Ecological and water quality conditions of drains and land drainage canals in the Rangitaiki and Kaituna Plains

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Technical summary

1 A survey was undertaken of drains and land drainage canals in the Kaituna and Rangitāiki Plains to help fill knowledge gaps about these relatively unstudied ecosystems. This study had three main aims:

(a) To improve our state of knowledge about the habitat, water quality and ecological conditions of drains throughout the Kaituna and Rangitaiki Plains. This information is vital as part of informing the community as to the current condition of these waterways, and may have relevance in setting appropriate numerical bands for water quality parameters under the National Policy Statement for Freshwater Management.

(b) To highlight potential hotspots of contaminants or areas of low dissolved oxygen in drains which ultimately discharge into natural waterways. Some of this information will have implications for the management of sensitive downstream receiving environments (e.g. the Maketū and Waihī Estuaries).

(c) To provide good baseline information on water quality and ecological conditions to help improve the future management of drains. This information will be useful to guide operational activities in the drainage systems, and to inform environmental assessments of the effects of the drain discharges into other waterways.

This information is reported here and is followed by a discussion of potential ways to mitigate the adverse effects of the current conditions in the drainage network.

2 Twenty sites were surveyed: six in the Kaituna and 16 in the Rangitaiki. This data was further complemented by data obtained from the Council's Natural Environment Regional Monitoring Network (NERMN), consent applications, and other Council monitoring programmes throughout the region that assessed habitat quality (385 sites), water quality (68 sites) and invertebrate communities (435 sites). Fish were surveyed from six sites in the Rangitaiki Plains, and this data was supplemented by further data obtained from the New Zealand Freshwater Fish Database (NZFFD) from 47 sites, from lowland areas (< 20 m ASL) in the Kaituna and Rangitāiki Plains.

3 All sites were allocated to their appropriate water quality classification, based on the Regional Natural Resources Plan (RNRP). Two classifications were particularly relevant for this study; Drain Water Quality (DWQ) and Modified waterways with Ecosystem Values (MEV)[1]. All MEV streams (hereafter referred to as MEV streams) must have freshwater objectives set as part of the National Policy Statement for Freshwater Management (NPS-FM) process, as they are freshwater bodies based on NPS-FM and RNRP definitions, whereas DWQ drains do not, reflecting their utilitarian function and definition as artificial water courses. Allocating sites to their appropriate water quality classification gave us the basis for comparing conditions in DWQ and MEV sites based on:

- Habitat conditions.
- Water quality.
- Invertebrate communities.
- Fish communities.

[1] All modified water courses with MEV classification are land drainage canals as defined in the RNRP.

While the key objective of the drainage network (i.e., DWQ drains and MEV streams) is to drain wet land, it must be remembered that any waterbody represents aquatic habitat which will be colonised by species tolerant of environmental conditions there. A man-made waterbody classification thus has no ecological relevance or bearing on what species are, and are not found there.Habitat conditions

4 Habitat conditions were assessed using a national Rapid Habitat Assessment (RHA) protocol that assessed nine specific habitat parameters such as: deposited sediment, invertebrate habitat, fish cover, flow variability, bank vegetation and shade, and channel alteration. Drain Water Quality drains had the lowest RHA scores, followed by MEV streams. These scores reflected the high amounts of fine sediment in the drains and land drainage canals, as well as channel alteration, lack of bank vegetation and subsequent lack of shade. Significant relationships also existed between measured RHA scores and biological indices that assessed ecological health.

Water quality

5 Monthly water samples were collected for 17 months. Overall, water quality in the drains and land drainage canals is best described as poor, having very high nutrient levels, high turbidity and extreme levels of dissolved oxygen - both high and low. High Dissolved Oxygen (DO) levels reflect the large amount of primary productivity in these drains, while low DO levels most likely reflect a high biological or chemical oxygen demand in drain sediments from extensive accumulations of organic matter. Discharges from a monitored DWQ drain had demonstrable impacts on some water quality parameters (especially dissolved oxygen and ammonia (NH3-N)) in a natural stream. It is likely that other discharges from both DWQ drains and MEV streams may also have an effect on natural receiving environments, given that many of these waterbodies discharge directly or indirectly into estuaries or rivers with high cultural, recreational or ecological values. It is thus important to determine the fate of contaminants such as nutrients, *E. coli* and sediments from the drainage network on these receiving environments. Calculated catchment loads from the monitored drains were generally very high for parameters such as ammonia, despite the relatively small size of these drains. Such high catchment loads reflect the intimate contact of the drainage network to the surrounding agricultural land, and the high intensity of the land use (mostly dairy farming, but including kiwifruit orchards and urban/industrial areas).

Invertebrate communities

6 The invertebrate communities in the drainage network were numerically dominated by the common mudsnail (*Potamopyrgys)*, chironomid midges, and the shrimp-like crustacea, (*Tanaidicea)*. Other common invertebrates included the purse-case caddisfly (*Oxyethira)*, the water boatman (*Sigara)*, worms, leeches and flatworms, and ostracods. Calculated biotic indices in DWQ drains and MEV streams were often lower than in other waterway types. For example, the average MCI score for both the DWQ and MEV sites was only 74, well below the MCI score of 80 set as a lower threshold for Regional Council response in the NPS-FM. Although sites classified as DWQ are not fresh water bodies under the NPS-FM and RNRP definitions, the finding that MCI scores in MEV sites were below this threshold will require response, especially given the community's stated desire for better habitat and water quality in the drainage network.

7 Invertebrate communities differed between the different water quality classifications, with communities in DWQ and MEV sites being distinct to communities from other classifications types. The main structuring variable responsible for observed patterns to the invertebrate data appeared to be linked to the high ammonia-N concentrations characteristic in the DWQ and MEV sites, presumably reflecting the intensive land use in their surrounding catchments.

Fish communities

8 A total of 18 fish species were found in the Kaituna and Rangitaiki Plains. The most widespread were shortfin eels, inanga, and mosquito fish, found at more than 50% of sites. Species richness was significantly different in streams when classified according to their water quality class, and was equally lowest in the DWQ drains and MEV streams, and significantly higher in the waterways of all other water quality classes. Calculated Fish_IBI scores were also lowest in DWQ drains, intermediate in MEV streams and highest in streams classified as Other[2]. Most of the DWQ and MEV streams had fish IBI scores characterised as poor, although two sites were assessed as being in moderate or good condition.

9 Some of the sampled DWQ drains and MEV streams supported very large quantities of shortfin eels, despite poor habitat and water quality conditions, and presence of pump stations below some sites. Pump stations are likely to have major detrimental effects on downstream migrating behaviour of eels, as large migratory eels will almost certainly be killed when they pass through pump stations. It may be possible to minimise eel mortality in the drainage network and at pump stations by use of screens and traps.

10 Another potential stress on fish communities in the drainage network is related to macrophyte removal. Although BOPRC and private drain network operators remove excess macrophyte biomass by a variety of means (spraying, mechanical removal, and use of a plant cutter boat), the effect of these different techniques on fish has not been quantified. Targeted monitoring could be undertaken to ascertain whether these different methods pose risks to native fish. Grass carp (*Ctenopharyngodonidella*) are also used to manage macrophyte growth in some drains in the Rangitaiki Plains, while minimising adverse effects associated with mechanical or chemical control. These fish could also be used in other drains in the region, although the potentially low DO levels characteristic of many of the drains may restrict where these fish could be released.

Conclusions

11 The drainage network in the Kaituna and Rangitaiki Plains is characterised by very poor instream physical habitat conditions, with high amounts of fine sediment, a large degree of channel alteration, lack of bank vegetation and riparian shade. Lack of riparian vegetation in particular has major effects on both physical habitat and water quality, and results in excessive primary production (usually in the form of macrophytes), warm temperatures, high bacterial respiration and low DO levels. Excessive growth of aquatic plants is regarded as one of the major ecological stressors in the drainage network. Other features of many drains are the presence of either floodgates or pump stations in their lowermost reaches. These structures will also affect water quality conditions above them, and the ability of fish to successfully migrate up and down these waterways.

12 It may be possible to implement specific interventions to help alleviate some of the dominant stressors found within the drainage network. Key aims of mitigation would be to increase stream shade and reduce nutrient and sediment levels in the drains. These steps are likely to reduce macrophyte growth, and potentially cause less extreme dissolved oxygen levels. Such steps can be achieved by a combination of fencing, riparian planting, and the use of constructed wetlands at the mouths of drains, or linear wetlands within the drainage channel.

13 Other mitigation could include irrigation of the land with drain water (i.e., water recycling) during times when irrigation is needed, use of floating wetlands, and potentially even growing cash crops such as watercress in the drains to help shade the drains and take up nutrients.

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[2] The RNRP also has waterways classified as Contact Recreation, Aquatic Ecosystem, Fish spawning purposes, and Regional Base line. Because the aim of this work was to describe the ecological health of DWQ drains and MEV streams to all other waterways in the lowland plains, all these waterways were grouped as "Other".

14 Other mitigation measures would be to install fishfriendly floodgates, and to help develop alternative ways for migrating fish to pass through pump stations. It may also be possible to design and implement a trap and transfer system in the drainage network that could be used during the autumn when downstream eel migration is at its peak.

15 A large amount of targeted research is needed before any of these mitigation measures can successfully be employed. Some of this research is discussed with the hope that if these research programmes are implemented, then they may provide us with greater certainty as to what the effects of our current activities on the ecology of the drainage network is, and furthermore lead to ways that any adverse effects can indeed be minimised.

Contents

Part 1:

Introduction

1.1 An overview of drains

Many agricultural areas in New Zealand and throughout the world rely on extensive networks of drainage ditches to remove and control excessive surface water, and lower water tables (e.g., Hudson and Harding 2004, Needleman et al, 2007). A large variety of agricultural drains exist worldwide, including naturally meandering streams which have been artificially straightened and deepened, artificial channels cut into wetlands to drain and develop these areas into productive farmland, small lateral surface drains in developed pasture to help remove excess surface water, and subsurface tile drains to convey excess subsurface water from pasture. Many farm drains are fenced right to their edges to maximise the area of arable grazing land, and often have little or no riparian vegetation. They are consequently fully exposed to the sun, which often results in excessive algal or macrophyte growth. Because of their low elevation and gradient, pump stations are often positioned along drainage networks, to assist with the efficient and rapid removal of excess water, especially during times of heavy rainfall. These can effectively isolate the drainage network from natural waterways; with potentially large impacts on the longitudinal movement of biota above these pump stations. Peaty soils are also very common in these areas, and as the water table is lowered from the efficient drainage network, the soils oxidise and shrink which further lowers drainage network. This in turn requires more use of pump stations to drain water from these low-lying areas.

Although the benefit to agricultural productivity of maintaining an extensive and efficient drainage network is undisputed, there have been detrimental effects to the ecological communities of animals and plants that once lived in these lowland areas. Habitat conditions of the original meandering streams have been greatly altered as a result of channelisation, and wetland communities lost following wetland drainage.

Hudson and Harding (2004) provide the first overview of drain management in New Zealand, including a summary of ecological conditions in drains. Their review highlighted that water quality in drains is often characterised by very high nutrients, bacterial loads, and sediment, reflecting the intimate contact of small drains to the surrounding highly productive landscape. Low dissolved oxygen and an increased range of water temperatures are also commonly observed, due to a lack of overhead cover. While these water quality conditions undoubtedly place stresses on the biota found within the drainage network, it is important to also recognise that many drains discharge into natural rivers or estuaries. Given that these receiving environments often display signs of sedimentation and nutrient enrichment, it is important to manage the input of these contaminants into such sensitive receiving environments. Hudson and Harding (2004) also highlighted the extensive cover of aquatic macrophytes which can occur in drains, especially with introduced species such as curly leaved pondweed (*Potamogeton*), swamp willow weed (*Polygonum*), Canadian pond weed (*Elodea*), oxygen weed (*Egeria* or *Lagarosiphon*), and hornwort (*Ceratophyllum*). These plants usually grow to such an extent that they impede the hydraulic efficiency of the drainage network, and so macrophyte removal is a normal part of maintenance activities.

Excess macrophyte growth can also block water pumps, and negatively affect fish and invertebrate communities by smothering habitat and altering water chemistry. This excessive growth reflects the high nutrient regime in drains, combined with slow velocities, soft substrates and lack of shade. Many regional councils throughout New Zealand thus spend considerable time and effort in controlling excess macrophyte growth by a combination of mechanical removal (e.g., by drag lines, diggers fitted with rakes, or weed cutting boats), or spraying with chemicals such as glyphosate.

Aquatic invertebrate communities in drains are usually characterised by low species richness indeed often only the most tolerant invertebrates such as worms, snails and shrimp are found (Hudson and Harding 2004). Invertebrate densities can also often be very high, particularly where nutrient and light levels result in extensive macrophyte growth which represents an important invertebrate habitat. For example, Marshall and Winterbourn (1979), recorded invertebrate densities up to 280,000 individuals per square metre in a small drain in Canterbury, and Ryder (1997) found densities of amphipods such as *Paracalliope* and *Paraleptamphopus* up to 129,000 individuals per square metre. Lowland drains often support high densities of freshwater crayfish (kōura), and shrimp such as *Paratya*, especially as these habitats are often all that remains of the original wetland habitat where these animals originally dwelt.

Fish communities in drains are typical of what would be expected in low land waterways. Within New Zealand, many native fish are migratory, and use agricultural drains for temporary habitat, refuge or spawning. Fish such as shortfin eel (*Anguilla australis*), inanga (*Galaxias maculatus*), banded and giant kōkopu (*Galaxias fasciatus* and *G. argenteus*) are commonly found in drains, as well as more rare species such as mud fish (*Neochanna* spp) in some regions. In addition to pressures associated with macrophyte cleaning, access to and from the drainage network is often blocked by pump stations or tidal floodgates. This has particular relevance when considering the majority of native fish living within drains are migratory. An unintended consequence of pump stations is that downstream, migrating fish may be drawn into the pumps with potentially fatal consequences. This is particularly the case for large species such as shortfin and longfin eels, which only migrate to sea as large adults. Any loss through pump stations may have implications for overall population dynamics of these species.

Although drainage networks play important roles in maintaining productive agricultural landscapes, they also represent what may be the last remaining aquatic environments in these lowland areas and often have replaced what was previously wetland habitat. They may thus represent important habitats for a wide range of animals and plants – or at least have the potential to. A fundamental challenge therefore exists between the need to manage the drainage network to meet its primary function to convey excess water from agricultural areas, the need to recognise and maintain any ecological functions that this network supports, and to minimise the impact discharges from the network have on receiving environments. It is likely that there are many management techniques that can be employed at different spatial scales to help achieve all three goals (drainage, ecology and water quality).

1.2 The drainage network of the Kaituna and Rangitaiki Plains

The Kaituna (including the neighbouring catchment of Waihi Estuary) and Rangitaiki Plains are typical of lowland areas throughout New Zealand. Prior to European settlement, these areas were dominated by a system of wetlands, meandering streams and larger rivers. These waterways would have supported a unique assemblage of birds, fish, invertebrates, and plants that had adapted to these wetland environments.

They were most likely highly productive areas, and would have been valued by Māori for multiple values including:

- Food, from fish such as inanga, tuna, and kokopu, and invertebrates such as kākahi and koura.
- Fibre, from plants such as harakeke (flax).
- Medicinal plants such as harakeke (flax), mānuka (tea-tree) and patete (seven finger).

Following European settlement, wetlands throughout New Zealand were seen as an impediment to agricultural development, and so early settlers commenced an ambitious program to drain these waterlogged areas. For example, drainage of the Rangitaiki Plains for agricultural development began in the early 1890s, and the first Rangitaiki Drainage Board made two unsuccessful attempts at drainage in 1894, and 1910. However, the task of draining such a large area was beyond the scope of this relatively small organisation, and in 1910, Central Government took over responsibility for draining and developing the Rangitaiki Plains.

Early modifications to the natural wetlands within the Rangitaiki Plains included:

- Diverting water from the old Rangitaiki Channel and Orini Channel directly to the sea at Thornton.
- Straightening and dredging the Tarawera River, and diverting this from Matata Lagoon direct to the sea.
- Construction of a large network of canals, including the Te Rahu, Kopeopeo, Awaiti – Omeheu, and Awakaponga Canals in the 1910s and 1920s.

Similar schemes were run in the Kaituna and nearby 'Little Waihi' Catchments, whereby rivers such as the Pongakawa, Wharere, Pukehina, Raparapahoe, Ohineangaanga and Kopuaroa were channelised and straightened, and constrained within stopbanks to minimise flooding. Small lateral drains were also dug throughout the Kaituna Plains to help drain the water logged soil and allow agricultural development to occur. These modifications have had the effect of converting lowland swamps into a series of highly modified stream channels of significant length. The Kaituna Plains have an approximate area of 175 km^2, below the 20 m contour, through which approximately 270 km of waterways flow. Of this, only about 20 km is represented by the Kaituna River, which has been straightened in its lower reaches and confined between large stopbanks to minimise flooding, and whose flows from Lake Rotoiti are regulated by control gates. The other 250 km of waterways on the Kaituna Plains (93%), are represented by highly modified and straightened smaller waterways and drains.

A similar situation exists in the Rangitāiki Plains, which occupies an area of 335 km^2. A total of 513 km of waterways flow through these Plains, of which only 81 km are natural rivers – the mainstem of Rangitāiki, Tarawera, and Whakatāne rivers. However, even these natural rivers have been extensively modified; stop banks constrain the river to minimise potential flood damage to surrounding properties, parts of the Whakatāne River have been straightened, and the outlet of the lower Rangitāiki has been cut through to the coast at Thornton, as opposed to flowing to the sea via the old Rangitāiki Channel to the west, and Orini Canal to the east. The other 430 km of waterway length in the Rangitāiki Plains (83%) are represented by the large and small man made drains. These include many large named drains such as Reids Central Canal, Western Drain, Kopeopeo Canal, Te Rahu Canal, Orini Canal, Omeheu Canal, Awaiti Canal, Seacombes Canal, Old Rangitāiki Channel, as well as many smaller unnamed lateral drains that feed into these larger drains.

Bay of Plenty Regional Council runs and maintains four pumping schemes throughout the Kaituna and Rangitaiki Plains, each of which contain a number of different pumps and pump stations. These pumping schemes are responsible for managing water levels from large areas; for example, a total of 7650 ha of land is drained by pumps and pump stations in the Rangitaiki Plains (Table 1).

