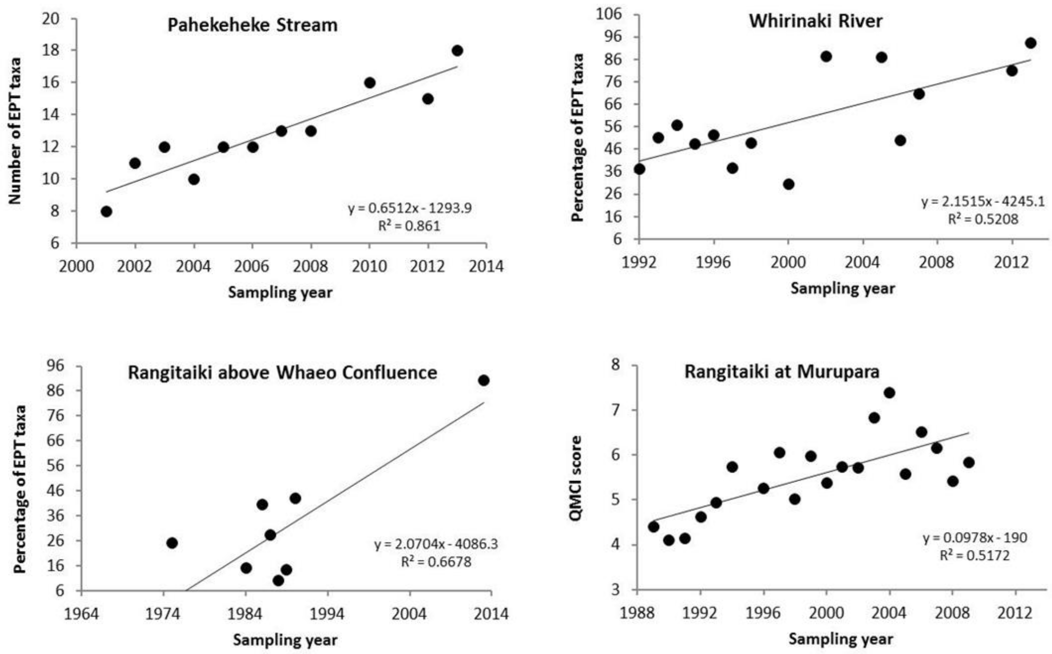


Figure 15 *Metrics observed to show significant trends over time at sites in the Rangitāiki Catchment.*

4.4.4 Quantifying ecological pressure

The greatest difference between MCI_{obs} and MCI_{nat} (about 36 MCI units) was found in samples collected from the TWINSPAN group 8, which were in the lower mainstem of the Rangitāiki below the Matahina Dam (Figure 16). These large differences would reflect, amongst other things, effects of human induced disturbances associated with both land use change, and changes in water quality and flow regime below the Matahina dam. Sites in TWINSPAN groups 3, 4 and 7 differed by c. 25 MCI units, and so are regarded as being the next most impacted (Figure 16). These sites had the greatest number of pasture streams, and had streams with high predicted CLUES N loads. This suggests that the high amount of pasture land use in this area may have been affecting instream health. Sites in TWINSPAN groups 3 and 4 also had the lowest segment flow and predicted low flows, which may also have contributed to their reduction in ecological health from what was expected.

Sites in TWINSPAN groups 1, 5, and 6 all deviated from predicted MCI scores by between 9–13 MCI units, suggesting only a moderate reduction in ecological health. Two of these groups (1 and 5) were in catchments dominated by pine plantations (average cover 75%), and sites in group 6 were also in catchments where pines were very common (42% of catchment area). The lowest difference between MCI_{nat} and MCI_{obs} was from sites in TWINSPAN group 2, where observed scores were generally less than four MCI units lower than predicted scores. These sites were all from catchments dominated by native bush, with a mean catchment cover of 72%.

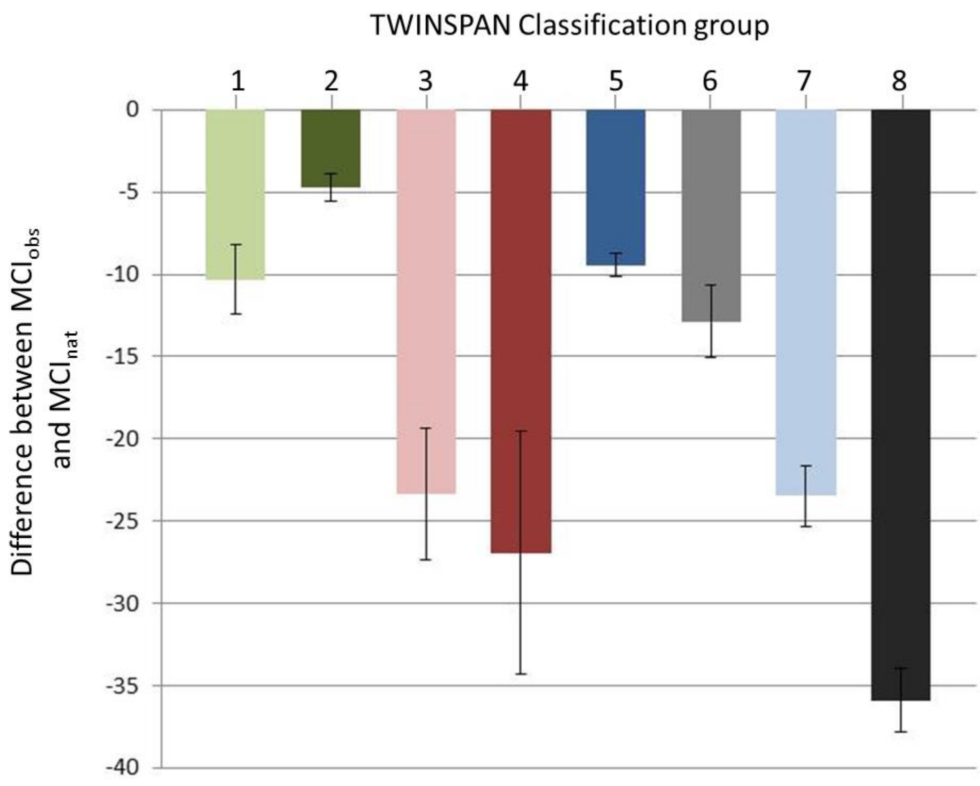


Figure 16 Differences in observed MCI scores (MCI_{obs}) and scores predicted in the absence of land use changes (MCI_{nat}) in the eight groups identified in a TWINSPAN classification of environmental factors. The bigger the difference in scores, the more stream health has deteriorated from its predicted, undisturbed state. Colour coding as per the original TWINSPAN classification given in Section 3.3).

4.5 Discussion

4.5.1 Assessment of current state

This study focused on characterising the ecological health of waterways throughout the Rangitāiki Catchment, based on an assessment of their invertebrate communities. Although many previous studies also examined invertebrate communities throughout the catchment (Bioresearchers Ltd, 1975, 1985, 1990; Boubee et al., 2009; Kingett Mitchell, 2001), most of these focused on larger streams, and in particular streams draining areas of exotic forest. As a consequence, information on the ecological health of many of the small waterways throughout the catchment was generally lacking. This potentially represented a major knowledge gap especially as smaller waterways are in more intimate contact with the surrounding catchment (Hynes, 1975), and as such may be affected more by human activities.

Despite sampling a large number of small waterways in catchments of contrasting land use, results of the current ecological survey showed that many of the streams sampled were in relatively good ecological condition. Sensitive invertebrates such as mayflies and caddisflies were widespread, and were found with high relative abundances. Many of the calculated biotic metrics also scored the streams in either “excellent” or “good” condition. Only a few sites had biotic metrics indicative of “poor” ecological condition.

Studies of the effects of land-use on invertebrate communities throughout New Zealand have generally shown that stream ecological condition is generally highest in native bush, intermediate in forestry streams, and lowest in pasture streams (Hall et al., 2001; Harding and Winterbourn, 1995; Quinn et al., 1997). The finding of so many streams ranked as either good or excellent was thus surprising, especially given the fact that 83 of the streams sampled were from modified catchments dominated by either exotic forest or pasture. Of streams draining pasture dominated catchments, the majority had scores indicative of good or excellent stream condition when assessed by MCI (85% of streams) or QMCI (66% of streams). Similar observations were made for streams draining exotic bush, where 80% and 68% of streams were assessed by either the MCI or QMCI respectively as being in either good or excellent condition.

Pasture streams flowing through productive farmland on the Galatea Plains in particular had good ecological condition. Of the ten streams sampled there, four had MCI scores indicative of “excellent” condition, and six of “good” condition (Figure 17). This suggests that these streams supported many sensitive invertebrate taxa that would normally not be found in degraded streams characteristic of pasture catchments. Seven streams also had QMCI scores indicative of either “excellent” or “good” condition, and two streams were in "fair" condition. Only the Ruarepuae Stream scored in the “Poor” category for the QMCI score. This stream was sampled in two locations: an upper location at the edge of the native bush in pasture, and a lower location on Waitaruna Road. Examination of MCI scores showed that both sites were ranked as “Good”, whereas the QMCI scores showed a reduction in stream condition, from “Fair” at the upper site to “Poor” at the Waitaruna Road site. The reduction in QMCI scores suggests that the relative abundance of taxa sensitive to pressures associated with pasture development - such as reduced water quality and habitat conditions - had declined at the lower site in Ruarepuae Stream. However, these taxa were still present in the stream, meaning that the MCI score (which relies only on presence/absence) had not decreased.

Apart from the Ruarepuae Stream, most of the other streams in the Galatea Plains still supported sensitive taxa with high relative abundances, explaining the relatively good condition when assessed by the MCI or QMCI. The relatively good ecological condition of these streams most likely reflects the fact that many of them flowed from upstream areas of native bush in the Ikawhenua Ranges, which may have conferred a degree of resilience to the streams as they flowed through pasture land cover. Thus having a large proportion of flow from unmodified catchments upstream is likely to buffer these streams from the adverse effects associated with agricultural activities. A similar degree of “buffering capacity“ has been observed by Storey and Cowley (1997) and Harding et al (2006) in small forest fragments in streams draining pastoral catchments.

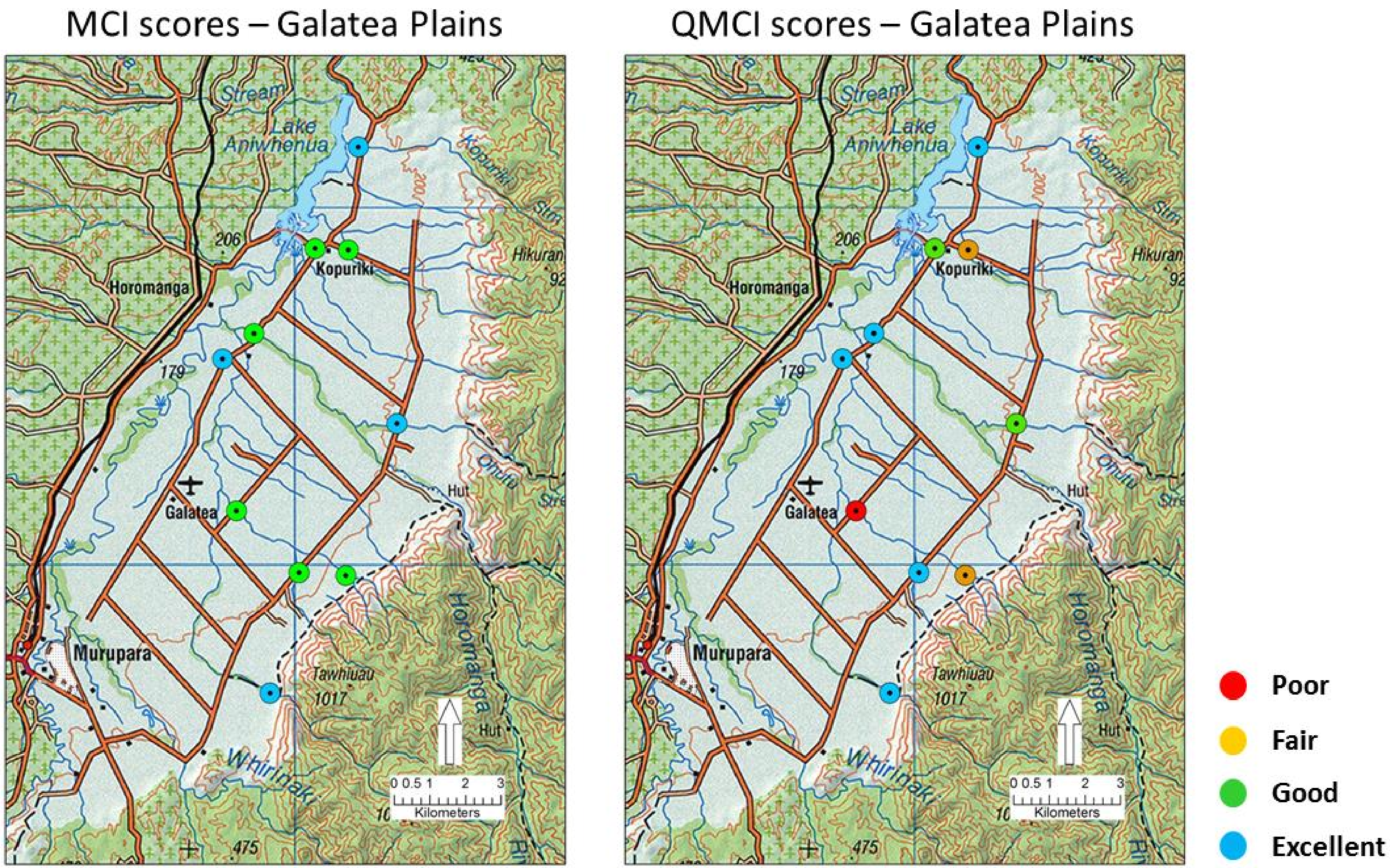


Figure 17 Calculated MCI and QMCI scores of streams flowing through the Galatea Plains.

4.5.2 Interactions with environmental factors

Analysis of the invertebrate data showed the importance of large-scale factors such as catchment land use in controlling invertebrate communities throughout the Rangitāiki Catchment. This result is not surprising, and is consistent with other work in New Zealand. The mechanism behind land-use changes affecting stream health most likely reflects changes to smaller scale factors such as shade, organic matter inputs, flow regimes, and changes to water chemistry (temperature, dissolved oxygen, nutrients). Many of these factors are themselves highly inter-correlated (Figure 18). For example, removing stream shade increases stream temperature (Rutherford et al., 1997), which in turn can lead to increased algal biomass in the stream. Removing trees will also alter the frequency, quantity and timing of organic matter inputs into a stream (Benfield, 1997; Campbell et al., 1992; Scarsbrook et al., 2001). Coarse organic matter such as logs and branches can represent important instream habitat, especially in soft bottomed streams where they can play key roles in structuring not only the physical and hydrological environments (Hilderbrand et al., 1997; Trotter, 1990), but also in providing valuable habitat for both fish and invertebrates (Bilby and Likens, 1980; Collier and Smith, 2003). Finer organic matter such as leaves and twigs become quickly colonised by a variety of fungi and bacteria, which then increases their palatability to a range of invertebrates (Rounick, 1982; Rounick and Winterbourn, 1983; Winterbourn, 1976). Thus converting forest catchments (either native bush or pine plantation) to pasture can have huge effects on invertebrate communities as a result of direct changes to the energetic inputs into streams, as well as changes in habitat conditions.

Removal of trees will also greatly alter flow regimes within streams (Dons, 1987; Fahey and Rowe, 1992). In particular, pasture streams experience increased flood peaks and mean annual catchment yields, as well as increased summer base flows (Duncan and Woods, 2004). This reflects the absence of tall vegetation which intercepts rainfall before it reaches the ground. Instead, most rainfall falling onto pasture catchments will quickly enter the soil water via infiltration, or flow across the surface of the catchment as overland flow and into the stream, leading to higher flood flows. These higher flood flows are likely to cause increased erosion, bank undercutting and collapse - especially in the pumice dominated areas in the uppermost and eastern areas of the Rangitāiki Catchment.

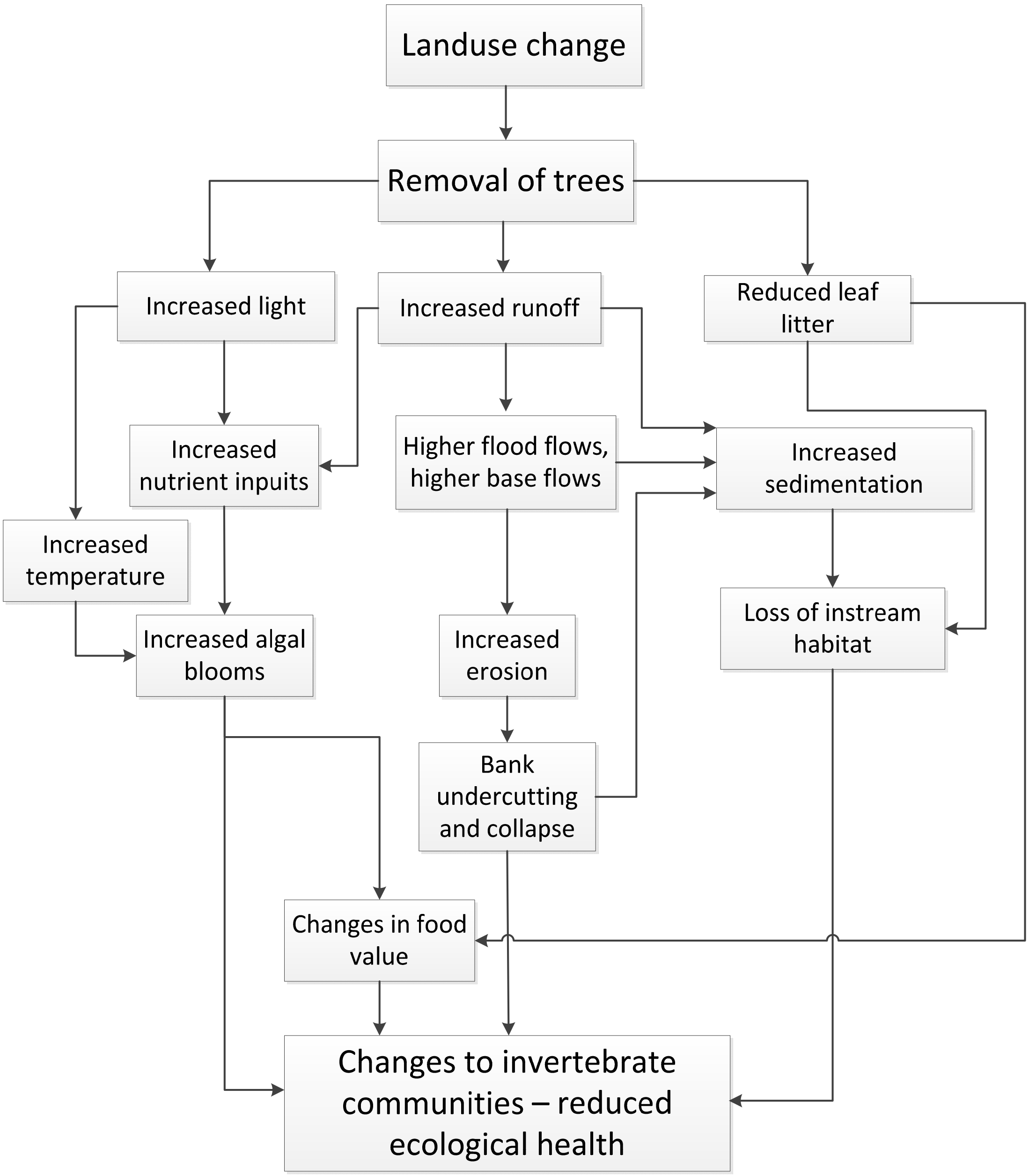


Figure 18 Diagrammatic representation of links between environmental factors that are affected either directly or indirectly by land-use change, in this case removing trees from a catchment. Such an activity would be expected when converting native bush or pine plantation to pasture.

Our analysis also showed that invertebrate communities were regulated by small-scale factors such as stream hydraulics (expressed as percentage of riffles and flow diversity within a site), substrate size and local bank conditions. The effect of stream hydraulics on biota is well-known, with some animals preferring fast-flowing water whilst others preferring slow-flowing water (Jowett et al., 1991; Statzner and Higler, 1986). This may explain why habitats such as riffles and runs often support very different invertebrate communities (Jowett, 1993; Pridmore and Roper, 1985), reflecting in part differences and flow regimes, substrate size, and even food availability. Substrate size was also identified as an important small-scale factor influencing invertebrate communities, and this is not surprising given the number of studies highlighting the importance of this variable (Biggs et al., 2001; Collier et al., 1998; Death, 2000; Hart, 1978).

The analyses also identified estimated CLUES nitrogen load as an important factor influencing stream health throughout the catchment. This is not likely to reflect a direct toxic effect, as the vast majority of sites had predicted nitrogen loads less than 2.4 mg NO_3-N /L, the recommended guideline value for the protection of 95% of species in slightly to moderately disturbed systems for chronic exposure. Furthermore, measured water quality samples exceeded this level at only two of the monitored sites (both of which were in pasture catchments), again emphasising that nitrate levels throughout the catchment were well below potentially toxic levels. However, many pasture streams had both predicted CLUES nitrogen loads and measured spot nitrogen concentrations greater than 0.8 g/m^3 – a level recently suggested above which algal blooms can form which can have adverse effects on stream ecological values (Death, 2013). Thus, as nutrient levels increase, so does the likelihood of algal blooms, which can lead to the loss of sensitive taxa such as mayflies, caddisflies and stoneflies. This may explain the observed negative correlations between clues nitrogen load and both MCI score and the number of EPT taxa.

The results of this analysis clearly show that invertebrate community composition, and by default stream ecological health, is controlled by a range of environmental factors operating at different spatial scales. Due to the highly interlinked nature of many of these factors, it is difficult or potentially impossible to identify a single cause of degradation to stream health. Indeed, the task of identifying a dominant driver of stream health is akin to Lewis Carol's fictitious "Hunting of the Snark", a potentially unfulfilling search for an unobtainable quest. Given the wide variety of streams through the catchment, as well as the wide variety of human activities occurring there, it is highly likely that individual streams will respond in their own manner to these activities. Stream health in one waterway may decline as a result of increased nutrients leading to algal blooms which then displace sensitive taxa, whereas stream health in another waterway may decline as a result of increased bed movement and bank erosion arising from alterations in the flow regime due to land-use changes. What is clear, however, is that this complex interaction of factors does lead to often dramatic changes in stream health throughout the catchment. The challenge is, therefore, to identify this mixture of factors and see what management interventions can be implemented to reduce the adverse effects of land-use change, and increase observed stream health where it is deemed necessary to do so in order to meet management objectives. For example, provision of sufficient riparian vegetation around small waterways flowing through pasture areas has huge potential to help ameliorate any adverse effects associated with loss of overhead shade, increased nutrient inputs and stream temperatures, and potential increased algal blooms. However such management strategies would only succeed in relatively small streams where the interaction between riparian vegetation and aquatic ecosystems are the greatest.

4.5.3 Assessment of temporal changes

Comparison of calculated biotic metrics between contemporary and historic surveys revealed very few differences. For example, no difference in any metrics were observed in eight streams draining pine plantations, and only the number of EPT increased in pasture streams. Temporal trends of the four biotic metrics were examined in 11 individual streams (giving a total of 44 individual tests), and yet significant trends were only observed four times, and only for EPT, percentage EPT and QMCI. Eleven of these temporal comparisons spanned a 30+ year period, and seven comparisons spanned a 12 year period. Lack of any strong changes in invertebrate communities over these relatively long times suggests that a high degree of stability exists among the invertebrate communities.

Such stability has also been reported by Scarsbrook (2002) who assessed invertebrate community composition in 26 river sites monitored by NIWA as part of the National Water Quality Monitoring Network. He found that the invertebrate communities at these sites varied around a relatively steady state of community composition. Similar results have been found by Winterbourn (1997) in a study of invertebrate communities in mountain streams, which are often subject to a high degree of natural hydrological disturbance from unpredictable flooding. Scarsbrook (2002) also found only weak relationships between the magnitude of change in environmental conditions and resultant changes in community composition. He did show, however, that communities changed least when flow conditions over time remained relatively constant, and by corollary, changed the most with large changes in flow regime. This may help explain the lack of changes to invertebrate communities in the lower Rangitāiki River near Edgecumbe that were first sampled only in 1978 (35 years ago). The Matahina Dam was completed in 1967 (47 years ago), and this would have had a large effect on flow regimes below the dam (see Lessard et al 2013 for an example of hydrological alterations), and subsequently on the invertebrate communities. Community composition would have already changed by 1978, which is when the first invertebrate samples were collected. It is likely that the communities would have remained relatively stable after this period after they had adjusted to the new flow regime that they have been exposed to over the past 35 years. This is consistent with the theory of ecological disturbance postulated by Lake (2000) whereby large disturbances such as building a dam are regarded as "ramp disturbances", which often cause the ecosystem to be moved (often dramatically) to a new relatively steady state.

It is interesting to note that Scarsbrook (2002) found that water quality changes had no significant effect on the invertebrate community composition in the 26 sites he examined. One of the main reasons for this may simply reflect the fact that the magnitude of any observed changes in water quality may not have been enough to have caused changes to the invertebrate community. It must be emphasised that most changes in water quality (and in particular nutrients) would manifest themselves predominantly through increased algal biomass (Suren and Riis, 2010), and that this may not have occurred in any of the 26 sites examined in Scarsbrook's study. Lack of strong links between invertebrate communities and changes to a stream's nutrient status may explain why the Rangitāiki River at Murupara showed positive increases in QMCI scores over time, when the long-term water quality monitoring has shown significant increases in the concentrations of NOx-N. If such nutrient increases have not resulted in increases in algal biomass, then there is no reason why metrics such as the QMCI score would decrease.

Finally, it must be remembered that many of the sites examined were located in catchments dominated by pine plantations, and it is conceivable that some of these sites may not have been exposed to forestry operations. As such, these streams would have experienced relatively little in terms of disturbances from factors such as sedimentation, nutrient inputs, high light and temperatures due to lack of overhead canopy. Even if they had, research suggests that forest activities usually represent only a relatively short lived effect on stream communities (Harding et al., 2000; Quinn and Linklater, 1993).

4.5.4 **Quantifying ecological pressure**

Analysis of environmental data showed the existence of eight groups representing gradients in stream size, catchment land use, substrate and flow conditions, bank conditions and riparian vegetation, and predicted CLUES N yield. When considering the wide range of streams sampled, from small well-shaded streams draining native forest or pine plantations, to the large mainstem of the Rangitāiki River flowing through the intensively farmed Rangitāiki Plains, such large gradients are not surprising. The invertebrate communities, and by inference overall stream health, also responded to these gradients, such that stream health was considerably reduced in sites subject to multiple pressures.

That the biggest difference in stream health (assessed as differences between MCI_{obs} and MCI_{nat}) was found in sites in the lower mainstem of the Rangitāiki below the Matahina Dam was not surprising. Sites here were subject to effects of human induced disturbances from land use change, as well as changes in water quality and flow regime below the Matahina Dam. High numbers of filter-feeding insects such as *Aoteapsyche* were found at sites below the Matahina Dam, where they contributed 7–10% of total density. These high numbers are in contrast to other areas throughout the catchment where *Aoteapsyche* contributed on average 1% or less to total density. The high numbers of these filter-feeding animals below the dam most likely reflected the large amounts of phytoplankton in the river below Lake Matahina – indeed our water sampling showed that chlorophyll levels were c. 25 times higher in Lake Matahina than in Lake Aniwaniwa. The number of medium and large magnitude floods would also be greatly reduced below the dam, further increasing the habitat suitability for filter-feeding insects such as *Aoteapsyche*. These animals also have relatively low MCI scores (4), so their presence at sites below the dam is likely to have reduced the MCI scores here. Similar observations of large numbers of filter feeding insect taxa below dams were made by Harding (1992) who examined invertebrate communities of streams on the West Coast of the South Island, and reflects both increased food supply and increased flow stability at these sites.

Changes in flow regimes below dams have also been observed to reduce both the QMCI and percentage of EPT in the Opuha River in South Canterbury (Lessard et al., 2012). These changes were thought to reflect the reduced frequency and magnitude of high flow events, and subsequent increased frequency of algal blooms and bed armouring. The streambed of the Rangitāiki River below the dam is dominated by pumice sands and gravels, so bed armouring is highly unlikely, but the river is characterised by large amounts of aquatic macrophytes (eg., *Egeria* and *Ceratophyllum*) growing in places. It is highly likely that the modified flow regime below the dam has increased the habitat suitability for aquatic plants. This in turn would have had flow-on effects to the invertebrate community, altering it to a community more consistent with slow-flowing degraded conditions (Collier, 2004).

The next most impacted streams were in sites dominated by pasture streams, which had high predicted CLUES N loads. The finding of reduced stream health in pasture streams is unsurprising, and reflects well known observations of reduced ecological health in pasture streams. Sites in TWINSPAN Groups 3 and 4 also had the lowest predicted flow and low flow. These low flows may also have affected the invertebrate communities, either as a direct result of habitat preferences, or from indirect effects associated with low flows such as increased temperature during summer, or increased sedimentation and algae as a result of lack of flushing flows (Dewson et al., 2007a, b).