There are also private schemes and assets, such as the Waihi Drainage Society (under-written by Western Bay of Plenty District Council), which owns and manages 13 pump stations on the tributaries of the Waihi Estuary including the Kaikokopu, Wharere, Pongakawa and Pukehina. Many private farm drains also exist in both the Rangitaiki and Kaituna Plains, and these are operated by landowners.

Table 1 The number of pumps and pump stations found within different drainage schemes in the Kaituna and Rangitaiki Plains.

Scheme name	Pumps	Pumpstation	Grand total
Kaituna	8	9	17
Waihi Drainage Society	13		13
Rangitaiki Drainage	36		36
Rangitaiki Tarawera	2	2	4
Whakatane Waimana	4	3	7

Although this historic drainage work, channelisation and maintenance of an efficient network of drains and pumping stations has had huge benefits to agricultural productivity, there have also been detrimental effects to the natural animal and plant communities that once lived in these areas. For example, all that is left of the original extensive wetland environments in the Kaituna Plains are straightened, channelised and deepened natural waterways such as the Pongokawa, Puanene, Wharere and Kaikokopu streams. Within the Rangitaiki Plains, many drainage canals have been dug that flow through very low gradient areas and discharge into natural waterways through a series of flood gates or pump stations.

1.3 This study

The Resource Management Act (1991), has devolved substantial functions and responsibilities for the sustainable management of each region's natural and physical resources to regional and district councils (RMA, s.30 and 31 in particular). Of particular relevance for this report, section 35 requires regional councils to monitor the state of the environment (section 35 (2)(a)), as well as monitoring the effectiveness and efficiency of policies, rules, or other methods in their policy statements or plans (section 35 (2)(b)). The BOPRC operates a Natural Environment Regional Monitoring Network (NERMN) programme that monitors, amongst other things, water quality and invertebrate communities from selected waterways throughout the region (Donald 2012).

The vast majority of water quality and ecological monitoring done by BOPRC is from natural waterways, with little focus on DWQ drains or channelised MEV streams. This represents a gap in Council’s ability to describe the current state of waterways in the Kaituna and Rangitaiki plains.These drains and land drainage canals are currently managed with the main focus on maintaining their drainage values, and activities such as macrophyte removal and channel dredging commonly occur. Many of these drains are often below sea level, and so a network of floodgates is needed to keep sea water out at high tides, and to pump water from the drains into the receiving environment. It is possible that these management strategies have had adverse ecological effects within the Rangitaiki and Kaituna Plains. However, in the absence of detailed information, it is difficult to make specific recommendations to improve ecological outcomes without compromising drainage functions. Such outcomes are particularly relevant in terms of the National Policy Statement for Freshwater Management (NPS-FM), which clearly articulates that freshwater bodies need to be managed to safeguard ecological health and human health for contact recreation, and to support tangata whenua and other values.

This study was commenced in November 2015 to address these knowledge gaps. This study had three aims:

1 To improve our state of knowledge about the habitat, water quality and ecological conditions of drains and land drainage canals throughout the Kaituna and Rangitāiki Plains. This may provide valuable information for the community consultation process currently being undertaken by BOPRC as required under the National Policy Statement for Freshwater Management (NPS-FM).

2 To highlight potential hotspots of contaminants or areas of low dissolved oxygen in drains, which ultimately discharge into natural water courses. Some of this information will have implications for the management of sensitive downstream receiving environments (e.g., the Maketu and Waihi estuaries).

3 To provide good baseline information on water quality and ecological conditions to help improve the future management of drains. This information will be useful to guide operational activities in the drainage systems, and to inform environmental assessments of the effects of the drain discharges into other waterways.

This information is reported here and is followed by a discussion of potential ways to mitigate the adverse effects of the current conditions in the drainage network.

1.4 Site selection

Fish and Game commenced a study of drains throughout the Kaituna and Rangitaiki Plains in the austral summer of 2015, to determine factors influencing the presence or absence of waterfowl, and the number of chicks that they produce. They surveyed approximately 80 drains, 40 of which supported birds, and 40 which did not. A subset of these 80 drains were thus randomly selected for the current study to provide further information to Fish and Game for their work, and to obtain new information about water quality, invertebrate and fish communities in the drains. A total of six drains were subsequently monitored in the Kaituna Plains (Figure 1), and 14 drains in the Rangitāiki Plains (Figure 2). These 20 drains were the focus of detailed investigations of habitat condition, water quality and invertebrate communities. Fish communities were also sampled from six of these drains (Table 2), the other drains were not sampled as they were either too small, had excessive macrophyte growth, or because of floods in early April 2017 from cyclones Debbie and Cook.

Data from other sources was also accessed to further characterise and describe the state of the drainage network in the Kaituna and Rangitaiki Plains, and to put the observed data from these drains into context with other waterways throughout the region. These sources included the Council's NERMN monitoring network, data from consent applications, and data from other Council monitoring programmes such as a large-scale survey of the Rangitāiki Catchment (Suren 2013), and surveys initiated to fill knowledge gaps previously identified in both the Kaituna-Pongakawa-Waitahanui (Suren et al 2015) and Rangitāiki WMAs (Carter et al. 2015). This gave us a total of 385 sites for comparison of habitat data, 168 sites for comparisons of invertebrate communities, and 68 sites for comparison of water quality. Further information on fish communities in 47 sites from lowland areas (< 20 m ASL) in the Kaituna and Rangitāiki Plains was obtained from the New Zealand Freshwater Fish Database (NZFFD). The fish dataset was restricted to this lowland area only as fish community composition is strongly related to distance inland and elevation, so sites further inland or at higher elevations would be expected to have different fish communities.

Under the Regional Natural Resources Plan (RNRP, containing the former Regional Water and Land Plan), waterways have been assigned to one of nine water quality classifications, based on expert opinion of Council staff at the time. Each of the sites sampled in the Kaituna and Rangitāiki Plains were consequently assigned to their appropriate water quality classification, based on the water quality maps in the RNRP.

Two classifications were particularly relevant for this study, drain water quality (DWQ) and modified waterways with ecosystem values (MEV) which applies to land drainage canals identified in the RNRP (Schedule 3 and Definitions). These classifications are particularly relevant as waterways classified as MEV are defined as freshwater bodies under the NPS-FM and RNRP, and need to have freshwater objectives set as part of the NPS-FM process, and are subject to the attribute tables listed in Appendix 2 of the NPS-FM. DWQ sites are defined as artificial water courses (not freshwater bodies), so the Appendix 2 bands are not applicable. However, discharges from DWQ drains will be managed to support receiving environment objectives (rivers and estuaries) and it is likely drain discharge quality will need to improve. All other waterways throughout the region were also assigned to their appropriate classification in the RNRP to compare conditions in the lowland drainage network sampled for this study.

Table 2 *List of the 20 main drain sites in the Rangitāiki and Kaituna Plains sampled for this study, showing their location (in NZTM), water quality classification (DWQ = Drain Water Quality; MEV = Modified watercourse with Ecological Values), and whether habitat, water quality, invertebrates and fish were collected (Y). Fish surveys were unable to be conducted in all drains as a result of: 1 = flooding; 2 = being too small; 3 = having excess macrophyte growths which precluded deployment of the fyke nets.*

Site identification	Water management area	Site name	Easting	Northing	WQ class	Habitat	Water quality	Invertebrates	Fish
BOP_DRAIN_02	Kaituna_Maketu	Bell Road Drain at Te Puke	1894698	5817897	DWQ	Y	Y	Y	1
BOP_DRAIN_03	Kaituna_Maketu	Kaituna Drain at Pah Road	1896735	5815673	DWQ	Y	Y	Y	1
BOP_DRAIN_04	Kaituna_Maketu	Kaituna Drain at Kaituna Road	1902169	5814595	DWQ	Y	Y	Y	1
BOP_DRAIN_05	Kaituna_Maketu	Wharere Drain at Pukehina	1906387	5812911	MEV	Y	Y	Y	1
BOP_DRAIN_07	Kaituna_Maketu	Pukehina Drain at Pukehina	1907565	5812877	MEV	Y	Y	Y	1
BOP_DRAIN_08	Kaituna_Maketu	Pongakawa Drain at Cutwater Road	1906928	5813003	MEV		Y	Y	1
BOP_DRAIN_14	Rangitaiki	Reids Central Canal	1938326	5792045	MEV		Y	Y	1
BOP_DRAIN_15	Rangitaiki	Western Drain	1938530	5788430	MEV	Y	Y	Y	2
BOP_DRAIN_09	Tarawera	Awakaponga Canal	1930945	5797125	MEV	Y	Y	Y	3
BOP_DRAIN_10	Tarawera	Section 109	1932924	5796827	DWQ	Y	Y	Y	3
BOP_DRAIN_11	Tarawera	Awaiti Canal	1933519	5794286	DWQ	Y	Y	Y	Y
BOP_DRAIN_12	Tarawera	Omehue Canal upstream WWTP	1934562	5789921	MEV	Y	Y	Y	3

Site identification	Water management area	Site name	Easting	Northing	WQ class	Habitat	Water quality	Invertebrates	Fish
BOP_DRAIN_13	Tarawera	Omehue Canal downstream WWTP	1935295	5791681	MEV	Y	Y	Y	Y
BOP_DRAIN_22	Tarawera	Secombes Canal at Greig Road	1934753	5797959	DWQ	Y	Y	Y	Y
BOP_DRAIN_16	Whakatane	Eastern Drain	1942045	5790393	DWQ	Y	Y	Y	Y
BOP_DRAIN_17a	Whakatane	Waioho Stream upstream of Drain_18	1948129	5788962	MEV	Y	Y	Y	1
BOP_DRAIN_17b	Whakatane	Waioho Stream downstream of Drain_18	1948129	5788962	MEV	Y	Y		1
BOP_DRAIN_18	Whakatane	Langenberger Road Drain	1947950	5788941	DWQ		Y	Y	Y
BOP_DRAIN_19	Whakatane	Te Rahu Canal	1947277	5790826	MEV	Y	Y	Y	Y
BOP_DRAIN_21	Whakatane	Orini Canal off Thornton Road	1944957	5794132	MEV	Y	Y	Y	3

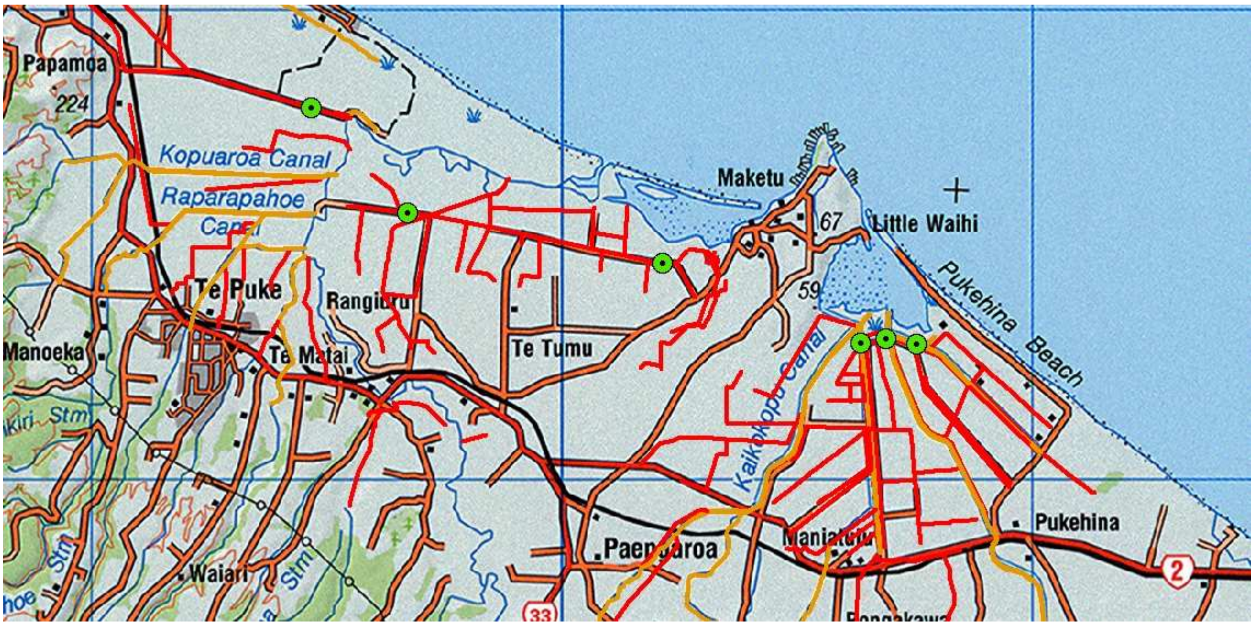

Figure 1 *Location of the six drain sites in the Kaituna Plains monitored for this study. Red lines = DWQ sites waterways; Orange lines = MEV waterways.*

Figure 2 *Location of the 14 drain sites in the Rangitāiki Plains monitored for this study. Red lines = DWQ sites waterways; Orange lines = MEV waterways.*

Part 2:

Physical habitat conditions

2.1 Introduction

The physical habitat condition of a waterway represents the living space for all aquatic plants and animals. Aquatic ecosystems are affected by both instream features, and those of the immediate streamside (or riparian) areas (Harding et al., 2009). Important habitat includes substrate size (e.g., boulders, cobbles, sand or mud) and the type of flow (e.g., fast flowing riffles, slow flowing runs or deep pools). Substrate size is of fundamental importance in influencing invertebrate and plant communities, as this is where these organisms mostly live. Many invertebrates such as caddisflies and stoneflies are thus found mainly in gravel bed streams and are generally intolerant of fine sediments. In contrast, midges and worms can easily borrow amongst fine sediments such as sand and silt, and snails can easily crawl over these fine materials. Substrate size is also directly related to factors such as water velocity and substrate stability - both of which exert large effects on ecological communities (Minshall 1984, Biggs et al. 2001).

Many native fish also live amongst the substrate[3] in contrast to swimming fish such as trout, and as such have strong preferences for different substrate sizes. For example, torrent fish and redfin bully have preferences for coarse substrates, while juvenile shortfin eels or lamprey have preferences for fine sandy substrates (Jowett and Richardson 1995). Variation in water velocity and depth is also important to stream ecology, and results in creation of different habitats such as riffles, runs and pools (Jowett 1993). These small-scale hydraulic differences can have profound effects on stream ecology (Pridmore and Roper 1985, Jowett and Richardson 1990), with, for example, fast flowing riffles often being far more productive in terms of and algal and invertebrate biomass than slow flowing runs and pools.

The nature of the streambank is also highly important, particularly for fish which often seek shelter within undercut banks. Banks which are continually eroding are also responsible for introducing large quantities of fine sediments into streams, which may have detrimental effects to downstream habitats if the streambed becomes smothered, and interstitial spaces are filled with this fine material.

Riparian vegetation is also vitally important to streams, as it provides both shade to the stream and can add to bank stability. Indeed, one of the issues identified in the RNRP is the lack of suitable riparian vegetation to stabilise the margins of surface water bodies and filter surface run off (Issue LM 12). The RNRP therefore emphasises the importance of managing riparian areas to minimise the adverse effects of land use activities. Our conceptual understanding of the benefits of riparian vegetation to stream ecosystems (e.g., Collier et al. 1995, Parkyn 2004) include:

- Reducing water temperature.
- Assisting with bank stability.
- Preventing excess algal or macrophyte growths.

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[3] The term "substrate" refers to the bottom or bed of the waterway. The term "streambed" was not appropriate here, as DWQ drains are not defined as streams.

- Providing shelter for fish and invertebrates.
- Providing spawning habitats for fish.
- Adding organic matter to streams to serve as a food source for invertebrates.

The ability of overhead vegetation to shade drains is particularly important in minimising excessive growth of algae or macrophytes.

Excess macrophyte growth is responsible for reducing the hydraulic efficiency of drains, so a large proportion of drain maintenance involves removing this material. Although macrophytes represent important habitat for both fish and invertebrates, excess growth can also have negative impacts on these animals through loss of habitat and can cause large variations in dissolved oxygen at night when these plants are respiring.

Many lowland waterways such as the Kaikokopu, Pongakawa, Pukehina and Wharere have also been extensively straightened by channelisation. Stopbanks have also been built along both sides of these and other waterways to contain flood waters. These stop-banked areas are also often grazed and kept clear of any overhanging vegetation. These activities have resulted in streams with uniform channel cross section profiles and very little overhanging vegetation shade, thus lowering habitat complexity. Furthermore, natural streams have meandering flow patterns, and intimate contact with the surrounding land. Channelised streams in contrast have lost habitat such as slow flowing backwater eddies and wetlands, as well as their connections with inflowing streams. Bank reinforcements and channel modifications have also often resulted in a relatively uniform channel cross section profile, thus lowering habitat complexity.

The current habitat conditions in the lowland drainage network are consequently a result of management focused primarily at removing excess surface and groundwater from the agricultural landscape. The object of this section of the report was to characterise habitat conditions in a number of drains throughout the Kaituna and Rangitaiki Plains, and to compare these habitat conditions with those from other waterways throughout the region. The secondary objective was to determine whether habitat quality was influencing the ecological health in the drains and land drainage canals surveyed.

2.2 Methods

Habitat conditions of each drain and land drainage canal were assessed using an early version of a national rapid habitat assessment protocol (RHA) developed by (Clapcott 2015). This standardised protocol assesses nine specific habitat parameters which are known to affect both invertebrate and fish communities in streams:

- Deposited sediment.
- Invertebrate habitat.
- Fish cover.
- Hydraulic heterogeneity.
- Bank stability.
- Bank vegetation.
- Riparian width.
- Riparian shade.
- Channel alteration.

Each of these parameters was scored from one to 20, with 20 indicating optimal conditions. For example, a stream where deposited sediment covers more than 75% of the streambed would score one, whereas a stream with no deposited sediment would score a 20. The final RHA score is based on the sum of each score, with scores ranging from 20 to 180.

Rapid Habitat Assessment scores were assessed for all the waterways examined in the Kaituna and Rangitaiki Plains. These waterways were then assigned to the relevant water quality classification as used in the RNRP. Calculated RHA scores were compared to other RHA scores collected throughout the region as part of the Council's ongoing NERMN monitoring programme, other Council sampling programmes, as well as data collected as part of consent applications. This combined dataset gave us a total of 385 sites for comparison of habitat data. This analysis was done using general linear models (GLM) in Systat 11.0, with the relevant water quality classification used as the grouping variable. A stepwise discriminant function analysis (DFA) was then used to determine which of the nine habitat parameters best discriminated between the different water quality classifications. This analysis also predicted class membership of each site, based on the values for the selected habitat parameters. The strength of these predictions was then assessed.