In contrast to the pasture streams, sites in TWINSPAN groups 1, 5, and 6 had a relatively small reduction in ecological health, with MCI_{obs} deviating from MCI_{nat} by 9–13 units. Streams in these groups drained pine plantations, and observations by Harding et al. (2000) and Harding and Winterbourn (1995) showed that stream health is generally only slightly impaired such streams. This reflects the fact that pine streams are generally well-shaded and protected by riparian vegetation, and generally have low nutrient inputs. However, subtle changes exist in the palatability of leaf litter entering these streams, and some of the shredding insect taxa that are found in native bush streams are absent from streams draining plantations (Harding and Winterbourn, 1995). Moreover, water chemistry is often different in pine plantations, and there are also short-term effects of forestry harvesting operations; both of which can lower stream health.

The lowest difference between MCI_{nat} and MCI_{obs} was from sites in the TWINSPAN group 2, which were from catchments dominated by native bush. Stark and Maxted (2007b) suggest that any ecological differences represented by <10 MCI units are in fact minimal, so most of the samples from these streams could be considered to be in "reference condition". Stream health at these sites would thus be representative of what could be expected in the absence of human disturbance.

Part 5: Water quality

5.1 Introduction

Water quality refers to the chemical and physical characteristics of water, and is a useful measure of the condition of water relative to the requirements of maintaining healthy ecosystems, or meeting human needs in terms of human contact and drinking water. New Zealand has a diverse range of aquatic environments from mountain springs to coastal estuaries, connected by an intricate network of rivers, lakes, wetlands, estuaries and groundwater systems. Such diversity means that natural water bodies vary in both their natural water quality conditions, as well as their response to human disturbances associated with land-use change and other activities. For example, Close and Davies-Colley (1990) found that base flow water chemistry of large rivers throughout New Zealand was influenced by a combination of flow factors, geology and land-use.

By world standards, our fresh water is generally of good quality (Close and Davies-Colley, 1990) and our rivers, lakes and wetlands support a unique array of flora and fauna which are highly regarded for their recreational value. Clean fresh water is also extremely important to Maori, and water is essential for its power to provide life. The Treaty of Waitangi (Te Tiriti o Waitangi) is also the underlying foundation of the Crown–iwi/hapū relationships with regard to freshwater resources, and so addressing tāngata whenua concerns over water quality issues are key to meeting obligations under the Treaty.

Despite its importance to all New Zealanders, water quality in urban and rural areas is degraded, and is coming under increasing pressure as land use intensifies. There is clear recent evidence of the adverse effects of land use intensification associated with farming and urbanisation (Larned et al., 2004; Wilcock et al., 1999; Wright, 2013), and this has implications for aquatic life, drinking water supplies, cultural values and water-based recreation. A large proportion of water pollution comes from diffuse sources, and these sources are especially prevalent in grazed livestock pasture, which occupies approximately 40% of New Zealand's land area. Furthermore, diffuse pollution from pastoral agriculture increases with land use intensification. In contrast, water quality in streams draining exotic plantation forests is generally better than that draining pasture streams, and can often approach the quality of rivers in native vegetation cover. Although periodic harvest operations can mobilise fine sediments, this disturbance is a relatively short lived, and water quality soon returns to preharvest levels.

The objective of this study was to characterise water quality conditions of the smaller tributaries streams draining different land uses in the Rangitāiki Catchment, and of Lakes Matahina and Aniwaniwa. Although NIWA runs long term water quality monitoring sites in the Whirinaki, Murupara and Te Teko, few water quality surveys have been conducted in many of the small rivers throughout the catchment. This was an important information gap, as smaller streams are in more intimate contact with the surrounding landscape than the larger rivers and are thus more likely to be affected by land use change. Moreover, small streams ultimately flow into the larger rivers and so play an important role in affecting their water chemistry as well. The focus of the current study was, therefore, to better characterise water quality conditions of smaller streams in the catchment.

This was also done in recognition of the importance to improve water quality in the Rangitāiki Catchment, as highlighted in the Draft Rangitāiki Discussion Document. Furthermore, recently released central government policy documents such as the National Policy Statement for Freshwater Management (NPS-FW 2014) have highlighted the need for councils to manage water quality to maintain the current status of ecological health and human health for recreation, and not let these degrade. This means that councils need to develop sustainable catchment load limits for nutrients, sediments and bacteria through implementation of the NPS-FW.

5.2 Methods

5.2.1 Field methods

As part of the invertebrate survey, spot measurements were made of conductivity and water samples collected from each site. Water samples were kept chilled and returned to the laboratory for analysis of nutrients (nitrogen and phosphorous) and bacterial contamination (*E. coli*). Invertebrate sampling in the Rangitāiki mainstem was not done until January/February 2014, so water samples for assessments of nutrients and bacterial contamination were not collected during this latter survey, although measurements of conductivity were still made.

Following the initial invertebrate sampling, a subset of 28 sites was selected from the original 95 sites for regular water quality monitoring for between four and six months. This relatively short period of time purely reflected logistical restrictions, and is acknowledged not to be long enough to draw robust conclusions about water quality. However, the sampling done was thought sufficient to characterise differences in water quality parameters in streams flowing through different land uses. Sites were randomly selected within each of the major REC classes that were surveyed, so that the number of water quality sites was in the same proportion of REC classes as the invertebrate sampling sites. Because water quality is closely associated with stream flow, a flow gauging was conducted at most sites where possible on the date of sampling. In this way, catchment yields from each stream could be calculated.

Monthly water quality samples were also collected from lakes Matahina and Aniwaniwa, and analysed for total nitrogen, phosphorous, chlorophyll and Secchi depth. These factors were used to calculate the Trophic Lake Index (TLI). The TLI is a useful measure of the overall trophic state of a lake, which is likely to reflect a combination of many other factors such as water quality conditions of the in-flowing rivers, lake size and shape, the amount of plant growth in a lake, lake stratification and lake residence time (Burns et al., 2000). Results of this monitoring were compared to long-term TLI values obtained from the Rotorua lakes. Although it is acknowledged that Lakes Matahina and Aniwaniwa are artificial lakes behind dams, and that the other Rotorua Lakes are natural, this comparison was made purely to help put the trophic state of these two artificial lakes into context with other lakes in the region.

5.2.2 Statistical analysis

All water quality data was investigated for normality and transformed where necessary. Most of the data was highly skewed to the left, meaning there were many observations where particular factors had low values and few observations where the same factors had high values. The fourth root transformation appeared to be the most successful way to normalise the data. All water quality data collected as part of the one-off large-scale survey was analysed by ANOVA to see how concentrations of the measured water quality parameters and *E. coli* counts differed between streams draining different land uses. We assessed the effects of land use on stream water quality as this has been shown to be a major factor influencing water chemistry (Larned et al., 2004; Wilcock, 1986).

Following the assessment of the broadscale water quality data, data collected from the subset of sites where monthly sampling had been conducted were used to calculate catchment yield (kilograms per hectare per year). All concentration data for nutrients (g/m^3) and for *E. coli* (colony-forming units (cfu) per 100 ml) was multiplied by flow data (m^3/s) to get catchment loads. This load data was then divided by the area of each catchment in hectares and multiplied by the number of seconds per year. The resultant catchment yield data was also fourth root transformed, and analysed by ANOVA to see whether it differed between catchments draining different land uses.

Monthly water quality data from lakes Matahina and Aniwaniwa consisted of total nitrogen, phosphorous, chlorophyll and Secchi depth. Values for individual parameters were compared between the two lakes to see whether they differed. The TLI was also assessed for each lake based on the average of all the water quality parameters. The overall water chemistry signatures from each lake were compared to those from the Rotorua Lakes using a Principal Components Analysis.

5.3 Results

5.3.1 Overall catchment conditions

Significant differences in water quality from streams draining different land uses were observed for four of the six water quality factors examined. *E. coli* counts and concentrations of ammonia-N were significantly higher in streams draining pasture land cover than pine or native bush (Figure 19). Total oxides of nitrogen (TOx-N – hereafter referred to as inorganic-N) and spot water temperature were highest in streams draining pasture and pine plantation, and lowest in streams draining native bush. Of interest was the finding that the average concentration of inorganic-N in pasture streams was higher than 0.8 g/m^3 (Figure 19). This is a level regarded as having significant adverse effects on invertebrate communities as a result of increased algal biomass (Death, 2013).

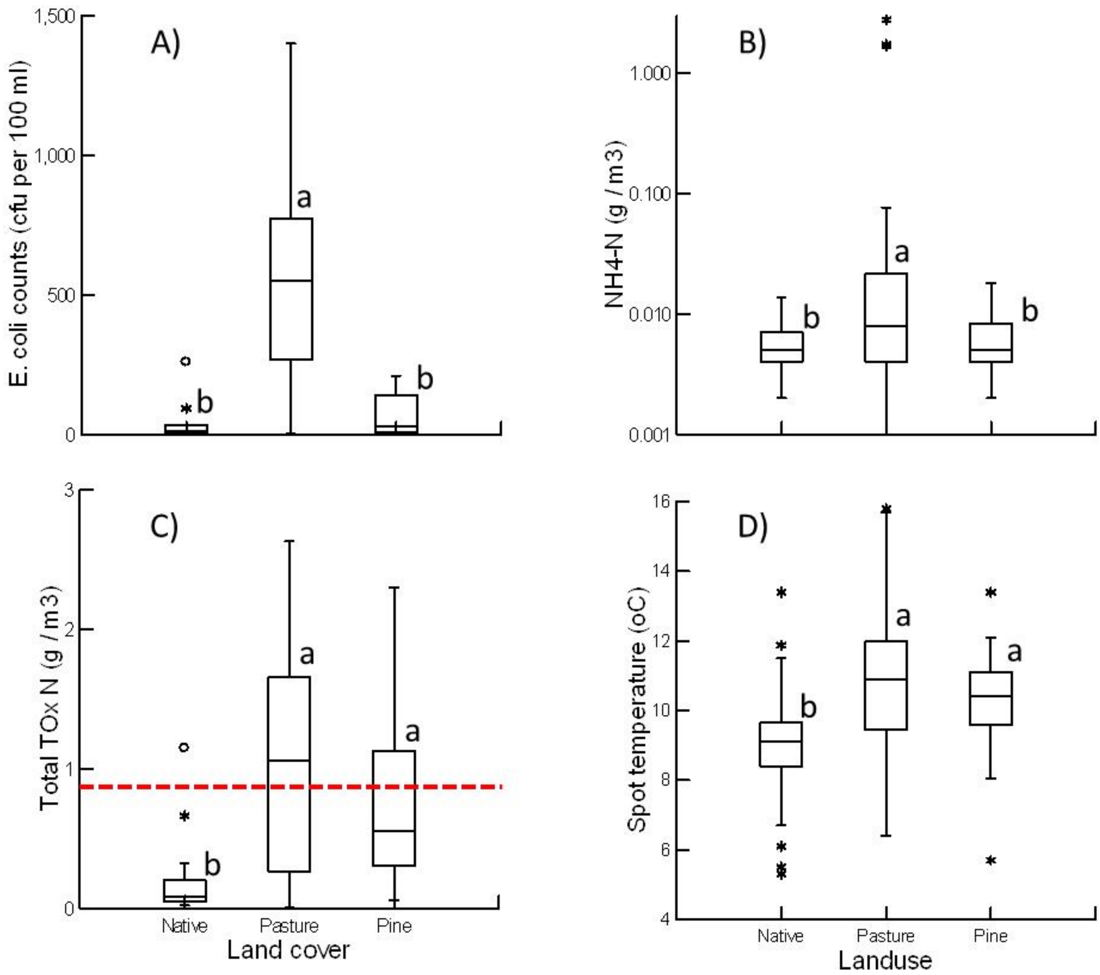


Figure 19 Box plots of A) measured E. coli counts; concentrations of B) ammonia-N and C) total inorganic nitrogen (TOx-N), and D) spot water temperature measured from streams throughout the Rangitāiki Catchment sampled as part of the invertebrate survey work from June 2013 to February 2014. Similar letters above each box plot show means that are statistically similar. The dashed red line in C) indicates the 0.8 g/m^3 inorganic nitrogen concentration above which can cause increased algal blooms (Death 2013).

5.3.2 Catchment yields

Of the 28 sites randomly selected for ongoing water quality monitoring, two dried out after only two sampling visits, and so were excluded from the analysis. Of the remaining 26 sites, 15 showed significant reductions in streamflow during the summer period. There were no significant differences in mean flow or catchment area between the three land cover types. As with the broadscale survey, concentrations of ammonia-N, inorganic-N, and *E. coli* counts were significantly higher in pasture streams, and lowest in streams draining pine plantations and native bush. However, when this concentration data was converted into catchment yield, only calculated inorganic- N yields differed between land use classes, and was higher in catchments dominated by pasture and pine (Figure 20).

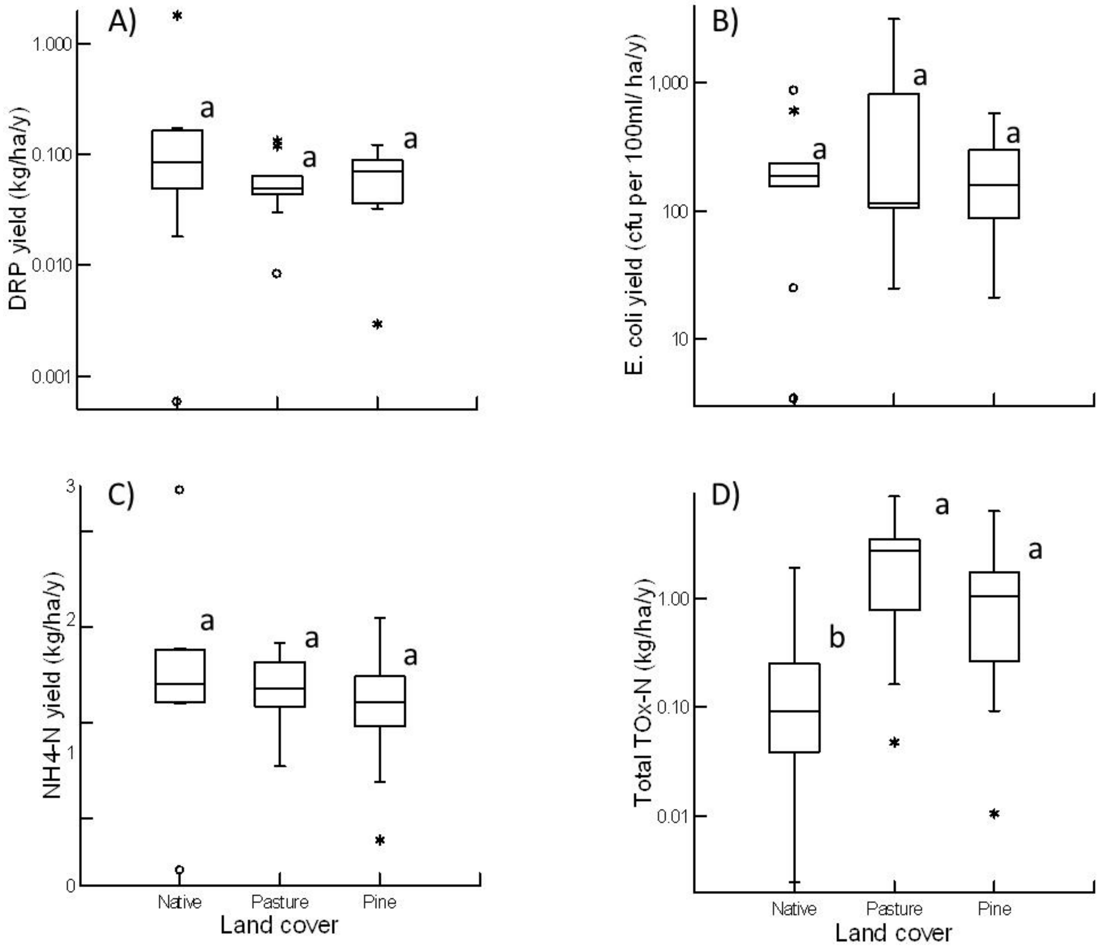


Figure 20 *Box plots of estimated catchment yields of A) E. coli counts, and concentrations of B) dissolved reactive phosphorus (DRP), C) ammonia-N and D) total inorganic nitrogen (TOx-N), calculated from monthly samples collected from 26 streams throughout the Rangitāiki Catchment. Similar letters above each box plot show means that are statistically similar.*

Regression analysis showed significant relationship between the percentage of either native bush or pasture in a catchment and the calculated catchment yield of inorganic- N, with higher catchment yields in catchments with increasing percentage cover of pasture (Figure 21).

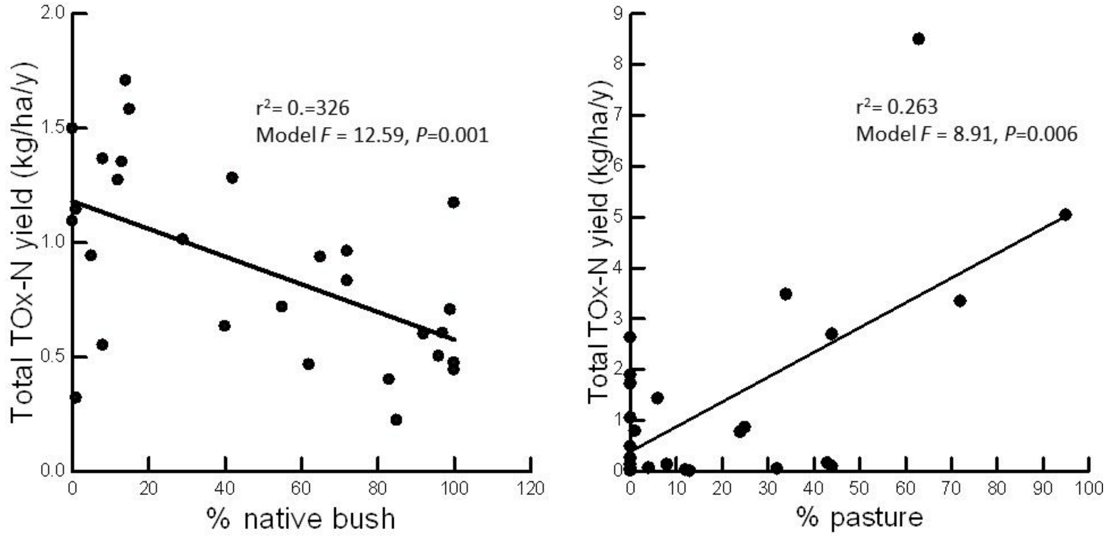


Figure 21 *Relationships between percentage of either native bush (left) or pasture (right) in a catchment and calculated catchment yield of Inorganic-N. Note that the catchment yield data has been fourth-root transformed to ensure normality.*

5.3.3 Lake water quality data

Lake water sampling was conducted for five months, from December 2013 until April 2014. Calculated TLI values during this time were all very high, with Lake Aniwaniwa having an average TLI of 4.41, and Lake Matahina an average of 5.61. This would place these lakes into the eutrophic and supertrophic categories respectively. Examination of water quality parameters collected during the five-month period showed that Lake Aniwaniwa had significantly higher levels of TN than Lake Matahina, and significantly lower levels of chlorophyll, and higher clarity (Table 14). Average total phosphorus was higher in Lake Matahina, but was extremely variable, ranging from a low of 36 mg/L in March 2014, to 350 mg/L in February.

Table 14 *Average WQ factors collected from Lakes Aniwaniwa and Matahina during a five month period from December 2013 to April 2014. Ns = no significant difference.*

Variable	Aniwaniwa	Matahina	t-test	p-value
TN (mg/l)	491 ± 73	342 ± 91	2.85	0.02
TP(mg/l)	44.4 ± 2.6	120 ± 132	4.0	Ns
Chlorophyll *a* (mg/l)	0.98 ± 0.6	24.9 ± 21	4.0	0.03
Sechi depth (m)	4.55 ± 0.9	1.8 ± 1.2	3.71	0.006

A PCA of water chemistry data showed that both lakes had similar water quality conditions to the other lakes in the Rotorua region (Figure 22). Lake Matahina was characterised by low axis one scores, reflecting its high chlorophyll, low clarity and high TN and TP loads. However, this lake had higher axis 1 scores than Lake Okaro, which was at the extreme end of this environmental gradient. Lake Aniwaniwa was positioned in a similar grouping as Lakes Rotoehu and Rotorua.

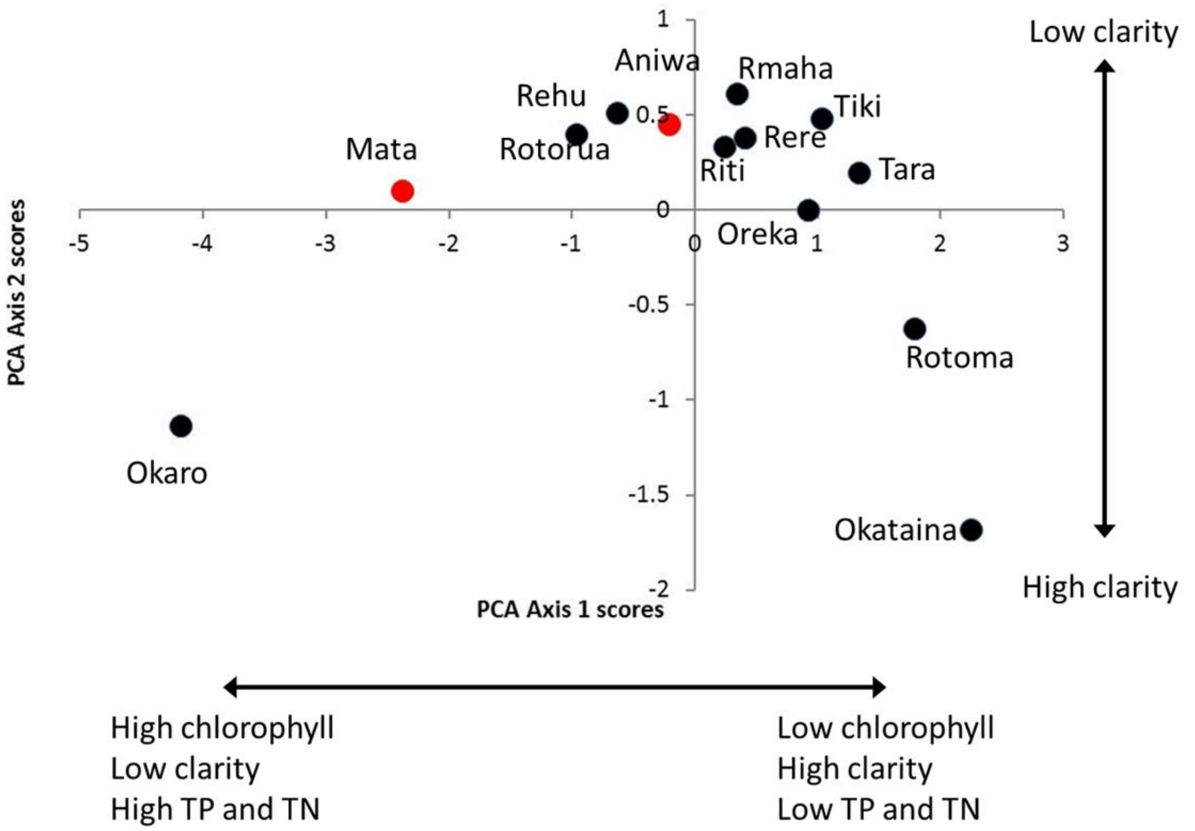


Figure 22 Results of a PCA on average water quality data from Lake Matahina and Aniwaniwa (red), and the lakes in the Rotorua region. Also shown are the water quality factors that were highly correlated with axis 1 and 2 scores. Lake abbreviations as follows: Mata = Matahina; Aniwa = Aniwaniwa; Rehu = Rotoehu; Riti = Rotoiti; Rmaha = Rotomahana; Rere = Rerewhakaaitu; Tiki = Tilitapu; Tara = Tarawera.

5.4 Discussion

5.4.1 River water quality

Results of the spot water chemistry monitoring showed that pasture-dominated streams had elevated levels of bacterial (*E. coli*) contamination and elevated concentrations of ammonia-N and inorganic -N. This is similar to previous studies investigating the effects of land use on water quality. For example, Larnard et al (2004) investigated state and trends in water quality in low- elevation rivers (<400 m elevation) across New Zealand, and found that *E. coli* and dissolved N and P concentrations in pasture streams were 2–7 times higher than in streams draining native and pine plantations. They also found that overall water quality in pastoral streams was not statistically different from that of urban streams sampled, whereas water quality from streams draining pine plantations was not statistically different from streams draining native forest.

The average inorganic-N concentrations in pasture streams was higher than 0.8 g/m^3 which is the level above which is regarded as having significant adverse effects on invertebrate communities (Death 2014). This level has been suggested as an upper limit of Inorganic-N to prevent excessive algal blooms from developing, which in theory can adversely affect both ecological and recreational values of rivers (Biggs 2000). As nutrient inputs into streams increase, so does the likelihood of algal blooms, which can lead to the loss of taxa such as mayflies, caddisflies and stoneflies, with a concomitant reduction in MCI scores. Unexpectedly, despite the relatively high levels of Inorganic-N, the overall ecological condition of waterways throughout the catchment was still relatively high, even in pasture streams.

Relationships between nutrient enrichment and resultant algal biomass accumulations are not always clear cut. For example, Wilcock et al (2007) suggested that elevated nutrients in soft-bottomed streams do not result in algal blooms, as algae are usually not able to grow on soft-substrates. These results suggest that soft-bottomed streams are more resistant to the negative effects of nutrient enrichment than gravel-bed streams. Examination of the habitat data showed that many of the streams surveyed had relatively fine substrates, dominated by medium-sized and fine gravel, and sands, silt and mud (Figure 23). This small substrate size reflects the often pumice-dominated nature of the streambed, which is continually slowly moving downstream in the water current. Such habitat conditions are not conducive to algal growth, irrespective of the prevailing Inorganic-N levels.

Figure 23 Examples of the fine, often pumice-dominated streambeds found in the survey that included a mix of small streams and large rivers, and waterways that flowed through a variety of land uses. Photos show small streams flowing through pine forest near Lake Matahina (RES_001) Lake Aniwaniwa (RES_067), Murupara (RES_091), or Otamatea (RES_033), streams flowing through pasture at Lochinver Station (RES_042), the mainstem of the Rangitāiki River above Lake Aniwaniwa (RES_124). Note that while many streams were devoid of plant growth, some supported high macrophyte cover (RES_033), or isolated communities of aquatic mosses on stable cobbles within the otherwise sandy stream (RES_042). The location of each of these sites is shown in Appendix 1 (in blue).

Similar observations of the controlling influence of substrate size were noted in the Boubee et al (2009) report. Here, substrate size was suggested to have constrained algal development in eight of the 12 sites surveyed in the Rangitāiki and Wheao Rivers, as these sites were dominated by fine pumice sands or mud. Analysis of the measured substrate data collected during the field work showed no significant differences in substrate size in any of the locations surveyed, suggesting that a high degree of spatial variability exists throughout the catchment in terms of substrate size. Boubee et al suggested that there are likely to be many areas within the Rangitāiki River where algal biomass was controlled by the inherently highly mobile pumice bed, and other areas where boulders and large and small cobbles occurred, and where algal blooms may have developed. Indeed, during the contemporary survey of the mainstem Rangitāiki in February 2014, some shallow fast flowing riffle sites dominated by cobbles were observed to have high cover of algae and cyanobacteria such as *Phormidium* (Figure 24).

Links between *Phormidium* cover and nutrients are not always clear-cut (Heath et al., 2012), so high nutrients do not always cause high algal biomass. However, *Phormidium* is N-limited in many rivers (Heath et al., 2012), and can reach high cover during times of stable flow and warm temperatures (Milne and Watts, 2007). Such conditions were evident in the January/February study. The observed increased in N in the Rangitāiki below Murupara (Boubee et al 2009) may also be responsible for these *Phormidium* blooms, which could in theory become more noticeable with any further increases in N, particularly during summer periods of low flow and warmer temperatures.

This wide range of substrate size conditions throughout the catchment poses considerable challenges for assessing the ecological effects of increased nutrients on waterways. This is because excess nutrients will only cause algal blooms where stable substrates exist and so excess nutrients are not considered a major problem in soft-bottomed streams (Wilcock et al., 2007). There is likely to be a highly spatially variable response of algae to nutrients, so setting any limits on an acceptable algal biomass in the river will need to consider this spatial variability. Some of this spatial variability can be modelled using the REC, where sediment particle size has been modelled for every NZReach. This information could be used to help select sites dominated by large substrates where algal blooms are predicted to occur.