A Principal Components Analysis (PCA) was then used to reveal any structure in the habitat data based on the individual scores for each habitat component, and to identify what the major habitat differences were between sites. Because all habitat scores were based on a relative score from 1 to 20, it was not necessary to standardise or normalise the data. A two-dimensional plot was produced that showed the relative location of each site based on their individual habitat scores. The PCA also highlighted what the major habitat gradients were in the data. Histograms were also made showing the percentage of sites with different rankings of each habitat variable, so a visual representation could be shown of habitat differences between DWQ, MEV and other waterways.

Finally, relationships between invertebrate communities and the total RHA score were assessed using regression analysis. Invertebrate communities were summarised by five biotic indices, including the macroinvertebrate community index (MCI), the quantitative MCI (QMCI), the number of mayfly stoneflies and caddis fly (EPT) taxa, the percentage of EPT to total taxon richness, and the percentage of EPT to total abundance[4]. Where significant regressions occurred, a backward stepwise regression was run to determine which of the nine habitat variables that contributed to the total RHA score were influencing the invertebrate communities.

2.3 Results

Habitat quality in the 55 low land waterways surveyed in the Kaituna and Rangitaiki Plains varied greatly, with RHA scores from 32 to 149. Most sites had RHA scores in the 90 to 110 range (Figure 3). Significant differences in RHA scores were found between sites of the different water quality classes. With the exception of a single Aquatic Ecosystem (AE) site, sites classified as DWQ had the lowest RHA scores, followed by those classified as MEV. All the other waterway classes (Aquatic Ecosystem, Contact Recreation (CR), Fish Spawning Purposes (FSP), Natural State (NS), and Regional Baseline (RBL)) had similar and higher RHA scores. Examination of the frequency of different RHA scores showed clearly that the DWQ and MEV water courses represented some of the lowest habitat conditions found in the 385 water courses surveyed (Figure 4).

[4] *Note that the caddisfly Oxyethira was excluded from the list of EPT as it is highly tolerant of degraded conditions and can tolerate warm temperatures and algal blooms.*

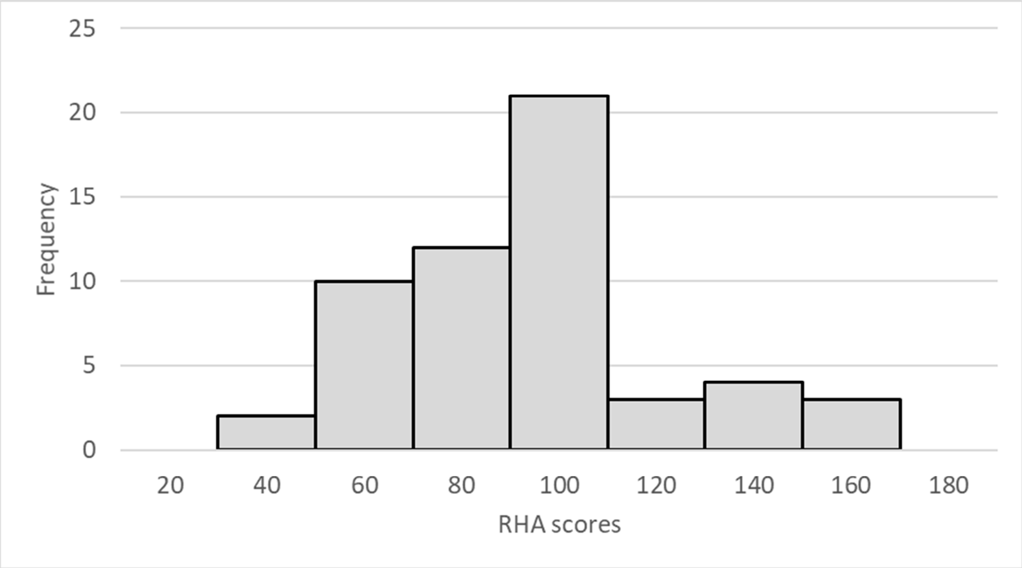

Figure 3 *Histogram showing the number of sites in the low land drainage network in the Kaituna and Rangitāiki Plains with a range of RHA scores.*

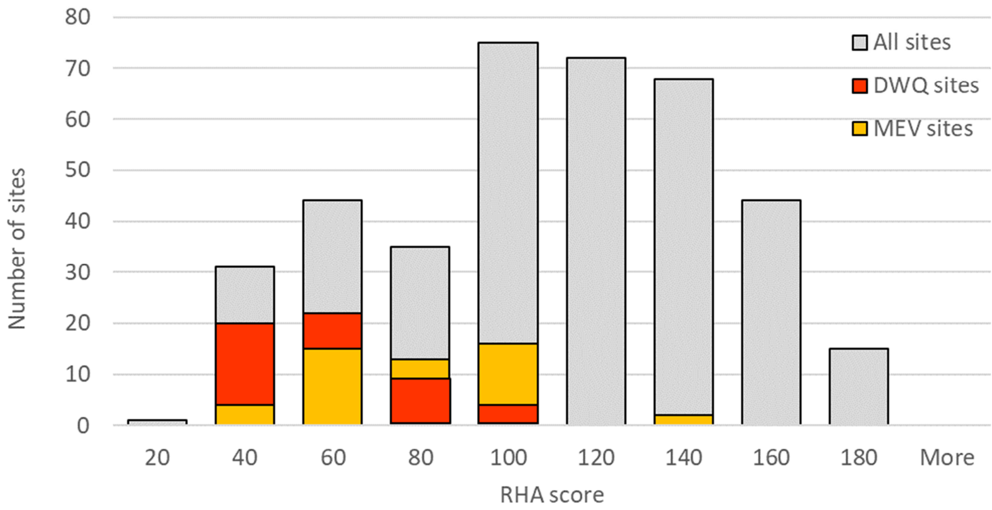

Figure 4 *Histogram showing the number of sites with a range of RHA scores from all 385 sites throughout the region where RHA assessments have been done. Also shown is the number of sites from DWQ and MEV classes for comparative purposes.*

Discriminant function analysis showed that the main habitat factors that discriminated amongst the different stream classes included deposited sediment, bank vegetation, riparian shade and the degree of channel alteration. This is not surprising when considering the fact that most of the drain sites sampled were highly channelised (Figure 5), had riparian conditions dominated by short vegetation and were generally exposed to the sun. Their substrates were also more often than not characterised by deep mud overlain by a thick organic layer of decaying material, including macrophytes, and their flow heterogeneity was also very low. All these factors scored very low for the DWQ sites monitored (Figure 6) and emphasised that these factors differed most between waterways assessed as DWQ and other types of waterways. This was not surprising however, especially when considering that DWQ drains were never built for the purpose of ecosystems/ecosystem health.

Eastern Drain

Awaiti canal

Figure 5 *Examples of the degree of habitat modifications characteristic of many of the drains showing their straightened nature, lack of overhead shade, and often dense macrophyte growth that narrowed the channel.*

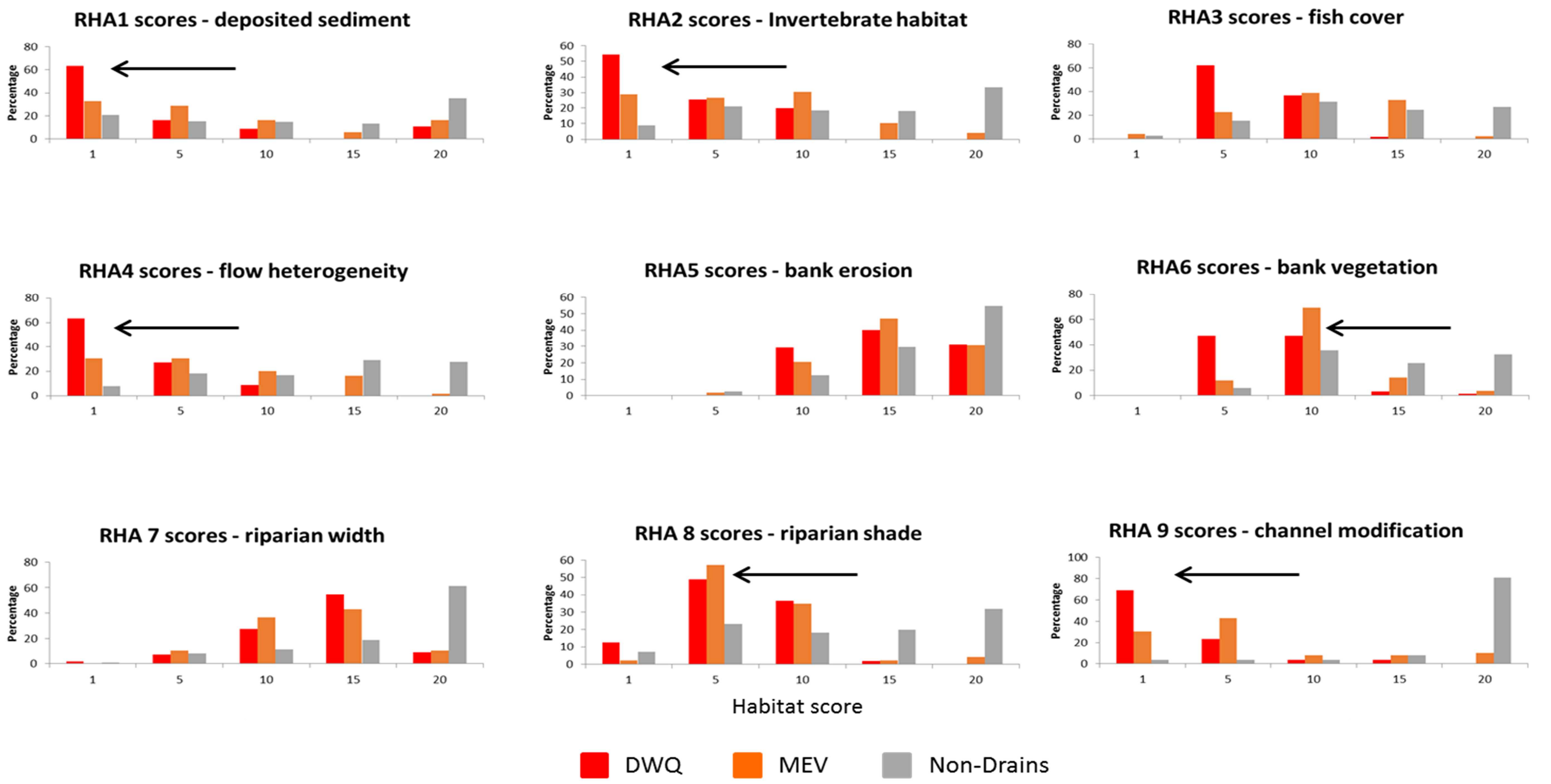

Figure 6 *Histograms of the individual RHA habitat scores for each of the nine habitat components assessed, showing the frequency (as a percentage of the total sites) of scores in each of the three water quality class groups; DWQ = drain water quality; MEV = modified water courses with ecological values; non-drains = a mix of Aquatic ecosystem, Regional baseline, Natural state, Contact recreation, Drinking water supply, and Fish Spawning Purposes. Arrows highlight habitat components in DWQ and MEV stream that displayed qualitative differences in their scores than in non-drain sites.*

Results of the PCA confirmed that the DWQ and MEV sites had different habitat conditions than the other waterways (Figure 7), as these sites were generally clustered to the upper right of the PCA plot. Sites from other waterway classes however were generally found with lower PC1 scores.

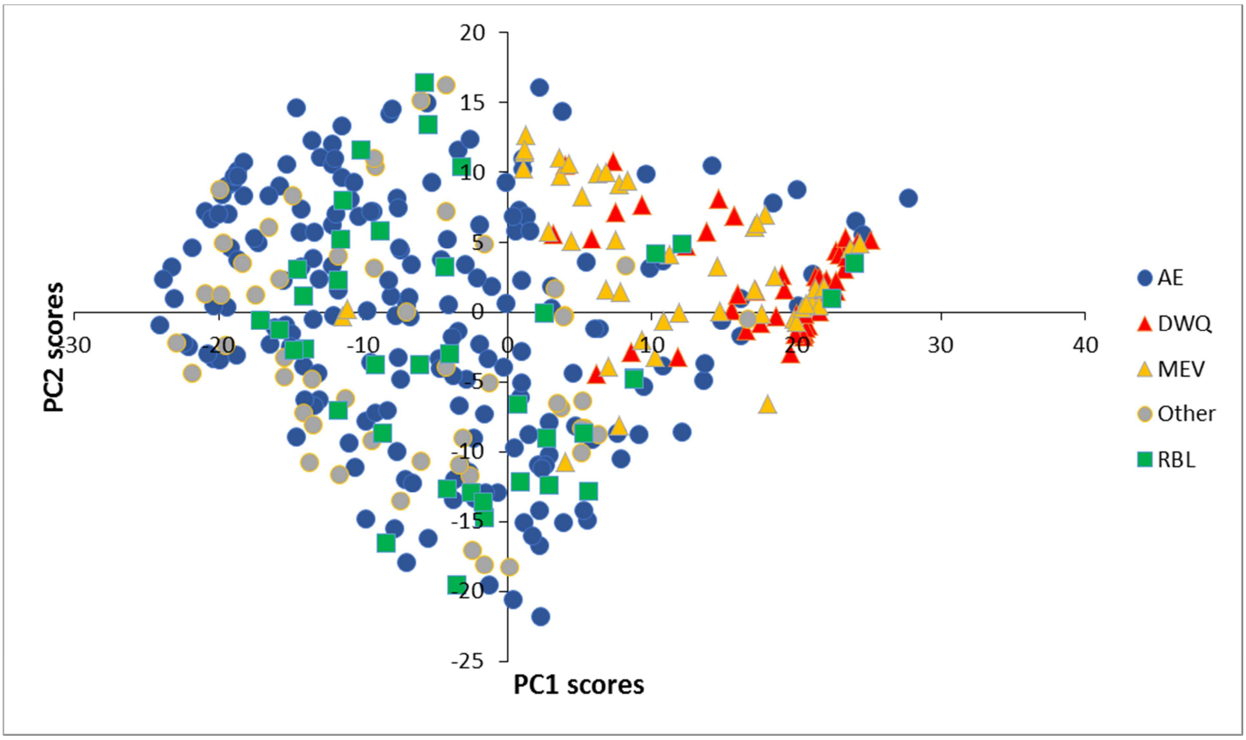

Figure 7 *Results of the PCA analysis of all habitat data from the 385 waterways, coded by their water quality classification. Note the strong clustering of DWQ and MEV sites to the right upper quadrat of the diagram.*

Regression analysis showed significant relationships between total RHA score and all five biological indices (Figure 8). In all cases, samples collected from sites with a classification of DWQ or MEV generally had lower RHA scores and lower values for each of the five indices. This highlights the nature of both habitat condition and ecological communities at these sites when compared to natural stream sites throughout the region.

A stepwise multiple regression analysis was done to determine which of the nine factors assessed for the RHA had the greatest influence on the biological indices. Important factors which were identified included the degree of channel modification and riparian buffer width which were found in all five regression models (Table 3). The degree of channel modification was also the strongest factor in each regression model. The assessment of invertebrate habitat, fish cover and hydraulic heterogeneity was found in four models, while assessments of deposited sediment, bank stability and bank vegetation were found in only two models. Riparian shade was identified as a significant explanatory variable for only one biological index, the MCI (Table 3, Table 6). This was an unexpected result, especially considering the importance of shade in helping maintain cool water temperatures and reducing in stream plant growth.

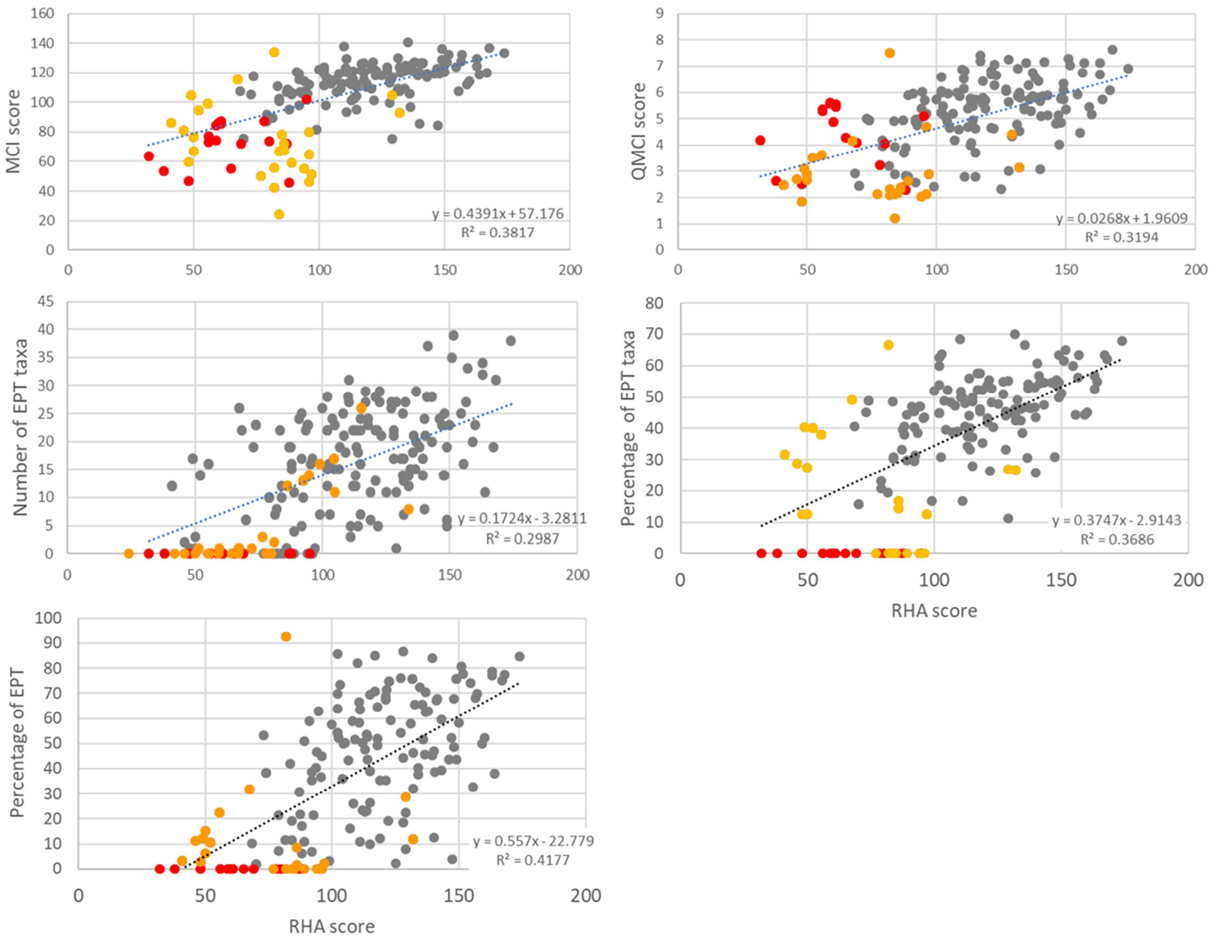

Figure 8 *Relationships between total RHA score and biological indices describing the invertebrate communities. Colour coding as per Figure 6.*

Table 3 *Results of stepwise regression models that were run to identify important independent habitat factors that were affecting various biotic indices. The table shows the regression coefficient for each factor (all with a significance value of P<005), as well as the overall percentage of the variance explained by the resultant regression models.*

Factor	MCI score	QMCI score	Number of EPT taxa	Percentage of EPT taxa	Percentage of EPT
1 Deposited sediment	-0.388	-0.028			
2 Invertebrate habitat	0.775	0.057		0.47	0.704
3 Fish cover	-0.422	-0.032	-0.211	-0.306	
4 Hydraulic heterogeneity	0.435		0.465	0.49	0.499
5 Bank stability			0.263		
6 Bank vegetation	0.413				0.855
7 Riparian buffer width	0.743	0.049	0.214	0.643	0.81
8 Riparian shade	0.292	0.049			
9 Channel alteration	1.43	0.074	0.661	1.269	1.557
Model R squared	0.577	0.417	0.418	0.512	0.469

2.4 Discussion

The key finding of this habitat assessment work was that drains and land drainage canals have poor to very poor habitat quality when compared to other waterways throughout the region. This however is not particularly surprising given the fact that many of the drains were constructed with a focus on maximising their hydraulic efficiency to remove excess water. This means that complex habitat features providing ecological benefits such as hydraulically rough streambeds and banks, meandering channels and overhead vegetation are generally absent from the drainage network as this was not their intended purpose. Factors that strongly discriminated the DWQ and MEV sites from other waterways included deposited sediment, bank vegetation, riparian shade and the degree of channel alteration. The high degree of deposited sediment in the drainage network presumably reflects the large accumulation of organic matter and their subsequent decay in systems that lack the high discharge high discharge water velocity to flush this material through the system. Lack of riparian vegetation and shade simply highlights the fact that the drainage network is present through productive agricultural land and is fenced right to the margins.

Results of the regression analysis show that the measured habitat scores were also related to a number of biological metrics that summarised the invertebrate communities (and therefore ecological health). A challenge thus exists to potentially find ways to improve habitat conditions and subsequent ecological communities, and yet maintain hydraulic functioning. Although some factors such as channelisation cannot easily be changed, others such as riparian buffer width and provision of overhead shade may be able to be improved. Indeed, recent research (e.g., Wilcock et al. 2013) has shown that riparian planting can have significant effects on improving water quality in highly productive dairy catchments, and so such planting is likely to have similar positive impacts on water quality in some of the drains.

Riparian planting may also have an additional benefit in providing shade to the drains, which may then reduce the extent of macrophyte proliferations. For example, an earlier report (Environment Bay of Plenty 1994), suggested that the suppression of excess macrophyte growth could be achieved by establishing riparian vegetation without undesirable side effects, providing appropriate tree species were used. This report also acknowledges numerous limitations as to the use of riparian vegetation to control macrophyte growth, as the effectiveness of such vegetation is largely dependent upon physical constraints such as the direction of the drain in relation to the sun (North-South running drains can be shaded better than East-West running drains), and the width of the drain (shallower drains can be shaded easier than wider drains).

Hydraulic heterogeneity also scored poorly in the drainage network, and this could possibly be improved by widening and narrowing parts of the channel. Pump station or flood gate arrangements could be investigated to restore flow.

For example, the old Rangitaiki Canal originally linked the Rangitaiki River to the Tarawera River. Flow in this canal is now blocked by a combination of floodgates in its lower reaches and stop banks and pump stations in its upper reaches. This has resulted in a highly nutrient rich, often stagnant water body (Chen 2014). Restoring flow to this canal and undertaking strategic riparian planting would likely result in significant positive ecological outcomes, both aquatic and terrestrial.

Part 3:

Water quality

3.1 Introduction

The essential function of drains is to prevent flooding of low land agricultural areas by efficiently removing surface water during times of high rainfall, and lowering the water table during and between rainfall events to prevent waterlogging of soils, and resultant stress to plant crops. Throughout both the Rangitaiki and Kaituna Plains, the drainage network is composed of small lateral ditches running through farmland which flow into larger DWQ drains and ultimately into natural waterways. Some of these small lateral drains are unfenced, while many others have only minimal setbacks along their margins and are often sprayed to remove all riparian vegetation (Figure 9). This lack of any riparian zone is likely to increase inputs of contaminants from the surrounding catchment. Many of these larger drains also discharge into larger natural rivers or estuaries, either naturally by gravity, or through a series of pump stations or floodgates. Because of their intimate contact with the surrounding highly productive agricultural landscape, water quality in both the small lateral drains and the larger drains they flow into is likely to be poor, and show high levels of nutrients, *E. coli*, and a wide range of other contaminants such as sediment.

Figure 9 Examples of two small lateral drains that are common throughout the Kaituna and Rangitaiki Plains. Such drains are either generally ephemeral (left) or perennial (right), and are often either unfenced (left), or fenced with minimal setback to maximise the area of arable or pasture land.

Drains often have high concentrations of total nitrogen, often to the point of being nitrogen saturated (Needleman et al. 2007), so that there is more nitrogen than can be used by plants. They also export considerable quantities of phosphorus, which usually enters drains through surface run off that carries phosphorus rich sediments or organic matter that can become deposited in the drainage ditch (Ballantine and Hughes 2012). Research has also shown that drains can be either a source or a sink for phosphorus, depending on the sediment equilibrium phosphorus concentration (EPC). Phosphorus will bind to sediments if streamwater concentration is greater than the sediment EPC, whereas sediments will release phosphorus if the stream water dissolved phosphorus concentration is less than the EPC, particularly under anoxic conditions (Ballantine and Hughes 2012). This means that sediments within the drainage network can act as large stores of phosphorus, which may be mobilised as part of drain maintenance activities such as macrophyte removal (Ballantine and Hughes 2012).

Sediment is also a significant water pollutant. Sediments can enter lateral drains from activities in the surrounding catchment such as stock grazing, or through activities associated with cropping. Sediment is also transported through the drainage network as a result of erosion and failure of banks. Sediments may undergo repeated cycles of deposition and resuspension (Needleman et al. 2007) as a result of stream maintenance activities, or high flows in the drain resulting from high rainfall.

Many drains are also characterised by extremely high primary productivity, reflecting a combination of high nutrient loads, slow flows and lack of shade. In particular, invasive aquatic macrophytes often accumulate to such an extent that they impede flows and decrease drainage efficiency. As a result, drains are often mechanically cleared to remove excess macrophyte growth. This can have detrimental effects on downstream water quality, with studies showing that drain cleaning can increase suspended solids and concentrations of some water quality variables such as phosphorus (Ballantine and Hughes 2012). Excess macrophyte growth will also result in accumulations of dead and decaying organic matter within drains, as macrophyte's either trap organic material, or as older parts of these plants naturally die. This large accumulation of organic matter can have large effects on oxygen concentrations in drains, resulting in large diurnal swings as a result of excess photosynthesis during the day and respiration during the night.

Changes in oxygen concentration in drains are also exacerbated by a lack of shade around most drains, causing high water temperatures. High water temperatures will increase plant and bacterial respiration, resulting in even lower oxygen levels. High water temperatures also mean that the oxygen is less able to dissolve amongst the water, further adding oxygen stress for animals such as fish. This is further compounded by the increased metabolism of fish at high temperatures.

It is clear that drain water quality is thus affected by both conditions in the surrounding agricultural landscape, as well as conditions within the drain itself, particularly in terms of changes to oxygen levels as a result of excessive macrophyte growth. However, it is also important to realise that the drainage network will also affect the ultimate receiving environments that they discharge into. Many drains flow into receiving waters such as lowland rivers or estuaries. The high nutrient and sediment loads in drains plus low oxygen levels may consequently affect water quality conditions in these receiving environments. Such discharges are covered by two rules in the BOPRC's RNRP:

- **Rule DW R3:** Where the discharge of water from existing farm drains and pumped drainage areas is permitted, as long as various criteria are met, including ensuring the discharge does not: produce scums or foams; change the colour of the receiving environment; emit an objectionable odour; render freshwater unsuitable for consumption by farm animals (126 FCU/100 ml); cause any more than minor adverse effects on aquatic life or cause persistent erosion. These criteria are to be met at a downstream distance of three times the width of the stream or river at the point of discharge.
- **Rule DW R8:** This rule applies if permitted activity rule DW3 R3 conditions cannot be met. Under DW R8 discharge of contaminants to water is a discretionary activity which will be assessed against (in addition to other objectives and policies) the Water Quality Classification Standards in Schedule 9. Where the standards cannot be met applicants for consent will have to demonstrate how adverse effects will be avoided, remedied or mitigated.

This section of the report summarises the results of an 18-month water quality monitoring program at 20 sites throughout the Kaituna and Rangitaiki Plains. There were eight objectives in this section. Firstly, water quality conditions of the drains were summarised to highlight differences between DWQ drains and MEV land drainage canals, in both the Rangitaiki and Kaituna Plains. Secondly, an assessment of seasonal patterns was done to see whether overall water quality varied seasonally. The third closely related objective examined relationships between drain water quality and rainfall, as it was expected that drain water quality would respond in a consistent manner to total rainfall prior to sampling.

The fourth objective was comparing the water quality signatures from DWQ and MEV sites to Appendix 2 of the NPS-FM that lists specific numerical bands for water quality attributes such as nitrate-nitrogen and ammonia (for toxicity), dissolved oxygen (below point source discharges), and bacterial contamination (*E. coli*). In this way, it was possible to put the water quality results into context with these national attributes, noting that only the MEV land drainage canals are subject to the objective setting processes.

Fifthly, changes in water quality above and below discharges from one of the many pump stations in operation throughout the plains was quantified. Schedule 9 of the RNRP identifies nine water quality classes and sets minimum standards against which discretionary discharges of contaminants to water will be assessed. For example, discharges into MEV waters shall not reduce dissolved oxygen concentrations and have no more than minor adverse effects on aquatic life. This objective was of particular interest as we wanted to determine the likely effect of potentially poor water quality being discharged from some drains into receiving waters.

The sixth objective compared water quality signatures of the DWQ and MEV sites to the nine current water quality classification standards in Schedule 9 of the RNRP. The seventh closely related objective then compared water quality signatures in the DWQ and MEV sites to water quality in six wetlands in the Rangitaiki Plains. This analysis was done as prior to land drainage activities, much of the Rangitaiki and Kaituna Plains were dominated by wetlands, so a comparison with these remnant ecosystems was regarded as providing useful context as to how the drains and land drainage canals differed from these original wetlands.

The last objective was to calculate average loads of various contaminants from the monitored drains, based on median concentrations of various attributes and on modelled flow data.

3.2 Methods

The eight objectives could conveniently be split into those that examined water quality conditions within the DWQ drains and MEV land drainage canals (Objectives 1-5), and those that compared the water quality signatures to other waterbody classes throughout the region (Objectives 6-8).

3.2.1 Within the drainage network

A monthly water quality monitoring programme was initiated at 20 drain sites; six in the Kaituna Plains and 14 in the Rangitāiki Plains. Samples were collected from each site and analysed for a range of parameters, including nutrients (total ammoniacal nitrogen (ammonia), total nitrogen (TN), total oxides of nitrogen (NOx-N), dissolved reactive phosphorus (DRP), total phosphorus (TP)), E. coli, water clarity and suspended solids. Spot measurements were also made in the field of dissolved oxygen (DO) (both as a concentration and % saturation), and temperature. Dissolved Oxygen was measured at the top of the water column, as pronounced vertical stratification was often observed in some drains where DO declined markedly at the streambed (personal observations). Conductivity and pH were measured in the lab. All meters were calibrated prior to use on each day.

All raw water quality data was checked to highlight outliers in the data that may have represented equipment error, or data entry errors. Eleven values were found to have percentage oxygen concentrations less than 5%, but these were thought to reflect real observations indicative of highly anoxic environments in some of the drains. A further 28 values were recorded as more than 140% saturated. Although supersaturated values greater than 100% are relatively common under conditions of high photosynthesis during algal blooms, values exceeding 140% may have reflected non-linear responses of metres at these are very high levels. This data was thus censored to the 95th percentile of all observations (140%). Following censoring, all data (except pH) was log +1 transformed to achieve normality. Five analyses were then done on this transformed data to describe water chemistry conditions in the drains and their relationship to rainfall.

Summary statistics were first calculated to describe the overall water quality conditions of the drains, when classified according to two different water quality classes (drain water quality (DWQ), and modified watercourse with ecological values (MEV)). These descriptive statistics were calculated for drains and land drainage canals separately in the Kaituna and Rangitaiki Plains. A two-way ANOVA was then used to determine whether water quality parameters differed between the Kaituna and Rangitaiki Plains, and between the two different water quality classes (DWQ and MEV). Statistical differences were set at the $P < 0.05$ value.

Secondly, all sampling dates were then allocated to their appropriate season[5] to determine whether water quality differed temporally during the 18 month sampling program. Another two-way ANOVA was used to determine whether water quality parameters differed between seasons, and between the water quality classes DWQ and MEV. This analysis also showed whether there was a difference in seasonal patterns between drains of different water quality classes. Such a difference would be shown by a significant season x WQ-class interaction effect.

One of the main roles of the drainage network is to drain excess surface and groundwater from productive farmland. Therefore, it was reasonable to assume that during times of heavy rainfall, the drainage network would be receiving more inputs from the surrounding land. The third analysis consequently examined relationships between total rainfall in the catchment and water quality conditions in the drains. Rainfall data was obtained from BOPRC rainfall sites at Te Tumu (for the Kaituna sites) and at Edgecumbe (for the Rangitāiki sites). The total rainfall prior to sampling was calculated using this rainfall data from six different time periods; the total rainfall between sampling periods, and the total rainfall at 20, 10, 5, 3 and 1 days prior to sampling.

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[5] *Seasons are defined here as: Summer (1 December–28/29 February); Autumn (1 March–31 May); Winter (1 June–31 August), Spring (1 September–30 November).*

Within each drain, all water quality data was normalised to its median value during the study period. Relationships between changes in water quality relative to the median and antecedent rainfall conditions were assessed by regression analysis. All normalised water quality data and antecedent flow data were also log transformed prior to this analysis. Scatter plots of the data showed that a linear regression analysis seemed to be the most appropriate.

The fourth analysis assessed three water quality parameters (total oxides of nitrogen, ammonia (adjusted to 20℃) and *E. coli*) against Appendix 2 of the NPS-FM. This appendix provides numerical bands for specified attributes and the National Objectives Framework (NOF) within the NPS-FM requires that councils set objectives for water bodies to maintain or improve water quality for these (and other) attributes. Although only modified watercourses with ecological values (MEV) are subject to objective setting under the NPS FM, water chemistry from the drain water quality sites (DWQ) was also assessed according to the same NPS-FM Appendix 2 framework for comparative purposes.

Dissolved oxygen is also a water quality attribute specified in Appendix 2 of the NPS-FM, with four different bands being set based on either continuous measurement of one day summer minimums, or seven day summer mean minimums.

The NPS-FM dissolved oxygen attribute only applies below point source discharges, presumably to ensure that point source discharges do not cause an adverse effect on the receiving environment by lowering dissolved oxygen. The Appendix 2 DO attributes are also based only on data collected from November to April inclusive; a time when greatest potential DO stress may occur due to warmer temperatures, and increased respiration rates. Furthermore, the DO attributes rely on continuous measurements, as DO displays strong diurnal variability, with peaks in the mid-afternoon (corresponding to high plant photosynthesis rates), and troughs in the early morning (corresponding to high night time respiration rates). Dissolved Oxygen measurements during this study were based only on single measurements performed monthly in the 20 sites, and so may have over estimated the true diurnal minimum (Dupree et al 2016). To overcome this, Dupree et al (2016) suggest combining the C and D bands into a single at-risk classification and thus improve comparability with the NOF D band classifications based on continuous measurements. All data collected from November 2015 until April 2016 inclusive, was thus allocated to the provisional SPOT DO attributes of Dupree et al (2016) to provide some context as to conditions faced by the biota in the streams.

The fifth analysis involved assessing the effect of pump station discharges from the drainage network into natural receiving waters; in this case the Landenberg Drain that discharged into the Waioho Stream. Water samples were taken from the Waioho Stream at one site 20 m above the discharge, and one site at about 50 m below the discharge. This downstream site was approximately six times the average width of the Waioho, double the RNRP guidance of three times the width of the stream at the point of discharge. However, this zone was probably still within the main discharge plume and was not completely mixed (personal observations, see Figure 14), so as such likely represents a worse case scenario. During the 18-month sampling period, samples were collected on five occasions when the pump station was in operation. The effect of this discharge on the water chemistry of the Waioho River was assessed by calculating the ratio of different water quality parameters at the downstream site relative to the upstream site. A t-test was performed on the data grouped according to whether the pump station was operating or not to assess whether there were any differences in this ratio.

3.2.2 Comparison to other WQ classes

The sixth analysis compared average water quality signatures from the DWQ and MEV sites to other waterways throughout the region when these were classified according to their water quality classification as found in the RWLP.