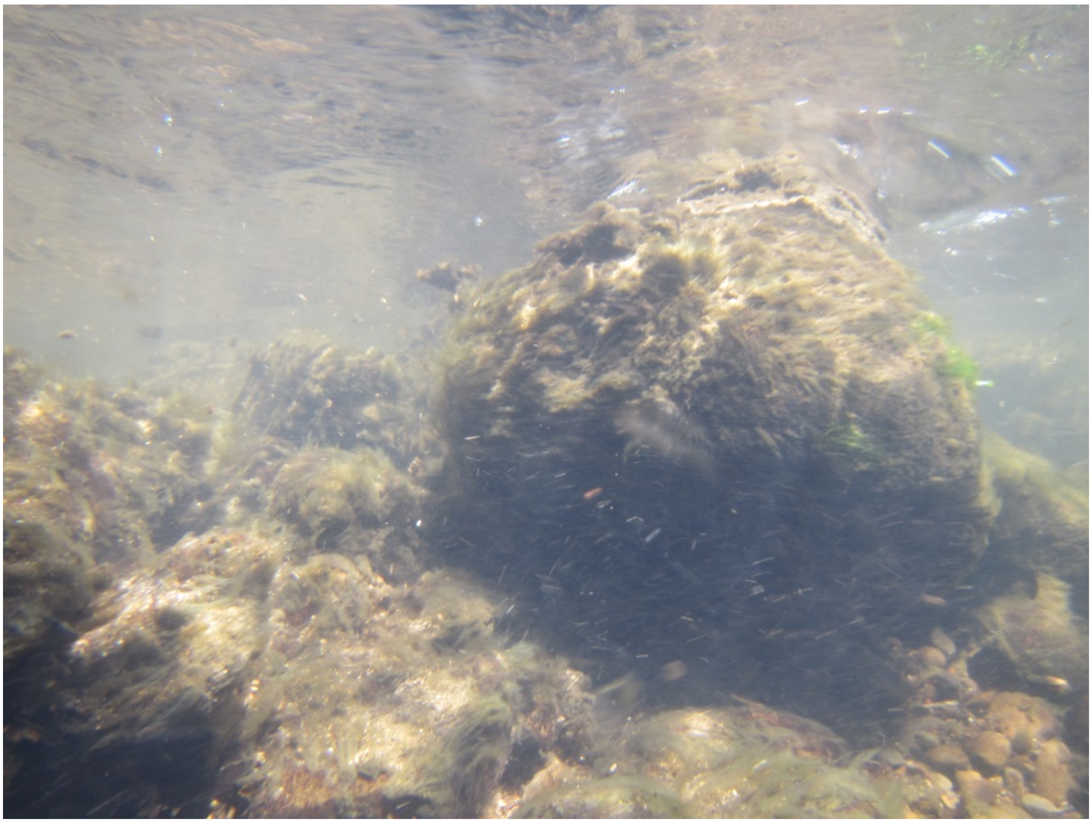


Figure 24 *Example of high cover of cyanobacteria (above) and a mix of filamentous green algae and diatoms (below) growing in areas where large substrates such as boulders and cobbles dominated the streambed in the Rangitāiki River at site RES_103, 3.5 km downstream from the Aniwhenua Dam. The location of this is shown in Appendix 1 (in yellow).*

Boubee et al (2009) identified high water velocities (over either bedrock or boulder areas) as another feature limiting algal growth in the Rangitāiki, constraining algal biomass in four of the 12 sites they sampled. High water velocity is likely to be a factor mostly in large, deep rivers, as shear stress at even relatively high velocities in shallow rivers is considerably less than in deep rivers (Carling, 1992; Downes et al., 1997; Statzner et al., 1991). As such, algae are more likely to be scoured from boulders in deep rivers than in shallow streams. Thus setting nutrient regimes to control algal biomass in the large mainstem of the Rangitāiki may need to consider the spatial extent of areas of high water velocity. This can be modelled as a function of slope in large rivers.

Although light is also a major controlling variable for algal cover (Biggs 2000), Boubee et al suggested it played only a relatively minor role in controlling algal biomass at sites in their survey, with shade in only two of the 12 sites implicated as reducing biomass. Their result may reflect the fact that many of the sites in their survey were from the mainstem of the Rangitāiki where shading is unlikely to be an issue. Riparian shading does, however, play an important role in limiting algal biomass in streams less than c. 8m wide (Quinn et al., 2001) and so it is likely that algal biomass would be low in streams flowing through either native bush or pine plantations. This contention was supported by casual observations made during the current survey, where algal biomass in forested streams was noted as being much less than in streams without shade. The importance of riparian shade should therefore not be underestimated as a potential management tool in cobble bed streams to ensure that high algal biomass does not develop.

5.4.2 Lake water quality

The Rangitāiki River flows into Lakes Matahina and Aniwaniwa, and so close relationships would be expected between water quality of the river and that of the lake. Preliminary results of the limited water sampling in the lakes showed that these two lakes were nutrient enriched, with Lake Aniwaniwa being eutrophic, and Lake Matahina supertrophic. Water chemistry of each lake was very different. Lake Aniwaniwa had higher total nitrogen and higher clarity and low phytoplankton biomass. This lake was largely macrophyte dominated. In contrast, Lake Matahina had much lower total nitrogen, and lower clarity reflecting the phytoplankton dominated nature of this lake. Water quality in rivers generally decreases in a downstream manner, (Harding et al., 1999), so the observed reduction in nitrogen between the Lake Aniwaniwa and Lake Matahina is surprising, especially given the fact that Lake Matahina is further down the catchment, and drains more pasture than Lake Aniwaniwa. The most likely cause for the reduction is the biological uptake of nitrogen by plants, and it is suggested that this may be happening in Lake Aniwaniwa as a result of the luxurious macrophyte growths in this lake.

The different water chemistry signals of each lake may also reflect the large differences in lake bathymetry. Thus, the wide and relatively shallow Lake Aniwaniwa represents an ideal habitat for aquatic macrophytes. This is in contrast to the deeper more incised nature of Lake Matahina, which would preclude the establishment of these plants. Furthermore, the lower clarity in Lake Matahina would also prevent macrophytes from establishing to any great extent here. This may explain why this lake appears to be phytoplankton dominated.

Part 6: Lake macrophytes

6.1 Introduction

Construction of the Matahina and Aniwaniwa Dams in the Rangitāiki River created two large lakes in the catchment. Lake Aniwaniwa is a relatively shallow lake that occupies surface area of 2.1 km^2. Lake Matahina in contrast is much deeper, and slightly larger (area = 2.5 km^2). Both lakes support growths of aquatic macrophytes - large rooted plants that grow mainly in the shallow areas around lakes. Aquatic macrophyte distributions within lakes are controlled by a mixture of growth and loss processes. Growth processes are influenced by factors such as light and nutrients, while loss processors are influenced by factors such as wave disturbance, and grazing by animals (de Winton and Schwarz 2004). Lake depth has a major effect on many of these processes, so macrophytes consequently show strong relationships to depth.

Aquatic macrophytes play many beneficial roles in lakes. They represent a major source of primary production, and also represent important habitats for both fish and invetebrates (de Winton and Schwarz, 2004). Many exotic macrophytes have been introduced to New Zealand and these have had major impacts on lake ecosystems throughout the country (de Winton and Schwarz 2004). Introduced macrophytes have spread throughout the country, and there are now only a few lakes left that contain only native vegetation. Introduced macrophytes out-compete native species which are generally of low stature, less competitive, and easily excluded from their shallow habitats. These changes have been monitored in the Te Arawa/Rotorua Lakes over time (Edwards and Clayton, 2012), where invasive plants such as *Lagarosiphon*, *Elodea* and *Ceratophyllum* have colonised many of the lakes and outcompeted native plants. Increased growth and dominance of exotic macrophytes has major effects on lake processes, resulting in a loss of native vegetation and displacement of native seed banks. Light penetration into the water column is also reduced, resulting in a buildup of organically enriched sediments which can lower oxygen levels (Champion et al., 2002). Excessive macrophyte growth can also restrict recreational activities such as fishing, boating and waterskiing, and may reduce efficiency of hydroelectric power generation by blocking intake structures (Champion et al., 2002; Closs et al., 2004).

de Winton and Schwarz (2004) suggested a model showing how the benefits of aquatic macrophytes can vary greatly according to their particular biomass (Figure 25). At low biomass levels, aquatic plants risk being lost from lakes due to lack of wave damping ability and sediment stabilisation. Maintenance of aquatic macrophytes at such low biomass may be unsustainable unless suitable restoration activity is implemented. At high biomass levels (as would be expected in lakes dominated by exotic macrophytes), problems occur through to loss of biodiversity and natural character, and control measures are recommended.

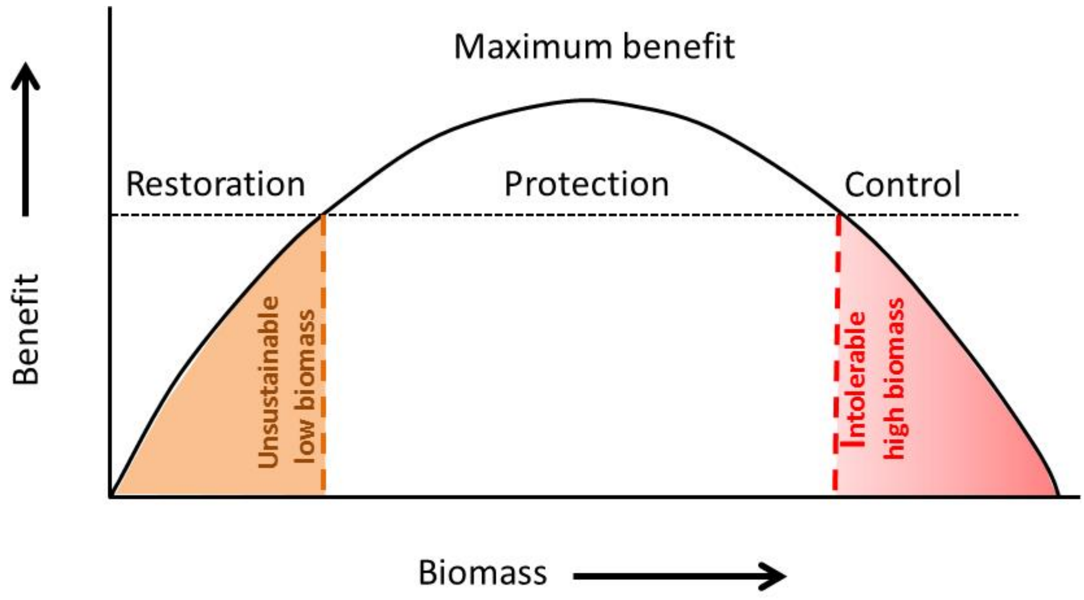


Figure 25 Conceptual model of relationships between macrophyte biomass and the benefits they confer on values such as ecology and recreation. Note how different management actions and responses differ across the gradient of plant biomass. (Modified from de Winton and Schwarz 2004).

Because macrophytes are such a dominant part of Lake Aniwaniwa, it was decided to undertake a survey of the plant communities in both Lake Aniwaniwa and Lake Matahina to better characterise them. As part of the consent applications for Lake Aniwaniwa, previous aquatic macrophytes surveys have been done in the 1980s, so it was possible to examine changes in macrophyte community composition and cover overtime in this lake.

6.2 Methods

NIWA has developed the LakeSPI (Submerged Plant Indicators) methodology for assessing ecosystem health in lakes using aquatic macrophytes. Measures of LakeSPI compliment traditional water quality monitoring such as the TLI by providing ecological information about lake health in terms of macrophyte communities, whereas the TLI is focused primarily on water quality parameters. Moreover, LakeSPI focuses on the littoral edges of lakes where arguably both human interaction and ecological values are greatest (Clayton and Edwards, 2006). LakeSPI uses submerged plants which are rooted to the bed of lakes, and which integrate the range of environmental conditions supporting plant growth over an extended period of time. Pressures faced by lakes include increased sediment and nutrient loading and the displacement of native vegetation by exotic, or invasive plant species. LakeSPI provides an effective means to assess these impacts. There are three components of LakeSPI:

- Native condition index, which captures the native character of the vegetation;
- Invasive impact index, which captures the invasive character of vegetation in the lake based on the degree of impact by invasive weeds;
- LakeSPI index, a synthesis of components from both the native condition and invasive impact condition. The higher the score, the better the condition.

LakeSPI surveys have been conducted in the 12 lakes of the Rotorua region (Edwards and Clayton, 2012). These surveys have shown the long-term negative effects of changing water quality and clarity to the macrophyte communities in some of the lakes. They have also highlighted the negative effect of invasive plant species on overall lake condition. As part of the routine LakeSPI survey conducted by NIWA in 2014, one-off LakeSPI surveys were also conducted on Lakes Aniwaniwa and Matahina to characterise their macrophyte communities, and put the results from these two artificial lakes into context with other monitoring results from the natural Te Arawa/Rotorua Lakes.

Within each lake, NIWA staff assessed aquatic plant communities at six transects (Figure 26), chosen in areas where the natural lake bathymetry was conducive to the establishment of macrophytes. At each transect, profiles were made from the lakes edge to the deepest point of plant cover, and records of species encountered and their depth distributions noted. The maximum and average height of different plant species were also recorded, as was the maximum and average % cover of macrophyte species along each transect. Percent cover was divided into six categories:

- 1 = 1-5%
- 2 = 6 – 25%
- 3 = 26 – 50%
- 4 = 51-75%
- 5 = 76-95%
- 6 = 96-100%

All information on the species present, the cover and depth range was subsequently used to generate LakeSPI scores.