Data from 67 waterways sampled during the same period as part of BOPRC's NERMN programme, or as part of a gap filling study (Carter et al 2016) were used in this analysis, the most numerous of which were classified as aquatic ecosystem (24 sites), followed by MEV (17 sites) and DWQ (six sites). Four waterway classes (contact recreation, fish spawning purposes, natural state, and water supply) had only limited replication, with only three or less sites. These classes were subsequently grouped to a new classification "other". All water quality data was log transformed, and GLM used to assess differences in water quality parameters and water quality classes. Following this analysis, a principal components analysis (PCA) was conducted to graphically represent the spatial signatures of streams in the different water quality classes. Both these analyses used the average values for each of the water quality variables based on data collected over 17 months between November 2015 and March 2017.

Agricultural drainage ditches are thought to represent ecosystems intermediate between streams and wetlands (Needleman et al. 2007). To further characterise the water quality signatures of the drains sampled, water samples were collected from six wetlands located in the Rangitaiki Plains in January, February and March 2017.

The seventh analysis then compared water quality signatures of the wetland samples to those of the drainage network collected during the same time period. Because wetland sampling covered only a three-month period, it was not possible to characterise average water quality conditions over a long (>12 month) period. A repeated measures GLM analysis was consequently performed on the log 10+1 transformed data to see whether water quality parameters differed between wetlands, and DWQ and MEV waterways. The repeated measures analysis allowed us to determine whether there were differences both between waterway classes, and also within each waterway class over the three-month sampling period.

Finally, the last analysis calculated specific catchment loads from each waterway, as this may have implications in setting catchment wide loads for sensitive receiving environments such as estuaries. Most of the drains sampled are not gauged, and although some pump stations had data as to how often pumps were running for, this data was not readily available or could not easily be converted into a meaningful estimate of flow. Instead, average flows from each site were obtained from a hydrological model of flows for all waterways in the Bay of Plenty region (Booker and Woods 2014). Each sampling site was allocated to its appropriate NZReach code (see Snelder and Biggs 2002), and this NZReach code was used to determine the average flow from the Booker and Woods model. While we acknowledge that modelled flows may not be accurate for these small waterways, their estimates are unlikely to be an order of magnitude out, and so are useful in giving an approximation of specific loads. The average nutrient and sediment concentrations and *E. coli* counts were calculated from each site and 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 (in kg/ha/year) was also fourth root transformed, and analysed by GLM to see whether it differed between the different water quality classifications.

3.3 Results

3.3.1 Summary water quality statistics

Measured water quality parameters were highly variable, both between drains in each of the Kaituna and Rangitaiki Plains, and in drains of different classes (Table 4, Table 5). Although median conductivity was generally fairly low (0.38 mS cm^{-1}), maximum values in excess of 20 mS cm^{-1} were observed in some drains (Table 4), indicating potential input of saline water at some sites on some occasions. These saline inputs occurred in the Kaituna Drain on five occasions, and the Orini Canal on six occasions.

There were no differences in conductivity between waterways in either the Kaituna or Rangitaiki Plains, or between the two water quality classes (Figure 10).

Measured % DO was also highly variable, ranging from very low minimums of < 5%, to supersaturated values > 110% (Table 4). Median percentage DO was significantly higher in the Kaituna drainage system, and significantly higher in MEV waterways (Figure 10). *E. coli* counts were also highly variable, and were higher in the Rangitaiki than the Kaituna Plains. There was however, no difference in *E. coli* counts between the two water quality classes (Figure 10). Median water temperature was relatively low in all waterways (19–20^{o}C), and did not differ with respect to location or water quality class. Maximum water temperatures in some of the drains were also relatively high, close to 30℃ (Table 4). Water pH was similar between waterways in the Kaituna and Rangitaiki Plains but was significantly higher in waterways classified as MEV (average = 7.1) than in waterways classified as DWQ (average = 6.8), although this slight difference is unlikely to have any ecological significance.

Table 4 Summary statistics for water quality parameters collected from 20 drains in the Kaituna and Rangitaiki Plains over 17 months, when allocated to the two different water quality classes (drain water quality (DWQ), and modified watercourse with ecological values (MEV)). Cond = specific conductivity (mS/cm); % DO = % dissolved oxygen; E. coli = counts of E. coli colony forming units per 100 ml; Temp = temperature (oC).

WMA	Class	Stats	Cond	% DO	*E. coli*	Temp	pH
Kaituna	DWQ	Median	0.618	38.8	190	19.6	6.9
		Minimum	0.259	1.9	20	8.8	5.9
		Maximum	21.9	140	5600	29.8	8.0
		Standard Deviation	4.887	36.1	818	4.825	0.45
	MEV	Median	1.19	88.6	160	17.2	7.2
		Minimum	0.117	1.3	9	9	6.5
		Maximum	8.34	140	3100	24.1	8.0
		Standard Deviation	2.574	31.9	596	3.632	0.32
Rangitaiki	DWQ	Median	0.392	33.0	405	18.8	7.1
		Minimum	0.200	0.3	3	9.4	6.3
		Maximum	35.5	140.0	10000	30.2	7.7
		Standard Deviation	3.94	27.95	1564	4.2	0.3
	MEV	Median	0.272	78.8	300	18.8	7.1
		Minimum	0.079	1.9	0	7.7	5.6
		Maximum	22.7	140	17000	27.6	9.4
		Standard Deviation	3.83	32.0	1462	4.3	0.6

Of the nutrient-related parameters, ammonia concentrations were significantly higher in waterways in the Rangitaiki Plains, and also significantly higher in those classified as DWQ rather than MEV (Table 5, Figure 10). Concentrations of DRP were also significantly higher in DWQ waterways, but similar between waterways in the Kaituna and Rangitaiki Plains. In contrast, concentrations of NOx-N were significantly higher in MEV waterways, but similar between waterways in the Kaituna and Rangitaiki Plains.

Both TN and TP were significantly higher in the Rangitaiki Plains waterways, but similar between the two waterway types (Figure 11).

Table 5 Summary statistics for nutrient data collected from 20 drains in the Kaituna and Rangitaiki Plains over 17 months, when allocated to the two different water quality classes (drain water quality (DWQ), and modified watercourse with ecological values (MEV)). DRP = dissolved reactive phosphorus; NOx-N = total oxides of nitrogen; TN = total nitrogen; TP = total phosphorus. All measurements in g /m^3.

WMA	Class	Stats	Ammonia	DRP	NOx-N	TN	TP
Kaituna	DWQ	Median	0.511	0.033	0.107	1.36	0.189
		Minimum	0.008	0.008	0.001	0.384	0.055
		Maximum	1.73	0.184	2.34	5	0.588
		Standard Deviation	0.452	0.041	0.407	0.857	0.139
	MEV	Median	0.06	0.101	1.29	1.58	0.171
		Minimum	0.004	0.061	0.003	0.52	0.119
		Maximum	1.38	0.209	2.53	3.96	0.437
		Standard Deviation	0.275	0.029	0.431	0.568	0.072
Rangitaiki	DWQ	Median	1.180	0.039	0.476	2.540	0.270
		Minimum	0.000	0.009	0.003	0.781	0.056
		Maximum	2.750	0.440	5.940	20.000	3.300
		Standard Deviation	0.776	0.060	0.915	2.269	0.450
	MEV	Median	0.056	0.033	0.567	1.210	0.113
		Minimum	0.001	0.006	0.001	0.054	0.033
		Maximum	11.000	3.800	4.190	67.000	8.400
		Standard Deviation	1.074	0.348	0.764	5.437	0.724

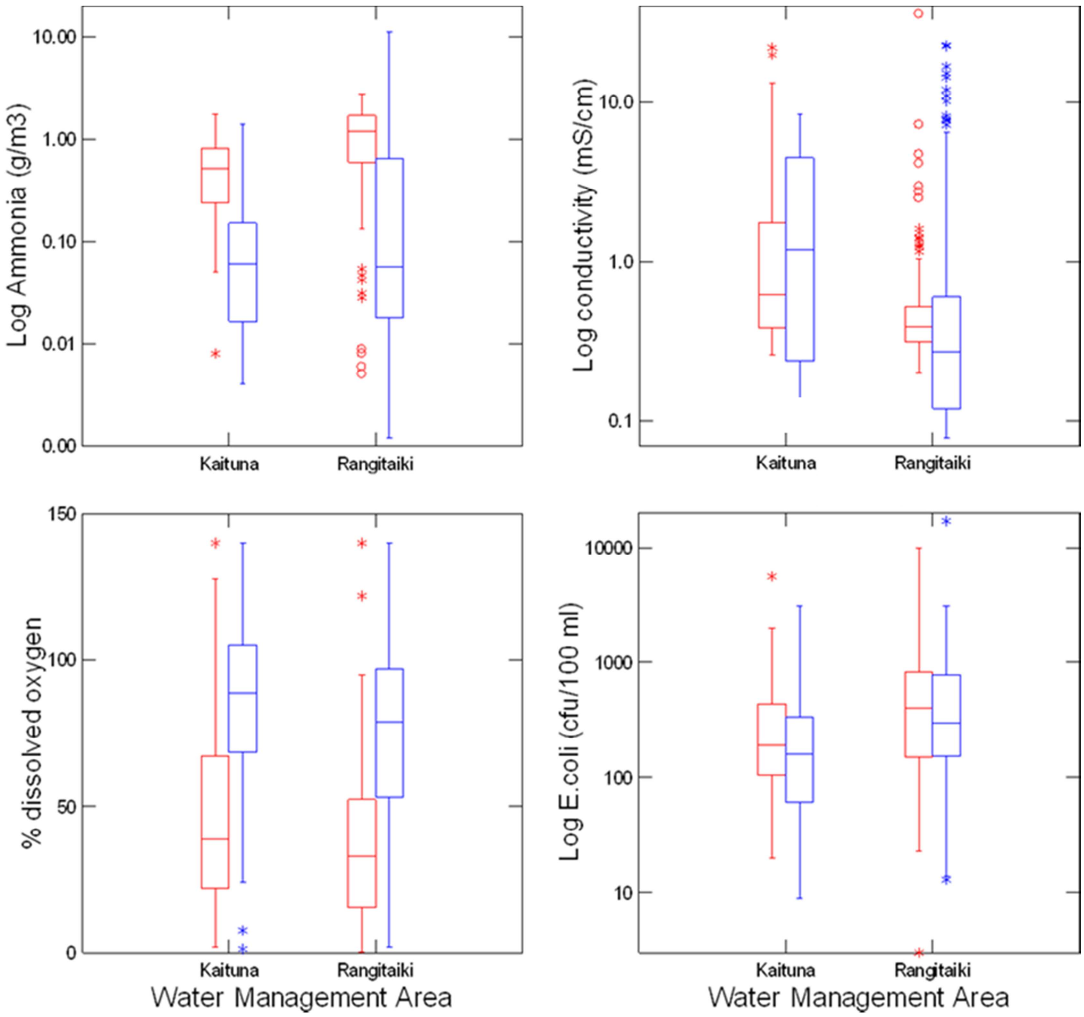

Figure 10 *Box plots of ammonia, conductivity, % dissolved oxygen and E. coil counts from drains in the Kaituna and Rangitaiki Plains, when coded by their water quality classification: red = DWQ; blue = MEV. The central horizontal line indicates the median value, and the bottom and top of the box indicate the 25th and 75th percentile values. The whiskers extend to the 10th and 90th percentiles. Numbers exceeding the 5th and 95th percentiles are also shown (stars or circles).*

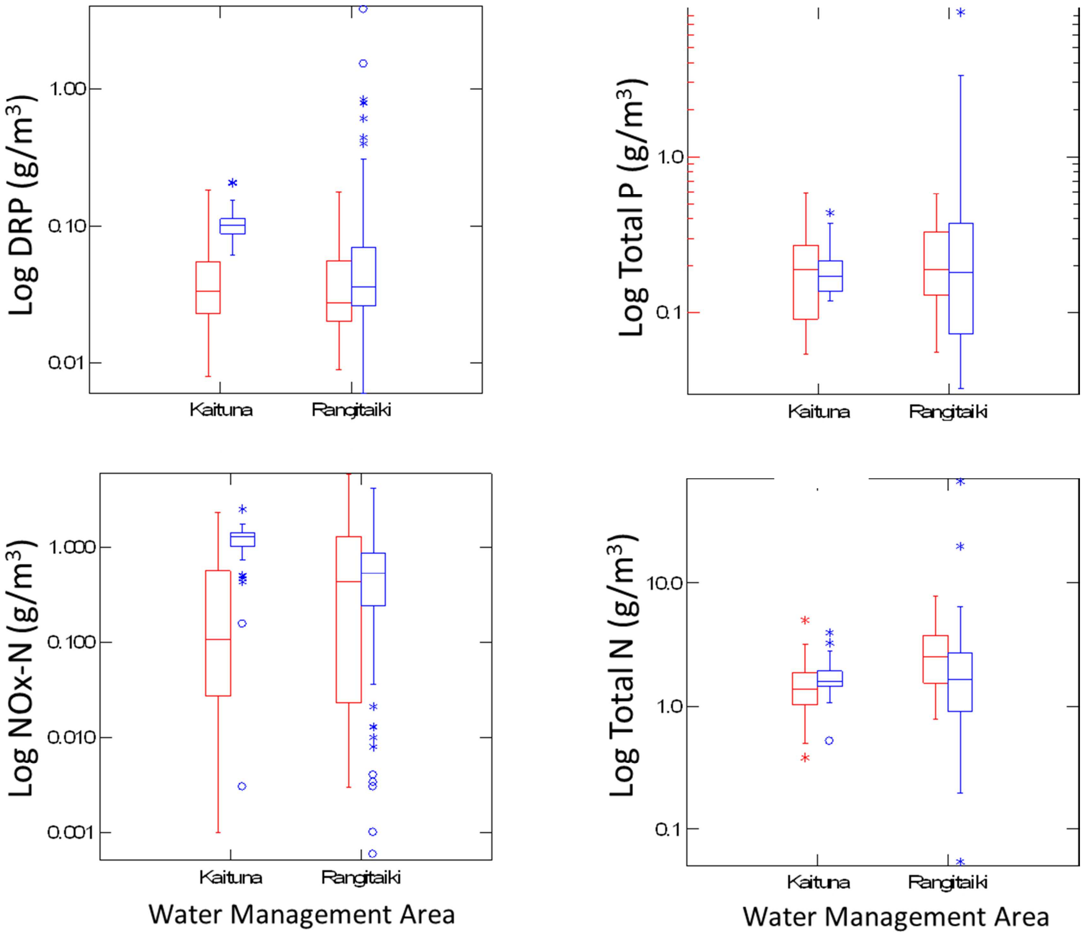

Figure 11 *Box plots of dissolved reactive phosphorus (DRP), total P, total oxides of nitrogen (NOx-N) and total N from drains in the Kaituna and Rangitaiki Plains, when coded by their water quality classification: red = DWQ; blue = MEV. Conventions as per Figure 10.*

3.3.2 Seasonal differences

Significant seasonal effects were found for all water quality parameters (Table 6). For example, pH adjusted ammonia concentrations were higher in Autumn and Winter (Figure 12), and TN concentrations were lowest in summer, intermediate in Autumn and Spring and highest in Winter in both WQ classes (Figure 12). *E. coli* counts were highest in Spring and Autumn in all drains and lowest in Summer and Winter. DO levels were lowest in Autumn only in drains classified as DWQ but showed no observable seasonable pattern in sites classed as MEV (Figure 12).

Table 6 *Results of a two-way ANOVA examining differences in selected water quality parameters between seasons, and between the two water quality classes (drain water quality (DWQ), and modified water courses with ecological values (MEV). Also shown is the Season x WQ class interaction. All significant values (P<0.05) are highlighted in bold.*

Parameter	Source	Type III SS	df	Mean Squares	F-Ratio	p-Value
Ammonia (adjusted)	SEASON	0.80	3	0.27	**11.40**	**0.00**
	WQ_CLASS	2.02	1	2.02	**86.57**	**0.00**
	SEASON * WQ_CLASS	0.15	3	0.05	2.13	0.10
	Error	7.69	330	0.02		
Conductivity	SEASON	1.24	3	0.41	**4.32**	**0.01**
	WQ_CLASS	0.00	1	0.00	0.00	0.95
	SEASON * WQ_CLASS	0.07	3	0.02	0.24	0.87
	Error	31.48	330	0.10		
Dissolved Oxygen	SEASON	0.91	3	0.30	**2.85**	**0.04**
	WQ_CLASS	11.98	1	11.98	**113.15**	**0.00**
	SEASON * WQ_CLASS	1.86	3	0.62	**5.86**	**0.00**
	Error	34.94	330	0.11		
pH	SEASON	0.91	3	0.30	**2.85**	**0.04**
	WQ_CLASS	11.98	1	11.98	**113.15**	**0.00**
	SEASON * WQ_CLASS	1.86	3	0.62	**5.86**	**0.00**
	Error	34.94	330	0.11		
E. coli	SEASON	9.29	3	3.10	**10.40**	**0.00**
	WQ_CLASS	0.27	1	0.27	0.92	0.34
	SEASON * WQ_CLASS	0.20	3	0.07	0.22	0.88
	Error	98.22	330	0.30		

Parameter	Source	Type III S S	Df	Mean Squares	F-Ratio	p-Value
Temperature	SEASON	2.86	3	0.95	**339.52**	**0.00**
	WQ_CLASS	0.00	1	0.00	1.42	0.23
	SEASON * WQ_CLASS	0.01	3	0.00	0.71	0.55
	Error	0.93	330	0.00		
Total suspended solids	SEASON	1.26	3	0.42	**2.80**	**0.04**
	WQ_CLASS	0.18	1	0.18	1.17	0.28
	SEASON * WQ_CLASS	0.97	3	0.32	2.17	0.09
	Error	49.38	330	0.15		
Total N	SEASON	1.12	3	0.37	**11.80**	**0.00**
	WQ_CLASS	0.46	1	0.46	**14.66**	**0.00**
	SEASON * WQ_CLASS	0.06	3	0.02	**0.63**	**0.60**
	Error	10.40	330	0.03		
Total P	SEASON	1.12	3	0.37	**11.80**	**0.00**
	WQ_CLASS	0.46	1	0.46	**14.66**	**0.00**
	SEASON * WQ_CLASS	0.06	3	0.02	0.63	0.60
	Error	10.40	330	0.03		

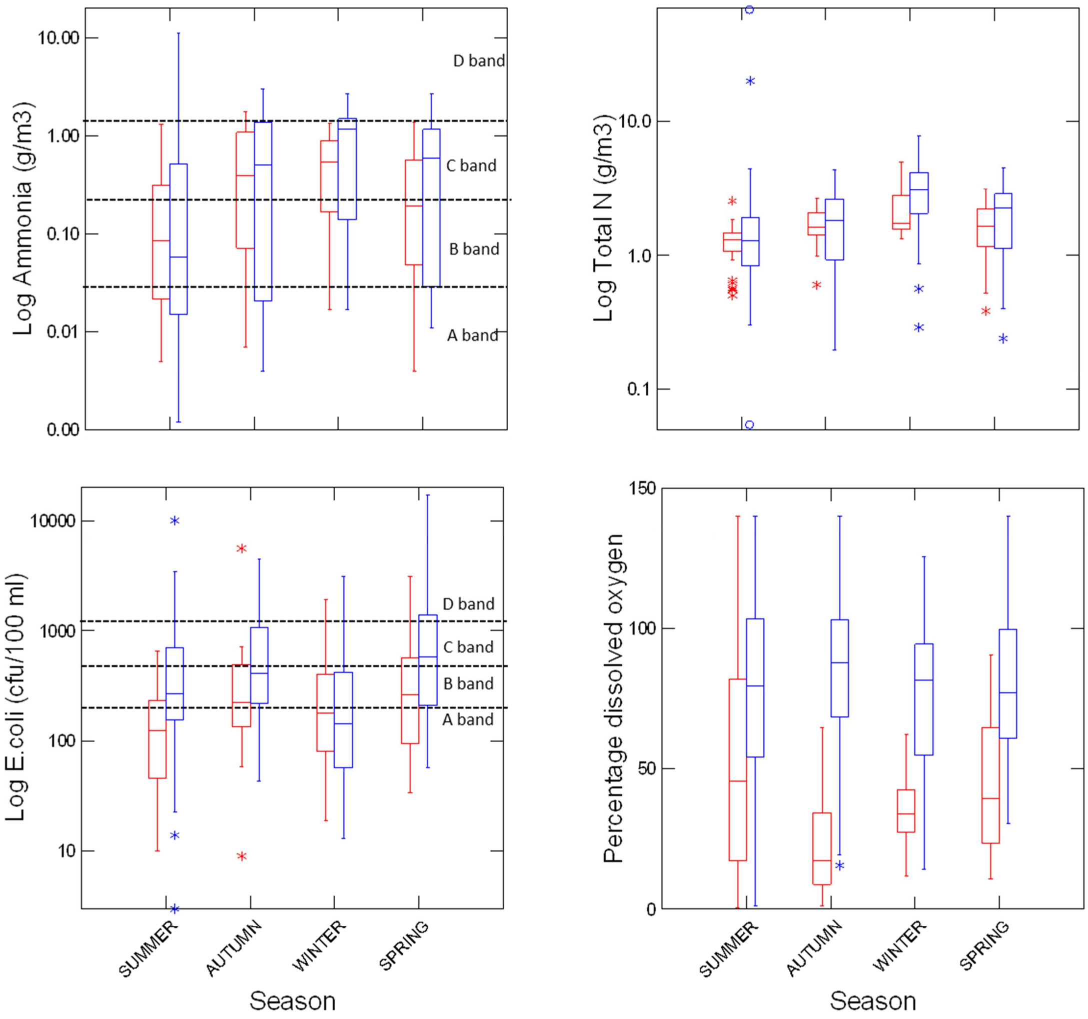

Figure 12 *Box whisker plots of ammonia, total nitrogen, E. coli and percentage dissolved oxygen collected from different waterways throughout the Kaituna and Rangitaiki Plains showing seasonal differences in drains classified either as DWQ (red boxes), or MEV (blue boxes). Also shown are the NPS bands for some attributes. Conventions as per Figure 10.*

3.3.3 Relationships between water quality and rainfall

Rainfall data from the Te Tumu and Edgecumbe monitoring sites were highly correlated (R^2 = 0.793, $P < 0.001$), suggesting that similar rainfall patterns occurred over both the Kaituna and Rangitaiki Plains. Pairwise correlation analysis of rainfall data from the six antecedent time periods showed a high degree of correlation between total rainfall between sampling occasions and the 20, 10 and five day rainfall totals, and between the five day rainfall totals, and the three and one day total. Based on this, only the total rainfall between sampling periods and rainfall in the three-day period prior to sampling was used. The one-day antecedent rainfall total was not used for this analysis as it contained more zero values than the three day antecedent rainfall totals.

Linear regression analysis of total antecedent rainfall prior to each sampling occasion against normalised water quality variables showed highly significant positive correlations between ammonia and rainfall in half the sites examined (Table 9). A similar strong response was also observed between total antecedent rainfall and total nitrogen, where nine sites displayed significant relationships (Table 9). Patterns between water quality variables and total antecedent rainfall for other parameters were not as clear cut.

For example, conductivity showed significant positive relationships for only two sites, and negative relationships for four sites. Dissolved oxygen showed positive relationship for two sites, and negative relationships for three sites. *E. coli* counts showed the lowest number of significant relationships to total antecedent rainfall, where counts were negatively correlated to rainfall only at the Waioho site downstream of Landenberger Road Drain (Table 9).

In contrast to the relatively large number of significant correlations between total antecedent rainfall prior to sampling and water quality parameters, few correlations were observed between water quality parameters and three-day antecedent rainfall (Table 8). These results suggest that in general the overall water quality signatures in the drains is more reliably determined by long term rainfall patterns in the catchment rather than short-term rainfall events. This may reflect the fact that many of the drains sampled may not receive inflows from the smaller lateral drains until a certain amount of rainfall has fallen within a catchment, or it may reflect a certain time lag in residence time between rainfall in a catchment and this entering the larger drains.

Table 7 Summary results of linear regression analysis of water quality parameters against total antecedent rainfall prior to sampling showing whether any observed trends were significantly ($P<0.05$) positively correlated to rainfall (blue, +ve), or negatively correlated (red, -ve).

Site	Ammonia	Cond	% DO	*E. coli*	TN	TP	TSS	Total
Bell Road at Te Puke	+ve				+ve			2
Kaituna at Pah Road	+ve		-ve-ve		+ve			3
Kaituna at Kaituna Road		-ve						1
Pongakawa at Cutwater Road	+ve		+ve				+ve	3
Pukehina at Pukehina	+ve				+ve			2
Wharere at Pukehina		-ve			+ve			2
Awaiti Canal at Matata	+ve		-ve		+ve	+ve		4
Awakaponga Canal at Matata	+ve	+ve						2
Eastern Drain at Awakeri								
Langenberger Road at Poroporo	+ve				+ve			1
Omeheu Canal at Poplar Lane		+ve			+ve	+ve	+ve	4
Omeheu Canal at Soldiers Road	+ve				+ve			2
Orini Canal							-ve	1
Reids Central Canal						-ve		1
Secombes Canal	+ve		-ve		+ve			3

Site	Ammonia	Cond	% DO	*E. coli*	TN	TP	TSS	Total
Sutherland Road	+ve							1
Te Rahu Canal		-ve	+ve					2
Waioho downstream		-ve		-ve				2
Waioho upstream								
Western Drain								
Total positive relationships	10	2	2		9	1	2	
Total negative relationships		4	3	1		2	1	
Total no relationships	10	14	15	19	11	17	17	

Table 8 Summary results of linear regression analysis of water quality parameters against the three-day antecedent rainfall prior to sampling showing whether any observed trends were significantly (P<0.05) positively correlated to rainfall (+ve), or negatively correlated (-ve).

Site	Ammonia	Cond	% DO	*E. coli*	TN	TP	TSS	Total
Bell Road at Te Puke		-ve						1
Kaituna at Pah Road								
Kaituna at Kaituna Road								
Pongakawa at Cutwater Road								
Pukehina at Pukehina								
Wharere at Pukehina								
Awaiti Canal at Matata								
Awakaponga Canal at Matata	+ve							1
Eastern Drain at Awakeri		-ve	-ve			+ve		3
Langenberger Road at Poroporo								
Omeheu Canal at Poplar Lane								
Omeheu Canal at Soldiers Road								

Site	Ammonia	Cond	% DO	*E. coli*	TN	TP	TSS	Total
Orini Canal								
Reids Central Canal								
Secombes Canal			-ve					1
Sutherland Road								
Te Rahu Canal								
Waioho downstream								
Waioho upstream								
Western Drain								
Total positive relationships	1					1		
Total negative relationships		2	2					
Total no relationships	1	2	2	0	0	1	0	

3.3.4 NPS-FM Appendix 2 assessment

Examination of data collected in the 2016 calendar-year showed that most DWQ and MEV sites were in either the A or B band for total oxides of nitrogen (NOxN), although a single MEV site (Reids Central Canal at Edgecumbe) was classified in the C band for the 95th percentile of this parameter (Table 9). Two DWQ sites (Eastern Drain and Langenberger Road Drain) were also in the C band for the 95th percentile.

Half the sites were classified in the C band for median values of ammonia, while the other half were either in the A or B band Table 9). When assessed according to the maximum ammonia concentrations, 13 sites were in the C band, whilst only three and four were in the A or B band respectively.

Table 9 Analysis of 12 monthly data (January 2016 to December 2017) against NPS-FM Appendix 2 bands for total oxides of nitrogen (NOX N) and ammonia.

		NOxN		Ammonia (adjusted)	
Site	**WQ Class**	**Med**	**95th**	**Med**	**Max**
Bell Rd Drain at Te Puke	DWQ	A	B	C	C
Kaituna Drain at Pah Road	DWQ	A	A	C	C
Kaituna Drain at Kaituna Road	DWQ	A	A	B	C
Pongakawa Drain at Cutwater Road	MEV	B	B	A	A
Pukehina Drain at Pukehina	MEV	B	A	B	C
Wharere Drain at Pukehina	MEV	B	B	B	B
Eastern Drain	DWQ	B	C	C	C
Langenberger Rd (Drain18)	DWQ	A	C	C	C
Secombes Canal at Greig Road	DWQ	A	A	C	C
Awaiti Canal	DWQ	A	B	C	C
Awakaponga Canal	MEV	A	A	A	A
Omehue Canal u/s WWTP	MEV	A	B	C	C
Omehue Canal d/s WWTP	MEV	A	B	C	C
Orini Canal off Thornton Road	MEV	A	A	B	C
Reids Central Canal	MEV	A	C	C	C
Section 109	DWQ	A	B	C	C
Te Rahu Canal	MEV	A	A	B	B
Waioho Stream d/s Drain_18	MEV	A	B	A	B
Waioho Stream u/s Drain_18	MEV	A	A	A	A
Western Drain	MEV	B	B	A	B
Totals	A	15	8	5	3
	B	5	9	5	4
	C	0	3	10	13
	D	0	0	0	0

Examination of the six-monthly spot measurements of DO clearly showed that some of the sites were displaying oxygen levels where significant, persistent stress on organisms would be likely. Seven sites had DO concentrations classified as at risk on at least five of the six months during the spring-summer period (Table 10). Other sites however had much higher DO concentrations and were in either the A or B band for spot DO measurements. Spot DO levels in these sites were above levels thought to stress on aquatic organisms; however spot samples during the day are conservative, and had night time DO levels been measured then dips below the NBL are likely to have been much greater in number and magnitude.

Table 10 Summary table showing the number of times dissolved oxygen concentrations fell in the bands given for the one-day summer minimum as per Appendix 2 of the NPSFM.

Area	Site	A	B	At risk
Kaituna	Awakaponga Canal	4		2
	Bell Road Drain at Te Puke			6
	Kaituna Drain at Kaituna Road	2	1	3
	Kaituna Drain at Pah Road			6
	Pongakawa Drain at Cutwater Road	4	1	1
	Pukehina Drain at Pukehina	3		3
	Wharere Drain at Pukehina	1	2	3
Rangitaiki	Awaiti Canal	1		5
	Eastern Drain			6
	Langenberger Road Drain	1	1	4
	Omehue Canal downstream WWTP	1		5
	Omehue Canal upstream WWTP	1	1	4
	Orini Canal off Thornton Road	3		2
	Reids Central Canal	2		4
	Secombes Canal at Greig Road			5
	Section 109			6
	Te Rahu Canal	1	2	3
	Waioho Stream downstream of Drain_18	3	1	1
	Waioho Stream upstream of Drain_18	4	1	1
	Western Drain	2		4

3.3.5 Effects of Landenberger Road Drain on Waioho Stream water quality

The ratio of water quality parameters between downstream and upstream sites were calculated on each sampling occasion. Over the 16-month period, samples were collected on five occasions when the pump was operating.

A t-test was therefore used to determine whether the ratio of different water quality parameters differed during times when the pump station was operating or not. This tested the null hypothesis that there would be no difference in the downstream to upstream ratio during periods where the pumps were operating and where they were not.

Of the 10 water quality parameters assessed, seven displayed significant differences in the calculated downstream/upstream water quality ratios (Table 11). Of these, all but dissolved oxygen had higher ratios during times of pumping; suggesting that the discharge from the Landenberger Road drain was significantly increasing concentrations of these parameters in the Waioho Stream. The ratio of dissolved oxygen was significantly lower during times of pumping; suggesting that the Landenberger Road Drain discharge was reducing oxygen concentrations in the Waioho River.

Table 11 Results of two sample t-tests determining whether the ratio of different water quality parameters downstream and upstream of the discharge differed significantly between times when the Landenberger Road pump station was discharging and when it wasn't. The t-test statistic was negative when the ratio was greater during times when the pump station was operating and positive when the ratio was less during times when the pump was operating.

Water quality parameter	T-test	P value	Summary result
Ammonia	-11.74	<0.001	d/s > u/s
Conductivity	-9.63	<0.001	d/s > u/s
Dissolved oxygen	4.38	0.012	d/s < u/s
Dissolved reactive phosphorus	4.92	N.S	No effect
E. coli	-0.97	N.S	No effect
Nitrate -nitrite (NOx-N)	-2.71	0.05	d/s > u/s
Total nitrogen	-3.66	0.01	d/s > u/s
Total phosphorus	-1.58	N.S	No effect
Temperature	7.95	0.039	d/s > u/s
Turbidity	9.17	<0.001	d/s > u/s

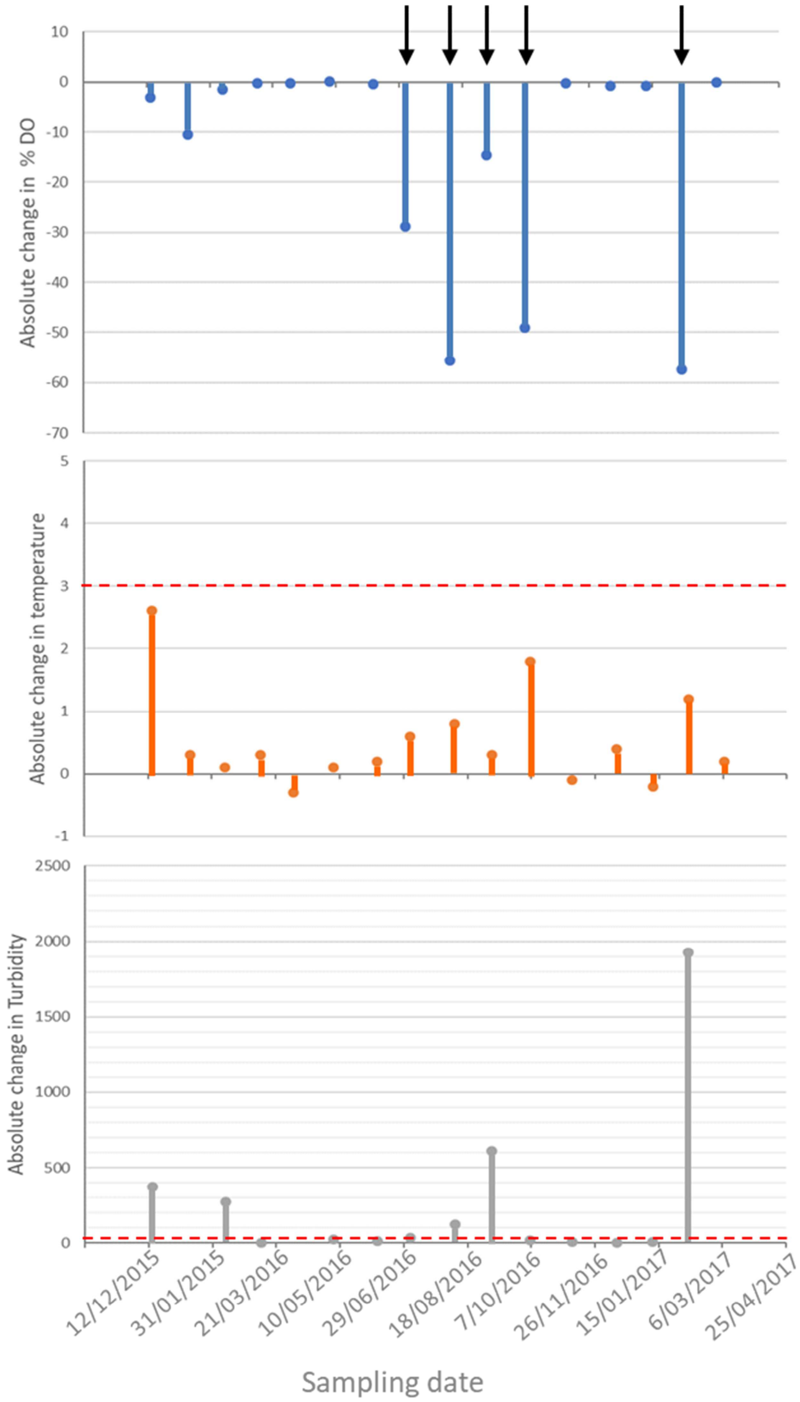

Figure 13 *Absolute differences in percentage DO (dissolved oxygen), temperature and turbidity between the downstream and upstream sites in the Waioho Stream during the sampling period. During this time, samples were collected when the Landenberger Road pump station was discharging (black arrows) under the RWLP, discharges into waterways classified as MEV cannot exceed the temperature by more than 3°, or Secchi disc by more than 20%. These standards/criteria are shown by the red dashed lines.*

Figure 14 Example of the change to visual clarity in the Waioho Stream during times when the Landenberger Road Drain was discharging.