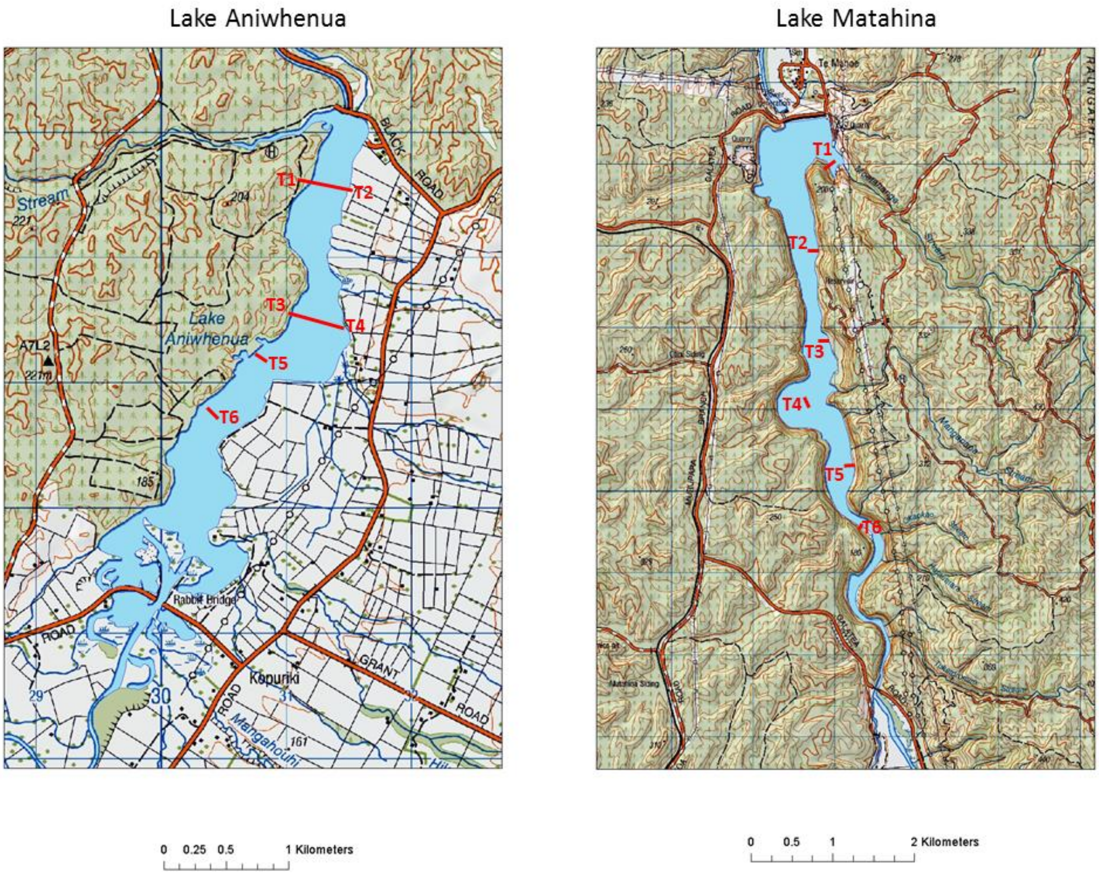


Figure 26 *Location of the LakeSPI transects in Lake Aniwaniwa and Lake Matahina as surveyed by NIWA Hamilton in March 2014.*

6.3 Results

A total of 12 species of aquatic macrophytes were recorded in the two lakes, of which only six were native species (Table 15). Five of the six transects in Lake Aniwaniwa were more than 100 m long, reflecting the shallow depth of this lake and the fact that macrophytes were commonly found right across the lake. The macrophyte flora within Lake Aniwaniwa was dominated by three introduced plants: hornwort (*Ceratophyllum demersum*), Canadian pondweed (*Elodea canadensis*), and curly oxygen weed (*Egeria densa*), all of which had an average maximum cover of between 51 and 75%. Hornwort and curly oxygen weed in this lake often occupied the entire water column up to 5.5 m deep, and these plants often grew in extensive beds within the lake. Four other macrophytes, of which two were native, were also found in this lake (Table 15), but with much lower cover. Calculations of the LakeSPI data showed that the LakeSPI index scored poor (12%), while the native condition scored low (8%) and the invasive impact condition scored very high (95%). These results contrast with earlier LakeSPI measurements at this lake conducted in 1983. Measurements at this time recorded the LakeSPI index as being moderate (24%), with a native condition index of 15%, and an invasive impact index of 78%. The reduction in LakeSPI score over this 30 year period reflects the increased cover of invasive macrophytes within the lake.

All transects within Lake Matahina were very short (less than 25 m), reflecting its generally deeper bathymetry. Ten macrophyte species were recorded in Lake Matahina (Table 15), with curly oxygen weed and hornwort dominating. Five native macrophytes were found in this lake. The average maximum cover was, however, much less than that observed in Lake Aniwaniwa (Table 15). Calculated LakeSPI scores also ranked this lake to be in poor condition, with a LakeSPI score of 10%, and native condition score of only 3%, and an invasive impact score of 96%.

*Table 15 List of macrophyte species and their average maximum cover found in a survey of six transects in each lake conducted in March 2014. * Indicates that these plants are native.*

Lake Aniwaniwa		Lake Matahina	
Species	**Average of maximum cover**	**Species**	**Average of maximum cover**
Egeria densa	96-100%	*Egeria densa*	51-75%
Ceratophyllum demersum	76-95%	*Ceratophyllum demersum*	26 – 50%
Elodea canadensis	51-75%	*Glossostigma submersum**	6 – 25%
*Chara australis**	6 – 25%	*Glossostigma elatinoides**	1-5%
Lemna minor	6 – 25%	*Lilaeopsis novae-zealandiae**	1-5%
Potamogeton crispus	1-5%	*Elodea canadensis*	1-5%
*Potamogeton ochreatus**	1-5%	*Callitriche*	1-5%
		*Nitella hookeri**	1-5%
		Potamogeton crispus	1-5%
		*Potamogeton ochreatus**	1-5%

6.4 Discussion

Both lakes displayed TLI values characteristic of highly nutrient rich systems. Lake Aniwaniwa had significantly higher levels of TN than Lake Matahina, and significantly lower levels of chlorophyll, and higher clarity. Both lakes also supported macrophyte communities indicative of lakes in poor condition, based on their LakeSPI score. This reflected the very high cover and biomass of exotic macrophytes such as hornwort and curly oxygen weed. The observed LakeSPI scores from Lake Aniwaniwa had also decreased considerably between 1983 and 2014, from moderate to poor.

Lake Aniwaniwa is only a relatively new ecosystem, and prior to the construction of the Aniwhenua Dam in the mid 1970's, the river was likely to have supported plants such as Canadian pondweed (*Elodea canadensis)*, curly pondweed (*Potamogeton crispus*), native pondweed (*Potamogeton ochreatus*), and water buttercup (*Ranunculus trichophyllus*) (Boubee et al 2009). Following completion of the Aniwhenua Dam in 1979, macrophyte surveys in the early 80s documented successional changes in the macrophyte communities, and showed that by 1984, both *Elodea* and *Lagarosiphon major* had become dominant plants. Roughly around the year 2000, hornwort and Egeria invaded Lake Aniwaniwa, and these two plants now form extensive surface reaching growths which have displaced much of the pre-existing aquatic vegetation (Boubee et al 2009). The results of the LakeSPI surveys also confirmed the dominance of these two plants.

There is little doubt that the extreme weed growth in Lake Aniwaniwa has negative impacts on many of the values of the lake, including aesthetic and recreational values. These plants may also have adverse effects on ecological values, especially if dissolved oxygen levels within the extensive weed beds drop during the night when plant respiration is high (Kaenel et al., 2000; Wilcock et al., 1998). As such, macrophyte growth in this lake would be considered as being in the "intolerable" range as suggested by de Winton and Schwartz (2004). Accordingly, appropriate control methods should be implemented in this lake.

Plants such as hornwort and *Lagarosiphon* are so widespread through Lake Aniwaniwa that management options are somewhat limited. Widespread eradication throughout this lake is not possible, so management should be focused on containment and control at specific sites. Possible control measures may include the use of mechanical harvesters or cutters, although these are very difficult to decontaminate, and may spread weed fragments to other lakes (Closs et al., 2004). Other control measures could include the use of herbicides such as Diquat.

However, high macrophyte biomass in Lake Aniwaniwa may be having beneficial effects on the ecology of the lower Rangitāiki River. Results of our spot water quality monitoring of the two lakes showed that inorganic-N levels were significantly lower in Lake Matahina than Aniwaniwa. This is despite Lake Matahina being located further downstream and draining a greater area of pasture. Results of long-term water quality monitoring also show a reduction in inorganic-N loads between the two lakes (Paul Scholes, BOPRC, Pers comm). Moreover, the significant increase in nutrient loads in the Rangitāiki at the NIWA monitoring site at Murupara (Boubee et al 2009) are not evident at the lower NIWA monitoring site at Te Teko, suggesting some form of nutrient uptake may be occurring between the two locations. These results suggest that the luxurious macrophyte growth in Lake Aniwaniwa may be removing nutrients from the water column, and as such improving the water quality in the lower Rangitāiki River. Consideration could be given into investigating the practicality of harvesting macrophytes from this lake as a way of further reducing the nutrient loads in this already nutrient rich lake. Such macrophyte harvesting is currently conducted in Lake Rotoehu, where it is used by BOPRC as a management tool to help remove nutrients from Lake Rotoehu in order to help meet its TLI objective (Bay of Plenty Regional Council 2007 Action Plan). If a similar harvesting were conducted in Lake Aniwaniwa, it could have benefits of both assisting in the nutrient removal and in improving recreational values of the lake.

Lake Matahina has much steeper sides than Lake Aniwaniwa, as well as high chlorophyll *a* levels and lower clarity. These factors may explain the lower macrophyte cover in Lake Matahina than was found in Lake Aniwaniwa.
Lake Matahina is thus fundamentally different to Lake Aniwaniwa and appears to be more phytoplankton dominated. As such, this lake is not expected to reduce nutrient concentrations in the water column to the similar extent than was observed in Lake Aniwaniwa. However, the high amounts of phytoplankton would act as an important food source to invertebrates below the Matahina Dam, and indeed high densities of filter feeding invertebrates such as *Aoteapsyche* were observed in the lower river to Te Teko. The high levels of phytoplankton in the lake may also explain the significantly reduced water clarity in the Rangitāiki River below the Matahina Dam. This reduced clarity may have adverse effects on aesthetic and recreational values and in particular on swimming. Indeed, examination of water clarity obtained from the NIWA and BOPRC monitoring sites along the Rangitāiki River show a significant reduction in clarity from 2.04 m at Murupara, to only 1.4 m at the Matahina Dam and Te Teko.

Part 7: Monitoring Recommendations

This “snap-shot” assessment of the ecological condition of waterways throughout the Rangitāiki Catchment has clearly shown that many of them are in either good or excellent ecological condition. However, ecological condition was reduced in small streams draining pasture catchments, and in the lower mainstem of the Rangitāiki River, reflecting changes in water chemistry, habitat conditions and flow regimes. Land use activities were also shown to affect water quality, with higher bacterial levels and concentrations of inorganic-N in streams draining pasture catchments than other land uses. Results of the ecological monitoring showed that increases in nutrients did not, however, have large negative impacts on invertebrate communities. This may have reflected the preponderance of mobile fine bed material in many streams, and the resultant lack of high algal cover in them. The results of this study have improved our understanding of ecological responses to various pressures throughout the catchment. It is hoped that this will assist both the RRF and the community to agree on desired outcomes for the river and other waterways throughout the catchment.

A number of recommendations are made below to allow for collection of more data to help better understand processes operating throughout the catchment, to help clarify management objectives, and help set limits for parameters such as nutrients and algal biomass. These recommendations cover aspects to do with: water quality in both rivers and lakes; assessments of algal cover; management of riparian areas; development of macrophytes and management strategies for Lake Aniwaniwa; undertaking cultural health investigations throughout the catchment. The recommendations will also provide fundamental data that will help underpin many of the objectives and actions identified in the Te Ara O Rangitāiki – Pathways of the Rangitāiki Draft Discussion Document.

7.1 Water quality monitoring

The ecological assessments were based on one-off surveys of invertebrate communities to assess stream health. Invertebrates were used to assess stream health as they integrate antecedent environmental conditions, so a one-off sampling regime is thought sufficient to accurately document stream health. Water quality is, however, far more temporally variable and so some of the observations that were made in the study need to be further validated by undertaking more sampling.

Recommendation 1

Continue with monthly water quality monitoring in some of the sites already selected for water quality monitoring in the study for up to a 12 month period. This will better characterise the effects of land use on sediment and nutrient exports from catchments, and in particular from small catchments where the interaction between land cover and water chemistry would be more intimate. Monitoring for a 12 month period will also provide a more accurate picture of the links between water quality and climate, as many water quality parameters very seasonally and respond to rainfall.

7.2 Algal monitoring

Despite the limited water quality sampling conducted, it was clear that nutrient exports from pasture streams were often high. Mean inorganic-N concentrations observed in some pasture streams in particular were above 0.8 mg/L, a level suggested necessary to maintain healthy ecosystems (Death 2014). The fact that we did not see a large amount of ecological degradation even in streams exposed to high nutrients was partially a reflection of the dominance of fine substrates in some sites throughout the catchment, where algae cannot grow. However, high algal biomass was observed at some sites during sampling, especially in the Rangitāiki mainstem where large substrates were common. In particular, occasional blooms of *Phormidium* were observed in some stable sites in the Rangitāiki River above Lake Matahina. Visual assessments of percentage cover of *Phormidium* at these sites showed that the MfE recommended guidelines for contact recreation (<40% of the streambed) were exceeded. Studies by Heath et al. (2012) in the Hutt and Wainuiomata Rivers near Wellington showed that very high cyanobacterial cover was observed, despite low nutrient levels in these rivers. They concluded that river flow and temperature were the only significant predictors for cyanobacterial proliferations. However, further work by Wood and Young (2012) suggested that nitrogen levels do limit cyanobacterial growth, and that New Zealand *Phormidium* is unable to fix nitrogen. They concluded that increased nitrogen concentrations are required before *Phormidium* will bloom. Given that nitrogen levels are increasing in the Rangitāiki River (Boubee et al 2009), then it is highly likely that *Phormidium* biomass will also increase at sites within the river, particularly during the summer when flows are low and temperatures high.

Recommendation 2

Monitor algal cover and/or biomass from selected sites in the catchment to determine whether a) *Phormidium* biomass reaches levels exceeding MFE guidelines, and b) whether algal cover/biomass exceeds recommended MfE levels (Biggs 2000) for the protection of ecological, recreational or aesthetic values. Such monitoring could also include soft-bottomed sites with macrophytes growing in them to see whether excessive filamentous algal biomass can cause macrophytes to die off. The results of this monitoring program would provide valuable information in terms of setting nutrient limits, and would give both the Rangitāiki River Forum and the general community valuable information as to the likelihood of algal blooms to form.

7.3 Management of riparian areas

Our analysis of driving factors influencing invertebrate communities throughout the catchment identified the importance of many factors, including large-scale factors such as location within the catchment, distance to sea, waterway size, stream slope and flow type. Small-scale factors such as HABSCORE and the degree of bank undercutting were also important. HABSCORE was based on scoring ten categorical factors measured at each site. Many of the HABSCORE factors such as the nature and size of riparian and buffer vegetation, stream shading, bank stability, and stock access are influenced by land use activities. Factors such as riparian vegetation, shading and stock access can be relatively easily manipulated by appropriate fencing and riparian planting in areas where this is currently lacking.

Recommendation 3

Work with land management offices and land owners to ensure that small waterways which are currently unfenced and which do not support riparian vegetation are fenced and planted. This recommendation could be extended to streams draining pine plantations, to ensure that where possible riparian buffer strips are either retained or allowed to establish in streams following harvest. A key part of this recommendation would be to quantify the current extent of streams in the Rangitāiki presently lacking such fencing and riparian vegetation, and in setting specific numeric objectives for the kilometres of fencing and planting to be achieved at sites where this is lacking.