3.3.6 Comparison of water quality signatures to other waterways

The GLM analysis of water chemistry collected over a 17-month period from the drains and other waterways in the region showed clear differences between the water quality classes. Drain Water Quality drains had significantly lower dissolved oxygen concentrations than waterways classified as either RBL, AE or Other. Dissolved oxygen concentrations in MEV drains were higher than in DWQ drains and not statistically significant from waterways classified as RBL (Figure 15). RBL waterways, as well as DWQ MEV waterways also had the lowest conductivity, whereas waterways classified as AE or Other had the highest conductivity (Figure 15). Particularly high readings were recorded from (the Wairoa at SH 2, which) was tidal, and at times was sampled during high tide when the salt wedge would have passed the sampling site.

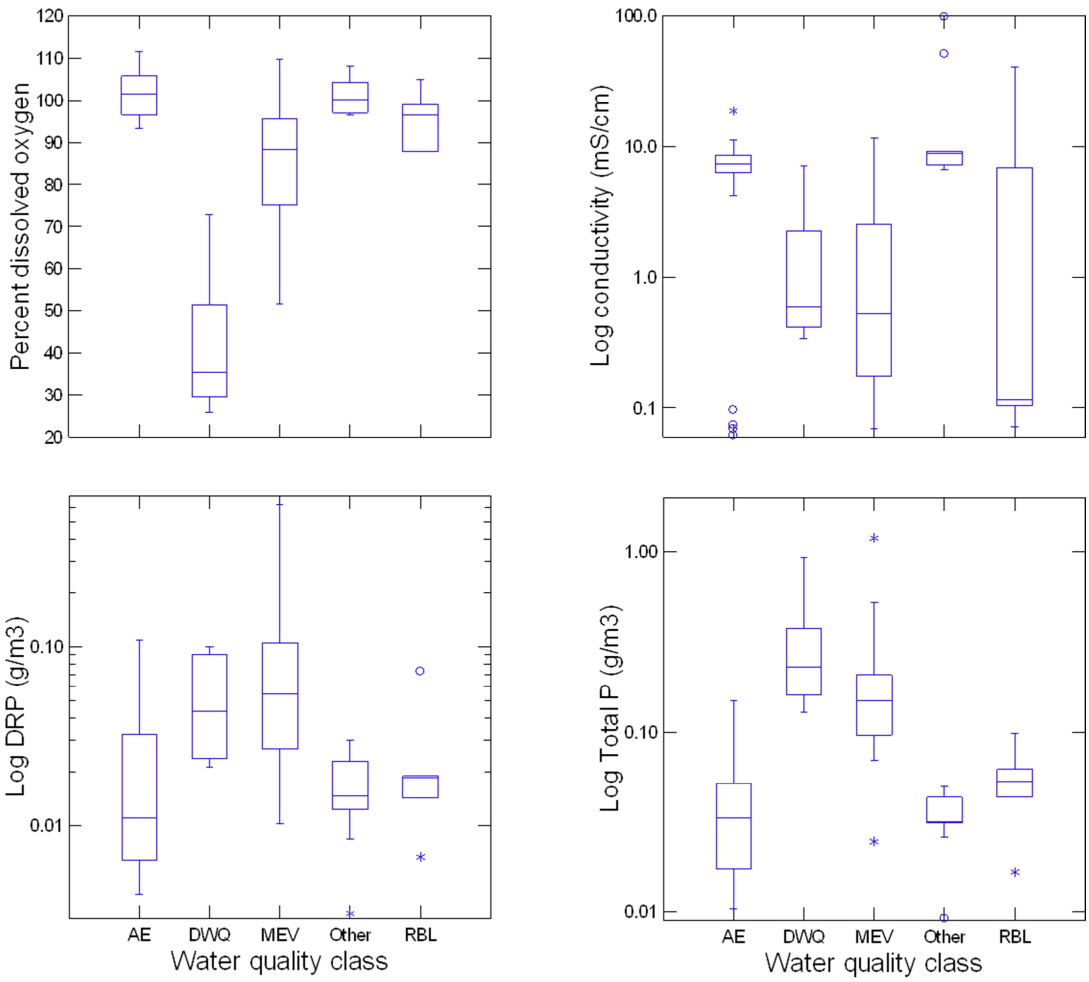

Figure 15 *Box plot of percentage dissolved oxygen, specific conductivity, dissolved reactive phosphorus (DRP), and total phosphorus (TP) collected from 68 waterways throughout the region, and assigned to water quality classes as per the RWLP. AE = Aquatic Ecosystem; DWQ = drain water quality; MEV = Modified watercourses with Ecological Values; Other = a mix of Contact recreation, Fish spawning purposes, Natural state and Water supply. Conventions as per Figure 10.*

Both MEV and DWQ waterways had the highest concentrations of DRP, ammonia, total nitrogen and total phosphorus, as well as having the highest mean average spot temperatures (Figure 15, Figure 16). Concentrations of NOx – N were highest in MEV waterways, whereas they were only intermediate in the DWQ waterways. Of interest was the finding that neither the 17-month average total suspended solids nor E. coli counts differed between the water quality classes.

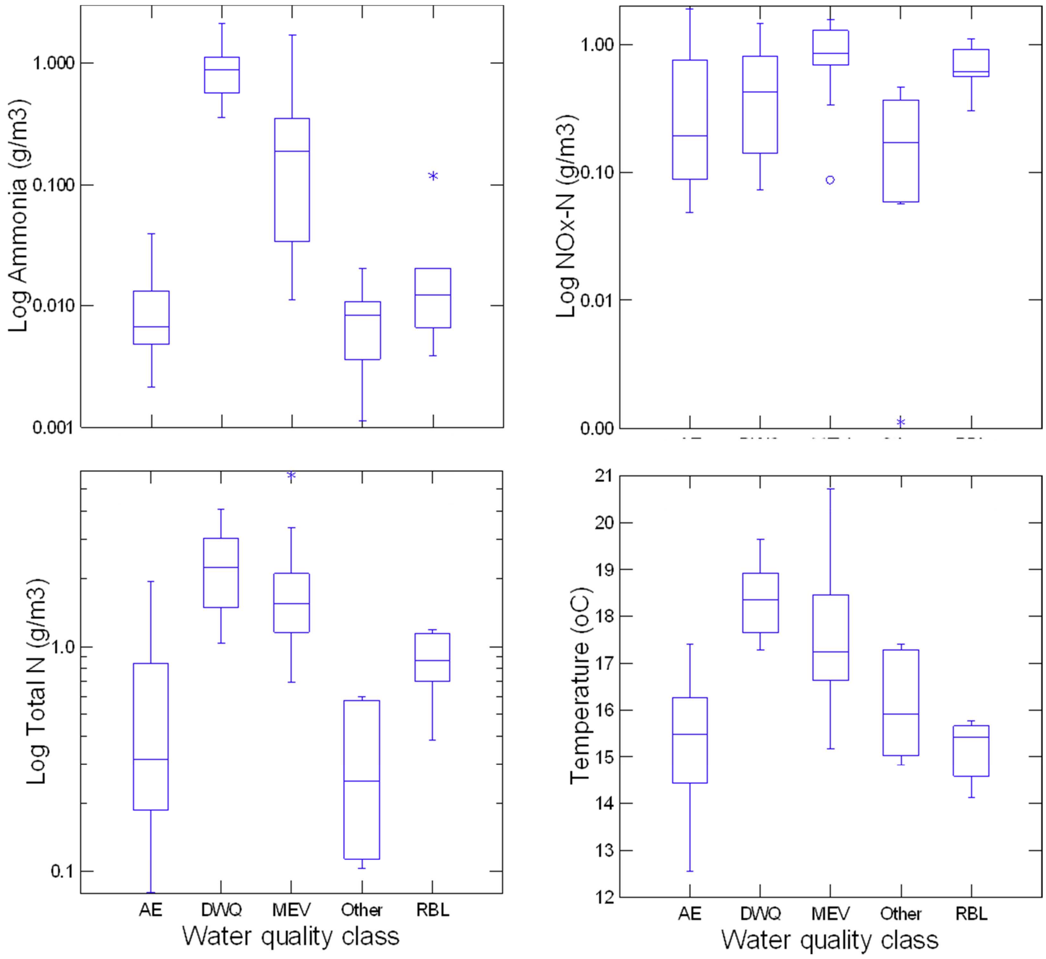

Figure 16 *Box plot of ammonium-N, total oxides of nitrogen (NOx-N), total nitrogen and temperature collected from 68 waterways throughout the region, and assigned to water quality classes as per the RWLP. Conventions as per Figure 10.*

3.3.7 Comparison of water quality signatures to wetlands

Repeated measures analysis showed significant differences between wetlands, and waterways assessed as DWQ and MEV for ammonia, percentage dissolved oxygen, NOx-N and total suspended solids. Ammonia concentrations were highest in DWQ sites, while dissolved oxygen concentrations were also the lowest (Figure 17). NOx-N concentrations were significantly higher in MEV waterways, and lowest in wetlands, while total suspended solids were significantly higher in wetlands (Figure 17). No other water quality parameters differed significantly between these three different waterway types.

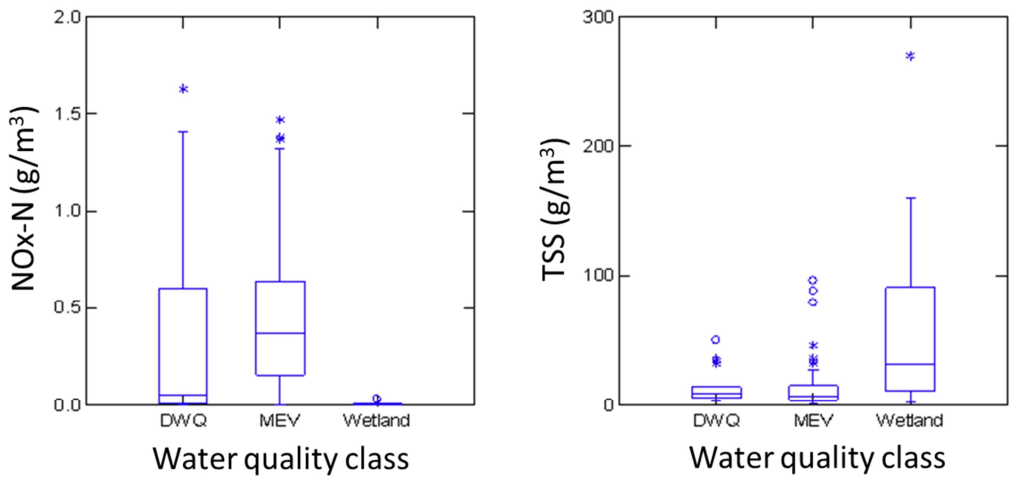

Figure 17 *Box plot of total oxides of ammonia, dissolved oxygen, nitrogen (NOx-N), and total suspended solids (TSS) collected from six wetlands and DWQ waterways, and from 14 MEV drains in the Kaituna and Rangitaiki Plains over a three month period. Conventions as per Figure 10.*

3.3.8 Calculation of specific catchment loads

Specific catchment loads for five of the measured water quality factors differed between the five water quality classes. Specific catchment loads of ammonia and TP were highest in the DWQ and MEV drains, and lower in the other waterway types. Catchment loads of TN were also highest in DWQ and MEV and lowest in waterways classed as either AE or Other, but intermediate in the RBL waterways. Catchment loads of DRP and NOx-N were significantly highest in MEV waterways, but loads of these nutrients were not significantly different between DWQ sites and the other waterways (Figure 18). No significant differences were found in catchment loads of either TSS or *E. coli* between the different water quality classes.

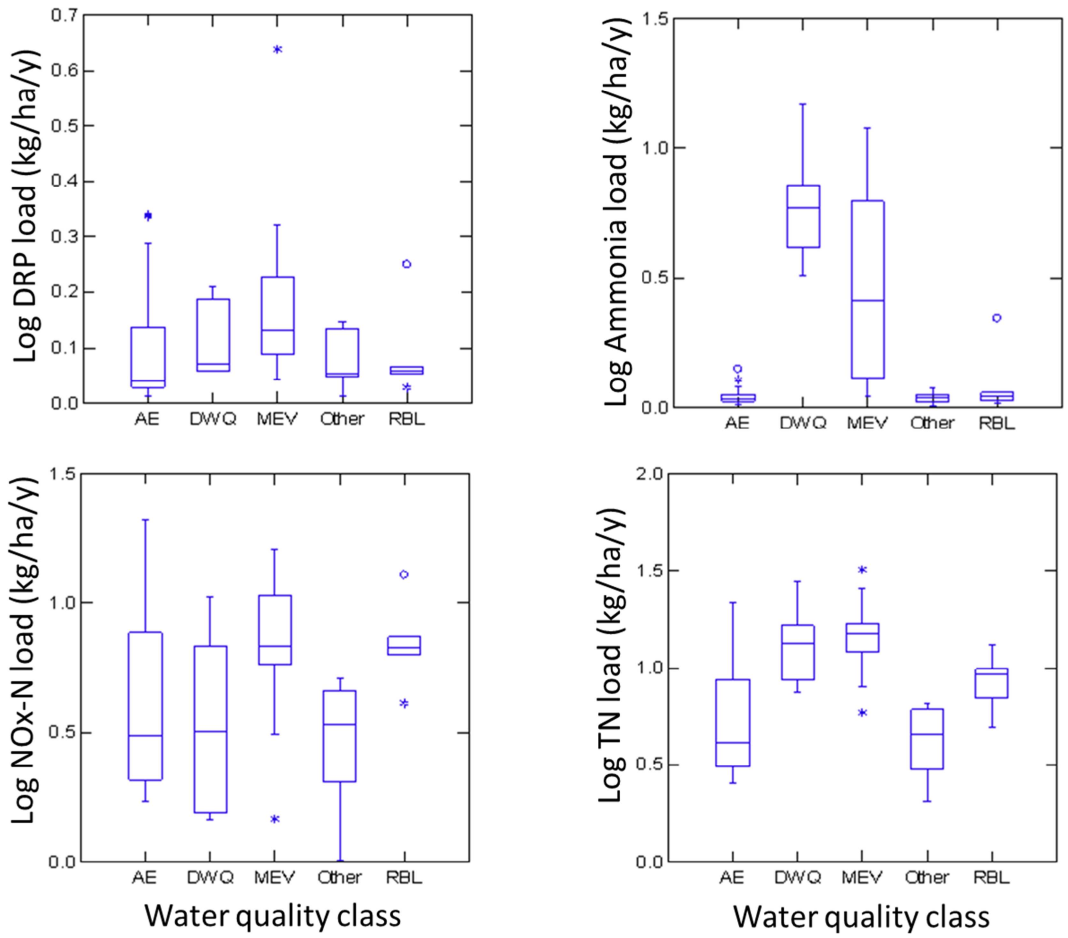

Figure 18 *Calculated catchment loads from the 61 monitored streams, based on observed averages from monthly water quality sampling, and modelled average discharge. Conventions as per Figure 10.*

Some of the differences in calculated loads may simply reflect landuse differences between streams in the different water quality classes. For example, over half the waterways in both the AE and other classes drained catchments dominated by native bush, so their specific catchment loads would expect to be lower than from waterways draining agricultural catchments. Catchment loads were next assessed for waterways in agricultural catchments (based on the REC Pasture land cover class) only to see if loads differed between the different water quality classes. This analysis showed that specific loads of ammonia, TN and TP were still highest in the DWQ and MEV drains, and lower in the other waterway types (Figure 19). Indeed, specific loads of ammonia were often between 25 and 50 times higher in the DWQ and MEV waterways than other waterway classes.

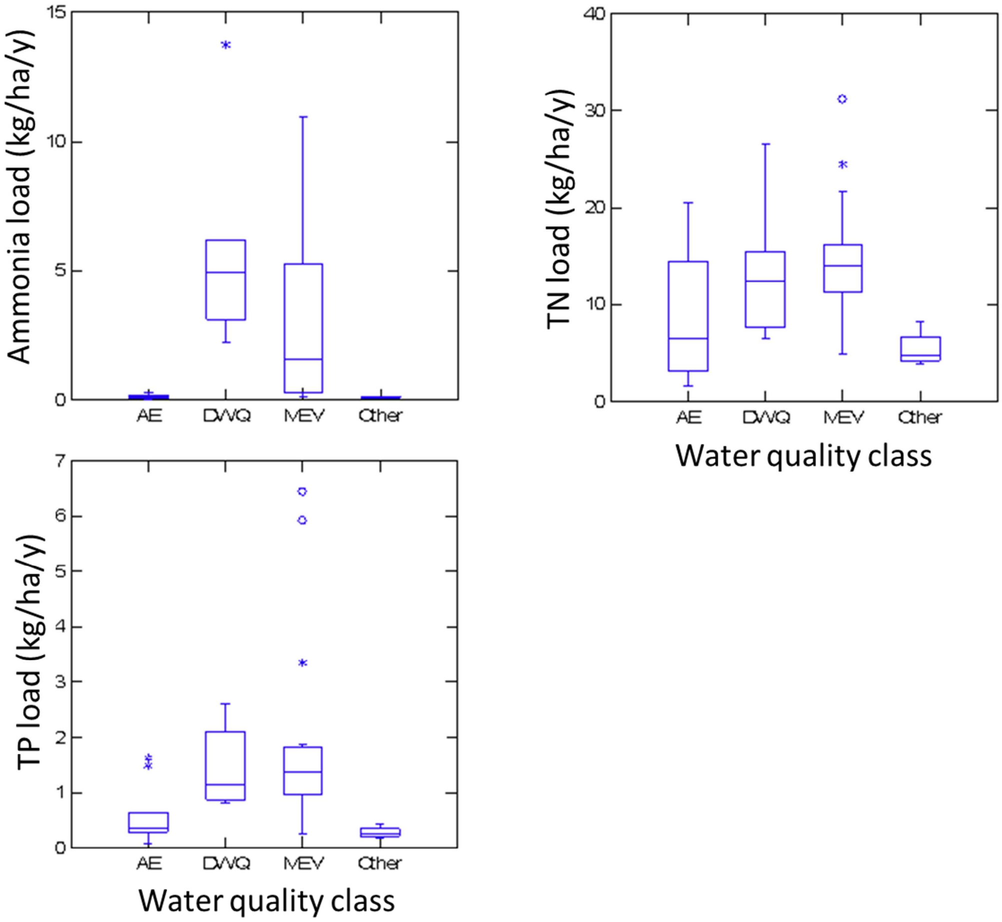

Figure 19 *Calculated catchment loads from 35 streams draining catchments dominated by agriculture, based on observed averages from monthly water quality sampling, and estimated average discharge. Conventions as per Figure 10.*

3.4 Discussion

This study provided valuable information about the water quality conditions in the 20 monitored drains in the Kaituna and Rangitaiki Plains, and compared their conditions to those in other monitored waterways throughout the region. Water quality in the drains is best described as poor, with very high nutrient levels, high turbidity, and extreme levels of DO, both high and low. Observed DO levels greater than 100% were not surprising and reflected the large amount of primary productivity in some of these drains. In contrast, other drains were characterised by consistently low to very low DO levels, even during the day when plant photosynthesis was assumed to be high. Low DO levels are a common phenomenon in many modified lowland waterways throughout New Zealand (Larned, Scarsbrook et al. 2004, Wilding, Brown et al. 2012), and has potential adverse impacts on fish communities (Wilding, Brown et al. 2012, Franklin 2014). The low levels presumably reflect a large degree of bacterial decomposition of the large amounts of organic matter present on the bottom of the drains, combined with a lack of re-aeration in these essentially stagnant waterbodies. Furthermore, the elevated temperatures in the drains due to lack of riparian vegetation would further increase bacterial decomposition rates, reducing DO levels even more. That DO levels remained low even during the day when photosynthesis was expected to be high highlights strong heterotrophic nature of some of these waterways, despite them being fully exposed to the sun where primary productivity would be expected to contribute to overall stream metabolism and increase oxygen levels.