7.4 Lake water quality monitoring

NIWA's long-term monitoring of water quality at Murupara has shown significant increases in nutrients, particularly inorganic-N from 1989 onwards. This increase was particularly evident after 2004, and was attributable to land use intensification in the upper part of the catchment, in particular the Otamatea River (Boubee et al., 2009). However, analysis of the long-term data collected at Te Teko has shown no similar increase in nutrients; indeed nutrient levels in the lower Rangitāiki appear lower than in the upper catchment (P. Scholes, BOPRC, Pers. comm). Such dramatic reduction in nutrient concentration at the two sites may reflect a large degree of nutrient uptake at Lake Aniwaniwa by the high cover of exotic macrophytes there. Our short-term analysis of the TLI data collected from Lakes Matahina and Aniwaniwa generally support this hypothesis and suggests that the two lakes may be performing useful nutrient stripping functions. The information gleaned from the short-term quality in both lakes forms the basis of the following three recommendations:

Recommendation 4

Continue on with monthly TLI sampling from the Lakes Aniwaniwa and Matahina to better characterise both nutrient and chlorophyll levels in these two lakes. This would provide useful information as to degree of nutrient retention in Lake Aniwaniwa and of phytoplankton biomass in Lake Matahina.

7.5 Lake macrophyte management

Recommendation 5

Members of the RRF and BOPRC work with the local communities, power companies, and other relevant stakeholders to develop an Action Plan for Lake Aniwaniwa where excessive growths of introduced macrophytes have significant adverse effects on values such as fishing, water-skiing and aesthetic values.

Recommendation 6

Undertake a cost benefit analysis of macrophyte control using methods such as herbicide application, or weed harvesting. Weed harvesting could be used to remove excess nutrients from Lake Aniwaniwa and could possibly even increase the storage capacity of the lake.

7.6 Cultural health assessments

This report has focused solely on western science assessments of stream health. It has shown that ecological health in many of the smaller waterways in the upper parts of the catchment is good or excellent, but that ecological health in the lower mainstem is generally lower. In the absence of detailed cultural stream health assessments throughout the catchment, it was suggested that the results of the ecological monitoring could provide some initial useful insights to members of the Rangitāiki River Forum as to the cultural health and Mauri of the waterways. However it is also acknowledged that there are important differences between western science and Maori cultural health assessments, and these differences need to be acknowledged in any interpretation of Mauri using western science assessments.

Recommendation 7

Undertake cultural health investigations throughout the catchment using the Cultural Stream Health Measure methodology as outlined by Tipa and Tierney and as used by Suren and Lee (2014). Any cultural assessment will need to include sites of relevance to iwi/hapu throughout the catchment, as well as, where possible, sites examined in this work. Any cultural stream health assessment will also need close liaison between different hapu to agree on a methodology used to assess cultural health.

Part 8: Relevance to the draft Rangitāiki River Document

The Rangitāiki River Forum has prepared Te Ara O Rangitāiki – Pathways of the Rangitāiki, a draft document for consultation that sets out its vision, goals and strategies. The forum has identified four desired outcomes for the Rangitāiki River:

- Mauri - Mauri of the water is protected;
- He Tangata - We will have a balanced, connected and respectful relationship with the rivers and resources of the Rangitāiki;
- He Taiao - We want to see a bountiful river– where native habitats and customary harvesting practices sustain people and where whitebait and tuna (eels) abound;
- He Awa - We want to see a celebrated, clean and healthy environment – characterised by clean water, healthy ecosystems and the return of some threatened species. People use and enjoy this environment for their spiritual, cultural and recreational needs and celebrate its heritage with pride.

The results of this ecological survey are relevant to two of these outcomes, Mauri and He Awa. By its very nature, the survey relied on western science assessments of stream health, and does not attempt to assess or quantify concepts such as Mauri. However, recent work has shown strong linkages between western science assessments and Maori cultural assessments using a Cultural Stream Health Measure (CSHM: Tipa and Teirney, 2006). Although the Tipa and Teirney methodology was developed for the South Island, Suren and Lee (2014) reported strong similarities in the ranking of stream ecological condition using the western MCI approach, and the CSHM approach in 37 streams in the Rotorua region. These similarities suggest that, for the time being at least, the MCI scores used in the study could act as a surrogate measure of a CSHM. Using this approach, it is clear that many of the sites throughout the catchment have generally good to excellent ecological health, and could therefore be assumed to have a high degree of Mauri, and would reinforce the concept of He Awa. The fact that many of the sites were in relatively intact native bush will hopefully confirm this assumption. However, as noted, ecological health in streams draining plantation forestry or even pasture was also generally high, and so it can only be assumed that these sites also maintain a relatively high degree of Mauri.

Sites in the lower Rangitāiki had low MCI scores. These sites also displayed the biggest difference between observed and predicted MCI scores. The reduced stream health here was thought to reflect a combination of changes in flow regime, reduced water quality from increased nutrient levels and lower clarity, and potential changes to instream habitat conditions from the high cover of introduced macrophytes in parts of the lower river. Sites in the lower Rangitāiki would thus be expected to have low Mauri, reflecting the extensive modifications to the river in terms of channel modification through presence of riprap and stop banks, little overhanging vegetation, or willows (many of which are being removed as part of bank protection work), and the fact that these sites are below a number of point-source discharges (for example the Fonterra Dairy Factory at Edgecumbe: Figure 27).

Figure 27 Examples of activities occurring along the banks of the lower Rangitāiki River below Te Teko including A and B) extensive bank protection works using riprap; C) presence of introduced willows, many of which are being removed; D) industrial water takes and discharges from the Fonterra Dairy Factory at Edgecumbe.

Suren and Lee (2014) also highlighted some important differences between western science and Maori cultural health assessments. For example, from a western science perspective, the effect of a contaminant will decrease as it is diluted, whereas from a Maori perspective, its impact will persist throughout the catchment, irrespective of the degree of dilution. Given this, it is highly likely that a large disconnect could exist between the Mauri of sites in the Rangitāiki below the Murupara oxidation ponds discharge, despite the fact that the high MCI scores observed in the Rangitāiki River below Murupara and above Lake Aniwaniwa suggested that the river there was in good or excellent condition.

Notwithstanding these differences, it is suggested that many of the tributary streams draining into the Rangitāiki still maintain a high degree of Mauri, and are likely to contribute to He Awa. In the absence of detailed cultural health assessments, it is suggested that the ecological data could be used as an interim surrogate to assess the degree to which waterways throughout the catchment meet these two outcomes. Detailed cultural health investigations of sites throughout the catchment are planned, and the combined results of both studies would go a long way to ensure the outcomes of the River Forum are met. Furthermore, assessments of stream condition using both western and Maori cultural methods can be carried out at sites over time to see whether any management or restoration interventions have improved their condition, expressed either as ecological health or as improved Mauri or He Awa.

The draft Rangitāiki River Document also identified eight specific objectives that need to be achieved to reach the desired outcomes. The following objectives are particularly relevant to the ecological work presented in this report:

- Objective 2: The habitats that support indigenous species and linkages between ecosystems within the Rangitāiki River catchment are protected and enhanced, "so the tuna (eels) are fat and plentiful in the Rangitāiki River waterways".
- Objective 3: Prosperity in Rangitāiki Catchment is enabled within the sustainable limits of the rivers and receiving environment
- Objective 4: Water quality is restored in the Rangitāiki Catchment, "so it is safe for people to swim in, take food from and find safe drinking water in places".

The results of our monitoring, combined with studies of Boubee et al (2009) has clearly shown that land use change is affecting water quality, particularly in streams draining pastoral land use, where bacterial loads and concentrations of inorganic-N are higher than in other land uses, and where pasture catchments have significantly higher catchment yields of inorganic-N. From a water quality perspective, these trends pose considerable challenges in meeting the above objectives, especially given the low water clarity and often high bacterial loadings in the Rangitāiki mainstem following rainfall events (Boubee et al 2009). However, the results of our ecological monitoring showed that these increases in nutrients do not appear to as yet have large negative impacts on invertebrate communities in the upper parts of the catchment. This was thought to reflect the preponderance of mobile fine bed material, and the resultant lack of high algal cover in many of the waterways examined.

However, there were areas in the mainstem of the Rangitāiki and in other gravel bed tributary rivers where high algal biomass was observed. Additionally, anecdotal evidence from Fish and Game officers who routinely drift dive the upper Rangitāiki River has suggested that filamentous green algae is becoming more common on macrophytes in this area (Rob Pitkethley, Eastern Fish & Game, pers. com. June 2014), presumably in response to increased nutrients (Figure 28). High algal growth on macrophytes can lead to their disappearance as algae smother the macrophytes and out-compete them for light (Bakker et al., 2010; Phillips et al., 1978). Indeed, this may have been observed by Fish and Game officers during part of their annual drift dive surveys in the upper Rangitāiki. In March 2013, abundant macrophytes grew in parts of the river, yet the following year these plants had disappeared (Figure 29). Given the importance of macrophytes as invertebrate habitat (Collier, 2004; Gregg and Rose, 1985), their loss from parts of the river is likely to have a large detrimental effect on other components of the ecosystem. If this is happening, it suggests that nutrient levels within the upper parts of the catchment may indeed be approaching levels exceeding the sustainable limits of the receiving environment.

Figure 28 Photo of submerged aquatic macrophytes in the upper Rangitāiki River near the confluence with the Otamatea River, showing the extensive growths of filamentous green algae on them. (Photo courtesy of Eastern Fish & Game).

Figure 29 *Photos of changes in macrophyte cover at the same site in the upper Rangitāiki River. In 2013, luxuriant macrophyte growths were observed, but these were covered with filamentous green algae. By 2014, macrophyte cover had decreased considerably, possibly as a result of being smothered by the filamentous algae. (Photo courtesy of Eastern Fish & Game).*

The Draft River Document has clearly highlighted an expectation of "restoring the water quality" throughout the catchment. Communities thus want water to be swimmable, drinkable, abundant, and suitable for ceremonies and to sustain mahinga kai (Bay of Plenty Regional Council, 2014). Challenges now lie ahead for BOPRC, as well as the community and relevant stakeholders within the Rangitāiki Catchment to decide what is meant by "restoring the water quality" and how success of this goal can be measured. A key component of any restoration activity is to have a clear understanding of the value within the catchment that is to be restored, as well as the pressures which are adversely affecting that value. However, values such as swimming, drinking and sustaining mahinga kai are all individually affected by factors such as water clarity, presence of *E. coli*, nutrients and algae (Table 16). Thus, rivers with low clarity, high levels of nutrients and bacteria, and high amounts of algae would not support these values. In contrast, rivers with high clarity, low levels of nutrients and bacteria and small amounts of algae would. At first glance management of the factors that influence these values would allow these values to be met.

Table 16 List of major factors that influence three values (swimming, drinking and food species) that the forum and community place on waterways in the Rangitāiki Catchment.

Swimming	Drinking	Suitable for food species
Clarity	*E. coli*	Habitat
E. coli	Cyanobacteria	Nutrients
Nutrients	(Nutrients)	Algae (and cyanobacteria)
Algae (and Cyanobacteria)		

However, our survey work has shown that increased nutrients do not always result in algal blooms. High nutrients may thus not preclude a river from being valued for swimming or being suitable for food species, especially if algal blooms do not occur. Algal biomass in the catchment could be controlled by a mixture of factors including nutrients, riparian shading, and substrate size. The latter two factors are highly variable throughout the catchment, so any objective focusing on reducing nutrient inputs would need to determine what an acceptable level of nutrient enrichment (and subsequent algal blooms) is - both in terms of location, as well as the time that blooms last for. Presently, no information exists on the spatial and temporal dynamics of algae throughout the catchment. Implementation of the algal monitoring programme will document the nature and extent of algal blooms throughout the catchment. Only with this information can we properly describe the current state of algal biomass throughout the catchment and determine whether unacceptable blooms occur, which may be having adverse effects on values such as swimming.

If unacceptable blooms are found, BOPRC needs to implement management interventions to limit algal biomass through shading (in small streams) or reducing nutrients through policies, methods and rules in their Water and Land Plan. Riparian planting to control shade only works in relatively small streams (up to 6 m wide (Quinn 2003)), and so is limited to mainly the smaller tributaries. However, because these small streams eventually flow into larger rivers, riparian planting in small streams may potentially lower nutrient inputs into the larger rivers by controlling direct runoff into the smaller tributary streams. However, controlling nutrients in the highly porous pumice geology found throughout the catchment may prove difficult. Traditional techniques of intercepting nutrients such as riparian planting (Howard-Williams and Pickmere, 1986; Reeves et al., 2004) may not be as effective in the Rangitāiki as they are in catchments with less porous soils. Instead, techniques such as constructing and/or improving wetlands that small streams flow into, and which may intercept a proportion of groundwater may be required. Depending on where they are located within the catchment, any constructed or natural wetlands could also provide suitable habitat for a wide range of fish, bird and invertebrate species.

While nutrient management is somewhat problematic in the absence of detailed information on algal blooms, management of water quality for bacteriological contamination is easier both in terms of defining acceptable limits, and in implementing activities designed to reduce bacterial loadings. Firstly, clearly defined numerical guidelines exist as to what constitutes safe bacteriological contamination for both recreation and drinking (e.g., ANZECC, 2000). Secondly, it should be relatively easy to exclude cattle from streams. Under the current Regional Water and Land Plan (RWLP), stock are meant to be excluded from "all rivers and streams with a Natural State (river) water quality classification" (Objective 62 (b)). Given that the Rangitāiki is classified as a Natural State river, then the regulatory mechanisms to achieve this already exist. The challenge is to implement and enforce these objectives, as there was clear evidence during our fieldwork that stock had direct access to the Rangitaiki River.

Not only were cattle observed in the water, but freshly deposited manure was evident in some locations at the edge of the river (Figure 30). Cattle have profound effects on water quality and stream ecology (Davies-Colley et al., 2004), and so ensuring compliance with the RWLP should go a long way to improve water quality. If this were done, then many of the objectives outlined in the Te Ara O Rangitāiki – Pathways of the Rangitaiki Draft Discussion Document could also be met.

Figure 30 Examples of A and B) cattle seen in the water in the mainstem of the Rangitāiki River (near site RES_139 and RES_131), as well as B) evidence of manure on the edge of the water (site RES_108). Cattle trampling will lead to C) deposition of fine sediments in the riverbed, as also observed at RES_108. The location of these sites is shown in Appendix 1 (in grey).

Implementation of the seven recommendations outlined in Section 7 will help achieve the objectives of the Rangitāiki River Forum as identified in the draft Rangitāiki River Document (Table 17). For example, the data obtained from implementation of recommendations 1 (water quality monitoring) and 2 (algal monitoring) will provide information of direct relevance in providing sustainable catchment load limits for nutrients, particularly nitrogen, and its link to algal blooms - if these occur. Monitoring water quality will also provide information on *E. coli* contamination from many of the smaller waterways where this data is lacking. This information is fundamental to ensure that water quality in the Rangitāiki is restored. Implementation of recommendation seven is also central in ensuring that Kaitiakitanga is recognised and provided for (Table 17).

Table 17 List of selected objectives and contributing actions as identified in the draft Rangitāiki River Document, and the relevant recommendations made for further monitoring to help achieve these objectives.

Objective: The habitats that support indigenous species and links between ecosystems within the Rangitāiki River Catchment are protected and enhanced.		Recommendation
Contributing actions	Encourage restoration with appropriate vegetation along waterways, where suitable.	3, 5 and 6
	Work with industries, land owners and agencies to fence of waterways, plant riparian margins and remove pest plants.	3
	Implement a coordinated programme to identify, prioritise, protect and enhance the existing ecosystems, significant sites and connections in the catchment.	7
Objective: Water quality is restored in the Rangitāiki Catchment.		**Recommendation**
Strategic action	Develop sustainable environmental flow and catchment load limits (e.g. nutrients, sediments and bacteria) through the Freshwater National Policy Statement framework.	1, 2 and 4
Contributing actions	Initiate strategies for managing water, waste water and stormwater in the district, in consultation with the community and tangata whenua, including investigations into treatment and discharge options.	1, 2 and 5
	Identify, forecast and assess emerging pressures on the resources in the Rangitāiki Catchment and likely opportunities and targets for restoring water quality.	1, 2, and 4
Objective: prosperity in the Rangitāiki Catchment is enabled within the sustainable limits of the rivers and receiving environment.		**Recommendation**
Contributing actions	Work with rural industries, iwi, land owners and other willing stakeholders in the Rangitāiki Catchment to articulate their aspirations for prosperity and values for freshwater through the Freshwater National Policy Statement framework.	1, 2 and 4
Objective: the relationships between the community and the Rangitāiki Catchment are recognised and enhanced.		**Recommendation**
Contributing actions	Develop and implement a cultural health index (CHI) for the Rangitāiki, Whirinaki, Wheao and Horomanga Rivers, which incorporates Matauranga Mauri (Maori knowledge) methods.	7
Objective: The practice of Kaitiakitanga (guardianship) in decision-making for managing the resources of the Rangitāiki River Catchment is recognised and provided for.		**Recommendation**
Contributing actions	Collect an inventory of waahi tapu (sacred) sites in the catchment.	7
	Develop a protocol for accessing, holding and using the waahi tapu (sacred site) information.	7
	Conduct a survey to collect information on tikanga (custom) associated with rivers of the Rangitāiki.	7

Part 9: Relevance to the National Policy Statement

The seven recommendations made in this study also have direct relevance to implementation of Central Government's National Policy for Freshwater. Part of the NPS has been the creation of a series of National Objective Frameworks (NOFs) that, amongst other things, sets grading bands (A, B, C and D) for a number of numerical water quality and ecological attributes. Under the NPS, communities are expected to work with councils to help identify desired states for waterways. All activities within a catchment are therefore to be managed to achieve these desired states. National bottom lines have been identified (Band D) which are the lowest permissible levels for particular attributes. The NPS also clearly states that ecosystem health and water quality cannot decline from its current (2014) state, effectively precluding communities from managing waterways to a lower state.

The following description is an example of how the recommendations made in this study link with the requirements of both the NPS, and also the aspirations of the draft Rangitāiki River Document. Reference to Figure 31 shows the inter-relationships of the different documents. There is clear evidence that, all other things being equal, increases in nutrients can result in increased algal blooms (Death 2013, Wright 2013), or increases in *Phormidium* cover (Heath et al 2012). Increased algal blooms can then lower ecosystem health and/or recreational values (Biggs 2000). Such a reduction in ecosystem health would be contrary to the intent of the NPS. Recommendations 1 and 2 are therefore to monitor water quality and algae. Under the NOF, councils are required to set bands for algal biomass (Box 1 in Figure 31), so information on the current extent of algal biomass throughout the catchment is fundamental. This information can only be obtained through monitoring by implementing Recommendation 2 of this report (Box 2). Community consultation and engagement (Box 3) will also help define values and objectives for waterways throughout the catchment (Box 4), and so any monitoring regime will need to be aware of these values, to ensure that relevant parameters or sites are being measured.

Once the current state has been properly described through monitoring by implementing Recommendation 2 (Box 5), further consultation with the community is required to determine what level of algal biomass is acceptable, and what NOF band is appropriate for various waterways throughout the catchment (Box 3). Note that community consultation is also part of the Rangitāiki River Document, and that discussions on acceptable levels of algal biomass have direct relevance to discussions about the desired outcomes for the Rangitāiki River in terms of Mauri and He Taiao. The algal monitoring programme will then allow an assessment to be made as to whether the current state of algal biomass at selected sites is greater than or less than what the communities desire (Box 6). If algal biomass is below an acceptable level, then there is no issue, as that site meets the requirements of the community. Under such a scenario, the outcomes of the Rangitāiki River Document are also expected to have been met. In contrast, where algal biomass is higher than desired, BOPRC will need to implement a system of policies, methods and rules to help reduce algal biomass at these sites (Box 7). This may or may not need further community consultation. Setting rules will however require further monitoring to determine whether their implementation is indeed having the desired effects.

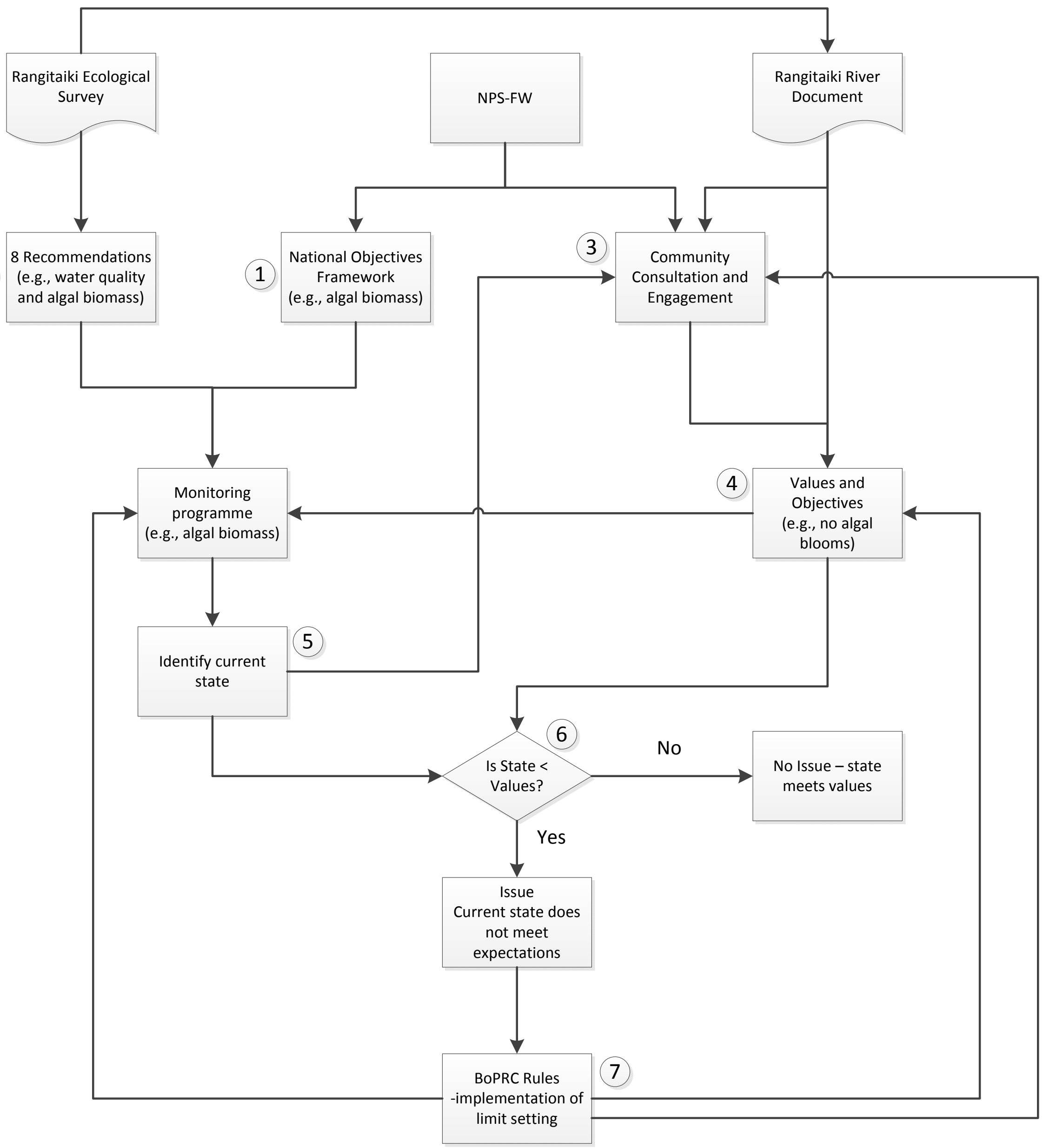


Figure 31 Flow chart demonstrating links between the recommendations made following the Rangitāiki Ecological Survey (this study) and the need to undertake a monitoring program as part of implementation of the NPS. Also shown is the need for community engagement from both the NPS and the Rangitāiki River Document in terms of setting desired values and objectives for the Rangitāiki River.

As part of monitoring algal biomass, nutrient samples will also be collected to help determine relationships between nutrients and algal biomass (Recommendation 1). This information will consequently help BOPRC in setting nutrient limits in waterways in order to maintain a desired algal biomass. Note that long-term monitoring is required in order to ascertain the success (or not) of any policies and methods and rules designed to limit nutrients with the objective of limiting algal biomass (Figure 31).

Part of the consultation process will need to establish clear links between land use change, nutrient enrichment of streams, increased algal biomass, and reduced ecosystem health or recreational values. A cost-benefit analysis is central to this consultation process, whereby communities need to understand the economic implications of selecting a particular NOF band for an attribute such as algal biomass. For example, increased land use intensification in the upper Rangitāiki, or Galatea Plains may result in enhanced economic benefits for some individuals living in the greater catchment. There may, however, be potential adverse effects to water quality and ecological and cultural values in the lower catchment if the further nutrient enrichment results in increased algal biomass, or *Phormidium* blooms. Although the initial field work conducted for this survey highlighted the lack of obvious algal blooms as a result of the fine pumice streambeds, anecdotal evidence of the reduction of macrophyte cover in parts of the upper catchment suggests that the river may be nearing a “tipping” point with regards to nutrient enrichment. Further monitoring is obviously needed to support this contention.

Nevertheless, any environmental "cost" to the river and the services it provides will need to be balanced against the "benefits" that land use intensification may bring. BOPRC cannot make these decisions in isolation of the community residing throughout the catchment, but only through informed and rigorous debate about the values that we are trying to protect, and potential trade-offs with competing values. Such debates can only be enhanced by the provision of robust good quality data, which can only be obtained by implementation of long-term robust monitoring programs as recommended in this report.

Part 10: Acknowledgements

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Part 12: Appendix 1

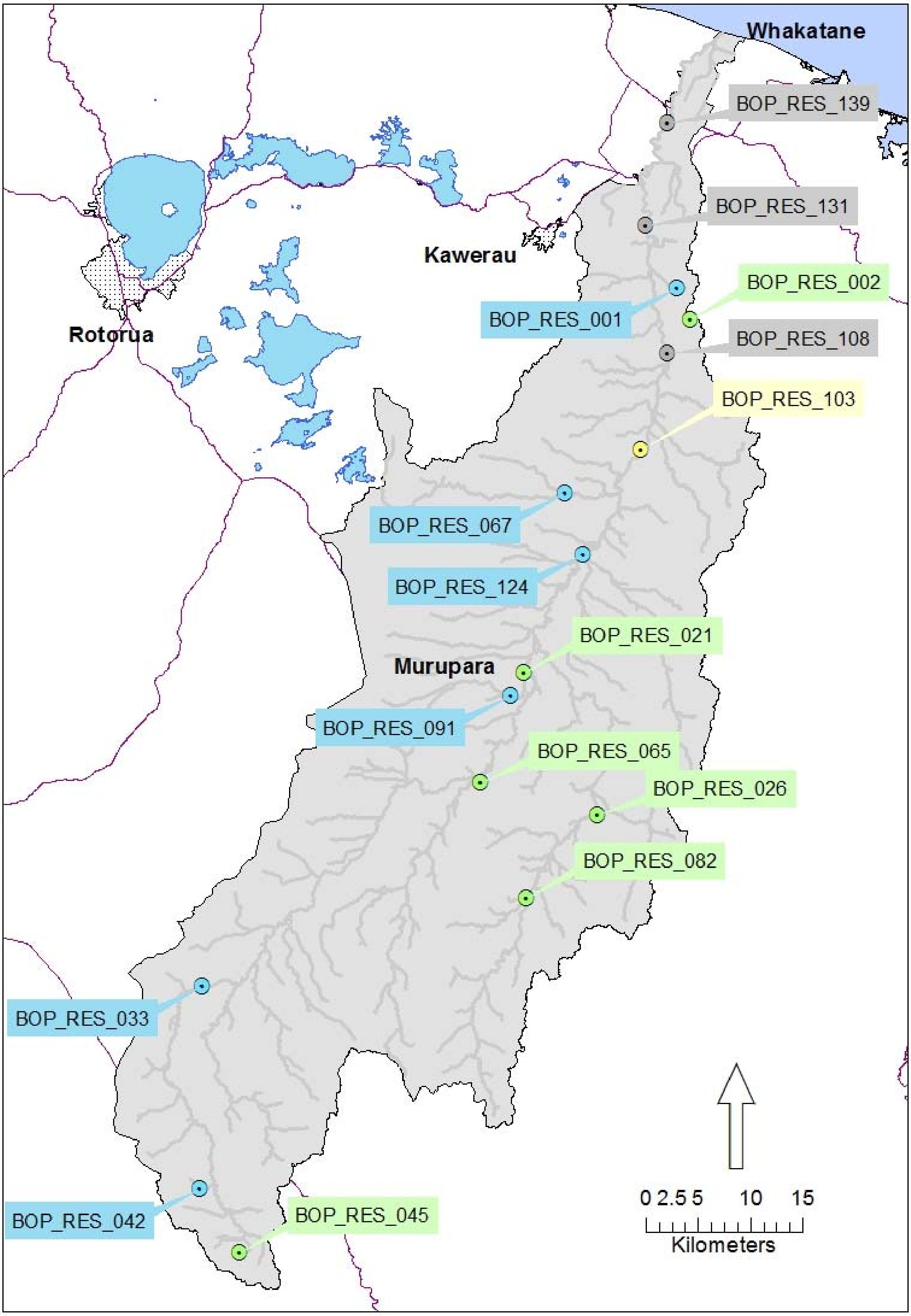

Figure A1 Map showing the locations of all sites referred to throughout the report.