The drains in both the MEV and DWQ water quality classes had high nitrate concentrations, but the levels were not likely to be toxic (no sites were in the D band and few were in the C band). This means that toxic effects of nitrate to biota are unlikely, or at best would affect only some sensitive species. In contrast, half the sites surveyed fell in the C-band for ammonia. This may be affecting some sensitive invertebrate species such as fingernail clam and freshwater mussel (kakahi).

Ammonia is one of several forms of nitrogen in aquatic environments. When present at high concentrations, it can become difficult for aquatic organisms to excrete, leading to a toxic build up in their internal tissues and blood, with potentially lethal effects. Environmental factors, such as pH and temperature, can affect ammonia toxicity to aquatic animals. Thus, ammonia is more toxic at high pH and a less toxic at low pH. In addition, ammonia toxicity increases as temperature rises. Although most of the DWQ and MEV sites had average spot pH values of around 7.5, water temperatures in the drains were generally high, so this may have increased potential adverse effects of ammonia on stream fauna.

Natural sources of ammonia include decomposition or breakdown of organic matter, gas exchange with the atmosphere and animal excreta. Ammonia is also produced for commercial fertilisers as urea. It can enter waterways via direct means such as excretion from animals, and indirect means such as nitrogen fixation, air deposition and runoff from the surrounding land. It is this direct runoff that may have been responsible for the observed significant positive correlations between total antecedent rainfall prior to sampling and ammonia levels. Waterlogged soils and a lack of uptake /transformation of ammonia because of N saturation and low DO in the soils may also explain these close links to total rainfall. Given the intimate linkages between the drainage network and the surrounding agricultural landscape, it was expected that a strong signal would be detected between total rainfall in the catchment and resultant water quality, but this was generally not observed for other water quality parameters.

Hickey and Vickers (1994) quantified the toxicity of ammonia to nine native New Zealand freshwater invertebrates, ranging from so-called sensitive taxa such as mayflies, stoneflies and caddisflies, to less-sensitive taxa such as snails and crustacea. They found a wide range of ammonia sensitivities between invertebrates. For example, shrimp (*Paratya*), and the so-called sensitive insects such as the mayfly *Zephlebia,* and the stonefly *Zelandoperla* were the least sensitive species to ammonia. In contrast, animals that are generally considered tolerant to stressors associated with landuse change such as the snail *Potamopyrgus*, the caddis fly *Pycnocentria*, and the crustacean *Paracalliope* were in fact the most sensitive species to ammonia. Ammonia sensitivity was thus poorly correlated to what is considered sensitivity to other stressors associated with land use changes. However, despite this, the finding that ammonia was a major driver of invertebrate community composition between drains and other sites throughout the region (See Section 4), suggests that it may indeed be one of the many pressures regulating invertebrate communities. Other pressures would include low quality habitat and low oxygen levels in some of the drains.

Richardson (1997) also assessed the acute toxicity of ammonia to fish, and the common lowland shrimp *Paratya*. Surprisingly, some New Zealand native fish were more tolerant to ammonia than some invertebrates. For example, the 96h LC50[6] at 15° C varied from 0.75 g/m^3 ammonia for shrimp and banded juvenile kokopu, to 2.35 g/m^3 for shortfin elvers. This latter figure is much higher than the observed LC50 for *Zephlebia* and *Zelandobius* (> 0.8 g/m3). Examination of median ammonia concentrations from the 20 drains over the study showed that median ammonia concentrations in seven drains exceeded the lower threshold for shrimp and banded kokopu, whereas no drains displayed a median concentration greater than 2.35 g/m^3. This suggests that ammonia levels in these drains may not have been particularly important in structuring fish communities in terms of causing increased mortality.

Effects of drains on the receiving environment

The effect of the discharge from the Landenberger Road drain into the Waioho Stream was assessed by monitoring water quality at one site 22 m above the discharge, and one site 50 m below. Both sites were on the true left of the river and as such, the downstream site may have still been in the mixing zone below the discharge. Thus, this study could be regarded as at best an initial scoping study to determine whether indeed the current discharge from the drainage network into natural watercourses may be having adverse effects. It showed that the discharge from the Landenberger Road drain was having consistent demonstrable impacts on at least some water quality parameters including dissolved oxygen and ammonia.

Although the discharge from this drain was sporadic, and into a flowing environment (the Waioho Stream), which itself flowed into another river (the Whakatane), other pumping stations discharge into smaller natural waterways, or to waterways that flow into sensitive receiving environments such as estuaries. Given that there is currently good evidence that sensitive receiving environments such as the Waihi Estuary and Kaituna-Maketū Estuary are stressed from excess nutrients and sediments, it is clearly an important task to determine the fate of contaminants such as nutrients, *E. coli* and sediments from the drainage network (both DWQ and MEV drains) on sensitive receiving environments.

Calculated catchment loads from the monitored drains were generally very high for parameters such as ammonia. This may reflect in part, the intimate contact of the drainage network to the surrounding agricultural land, and for MEV sites, many are at the bottom of large catchments where the effects of all upstream land uses would be magnified. Many of the paddocks in the Kaituna and Rangitaiki Plains contain small lateral drains, dug to help lower the water table and excess surface water after rainfall. The vast majority of these lateral drains have minimal riparian vegetation, with fencing placed right alongside the drain. While this obviously maximises the area of arable, productive land to farmers, it also means that there is little that can be done to capture contaminants that run off the land, or enter the drains as shallow groundwater. These small lateral drains then feed into a network of larger and larger drains, so that these effectively accumulate all of the nutrient, sediment and other contaminants from the landscape.

Although riparian planting around these drains may help minimise contaminants such as *E.coli*, sediment, phosphorus and organic N from entering the drains, other contaminants such as nitrate associated with urine patches is likely to still enter these lateral drains via movement through the soil water. One way that these nutrients can be intercepted before reaching the ultimate receiving environment would be by the use of constructed wetlands placed at strategic locations along the drainage network, or linear wetlands within the drains themselves. Such wetlands may offer additional benefits over and above that of improving water quality, such as provision of biodiversity hotspots for a range of plants, birds and native fish. However, the size and number of wetlands needed to achieve meaningful beneficial effects may be constrained by the areas where

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[6] LC50 is the lethal concentration of a substance that will kill 50% of the test organisms within a pre-defined period, in this case 96h.

constructed wetlands can be used. Moreover, considerable challenges are likely to exist in convincing farmers to trade off some of their currently productive land (with what may be arguably detrimental environmental effects) for the creation of wetlands that would serve to improve water quality conditions within the lowermost areas of extensively farmed catchments. Further research is thus needed to determine the efficacy of such treatment wetlands, and on the practicality of using these for beneficial environmental outcomes while not adversely affecting farming profitability. Notwithstanding the use of "end of pipe" solutions such as constructed wetlands, there is also a need to investigate ways to reduce nutrient/contaminant loads entering drains and land drainage canals, and/or to treat water before it enters freshwater bodies/receiving environments. In this way, the potential contaminant loads reaching the drains could be minimised.

Part 4:

Invertebrate sampling

4.1 Introduction

Aquatic invertebrates consist of the larval and adult stages of aquatic insects, snails, worms, and crustaceans (such as shrimp, or koura). These animals are relatively long lived (months to years), and are relatively sedentary, moving at most only a few kilometres for the most actively drifting insects. These animals also have a wide range of ecological tolerances for different physical, chemical and hydrological conditions. For example, aquatic insects such as caddisflies, stoneflies and mayflies are generally intolerant of warm waters, silty streambeds, and slow flowing water, while other invertebrates such as snails, midges and worms can tolerate these conditions. This explains why the invertebrate communities often differ greatly among streams flowing through catchments dominated by different land cover.

Thus, streams flowing through native bush are often dominated by sensitive mayfly, stoneflies and caddisflies, whilst streams flowing through catchments dominated by intensive agriculture or urban development are dominated by more tolerant animals that can tolerate conditions found in these modified catchments. 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. Stream invertebrates are used by all regional councils throughout New Zealand to help them assess the ecological condition of rivers, and to assist in their statutory responsibilities for environmental monitoring.

The Bay of Plenty Regional Council has been monitoring stream invertebrate communities throughout the region since 1992 as part of the NERMN programme. Currently, the ecological condition of 130 waterways flowing through a range of land-uses is monitored. Most of these waterways consist of generally unmodified systems, while the highly modified drainage networks characteristic of the Kaituna and Rangitaiki Plains have generally not been included in the monitoring. An exception to this is that four sites in each of the Kaituna and Rangitaiki Plains have been sampled as part of this ongoing NERMN monitoring programme. These are classified as modified watercourses with ecosystem values (MEV).

The aim of this section of the report is to characterise the invertebrate communities of the drains and land drainage canals in the Kaituna and Rangitaiki Plains. These communities were then compared to invertebrate communities in other waterways monitored throughout the region to give the results of the plains monitoring some context. Calculated biotic indices were also compared to recommended biological metrics described by (Carter, Suren et al. 2017) as part of the Council's NPS monitoring obligations. Major environmental factors responsible for structuring the invertebrate communities were also identified. These analyses were done on both the data obtained from this survey, as well as other data from other work done by BOPRC, including the annual NERMN invertebrate monitoring programme (Suren et al 2017), and targeted sampling in the Kaituna-Pongakawa-Waitahanui Water Management Area as part of studies conducted to fill information gaps in this area (Carter, Suren et al. 2015). This comparative work was important as it gave the results of the drain monitoring more context and allowed for a much wider range in observed environmental state to be assessed.

4.2 Methods

4.2.1 Field and laboratory methods

Invertebrate data came from a number of sources, including three BOPRC programmes (the annual NERMN invertebrate monitoring programme (Suren et al 2017); a large scale survey of the Rangitaiki Catchment (Suren 2014), and a gap filling monitoring programme run as part of Council's NPS-FM commitments (Carter et al. 2015, Suren, Suren et al. 2015). Other invertebrate data was sourced from a number of consent applications or previous monitoring surveys as part of compliance reports. Even with the above data, it was clear that a one off survey of drains in the Kaituna and Rangitaiki Plains was necessary, so an additional 20 sites were selected from where invertebrates were collected in the 2015–2016 austral summer.

At each of the drain sites, invertebrate samples were collected using standard methodologies for sampling in soft bottomed streams (Stark et al. 2001). All samples were sent to EOS Ecology in Christchurch where they were processed by counting and identifying the first 200 invertebrates, and by scanning the remainder of the sample for unrecorded taxa. This method provides robust information about the nature and composition of the invertebrate community, and is much cheaper than the more time-consuming process of identifying all the invertebrates collected (Duggan et al. 2003).

The new data from the 2015-2016 survey was combined with data from BOPRC's NERMN programme to give a total of 438 sites for potential comparison. All of these were assigned to their appropriate water quality classification as per the RNRP. Because of the strong effect of land cover on stream communities (e.g., Suren et al 2017) only sites draining agricultural or urban catchments were thus selected for further analyses. Furthermore, only sites < order six were selected, as many indices were not specifically developed for larger streams. This selection processes resulted in a total of 168 sites being used for all subsequent analysis and comparison between ecological conditions in DWQ and MEV waterways, and other waterways in the region. Examination of this data showed a total of 45 sites came from waterways classified as either DWQ (23 sites) or MEV (22 sites each), all of which were in the Kaituna (Figure 20) and Rangitaiki Plains (Figure 21). All of these sites were from catchments dominated by either agricultural (37 sites) or urban (eight sites) land use, as assessed using the REC land cover classes.

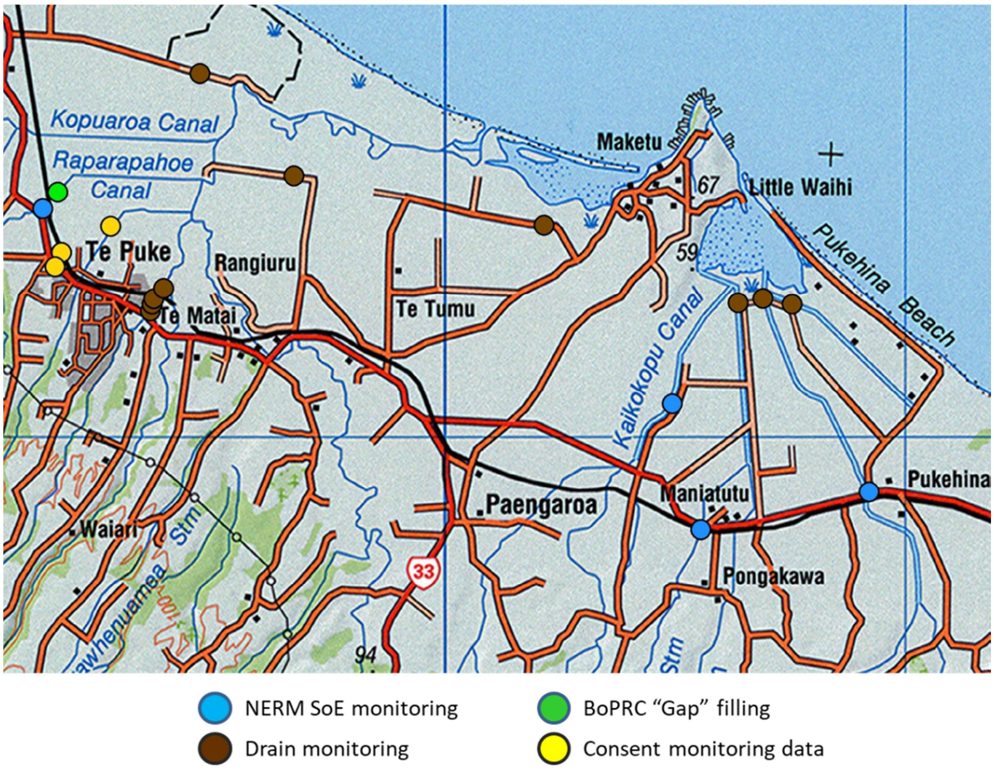

Figure 20 *Map showing the location of sampling sites to characterise the drain communities in the Kaituna Plains, classified according to the study source.*

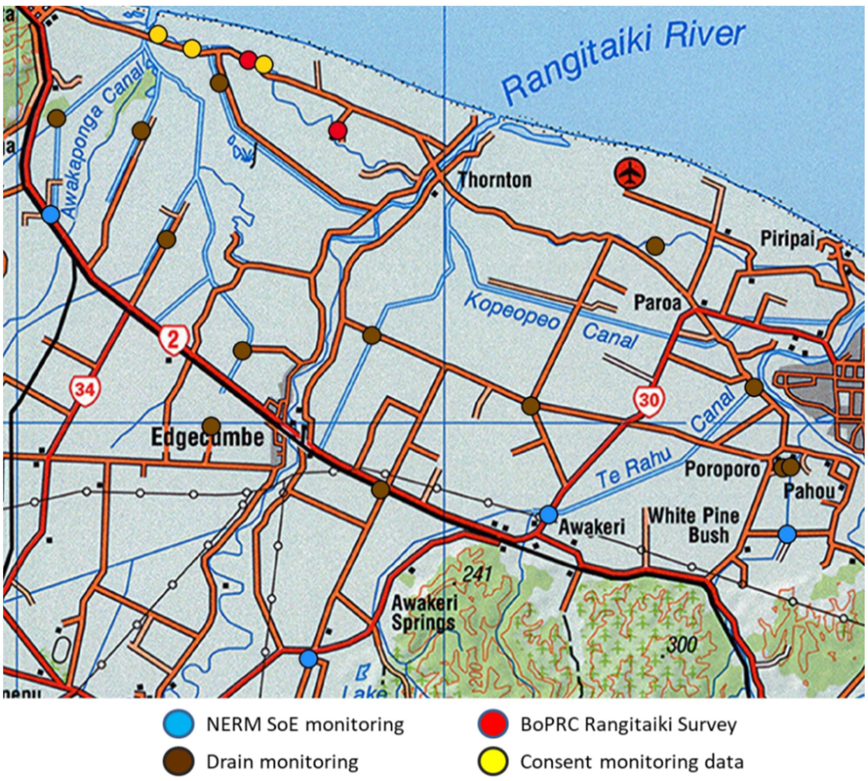

Figure 21 *Map showing the location of sampling sites to characterise the drain communities in the Rangitaiki Plains, classified according to the study source.*

4.2.2 Statistical analysis

Calculation of metrics

All invertebrate data was entered in the regional council's master invertebrate spreadsheet, and a number of biological metrics were calculated to describe the condition of the invertebrate community found in the lowland drainage network of the Kaituna and Rangitaiki Plains. These metrics included the macroinvertebrate community index (MCI), and its quantitative variant, the QMCI. The MCI and QMCI are used to assess the overall ecological condition of a stream, with MCI scores greater than 120 indicating streams in excellent condition, and scores less than 80 indicating streams in poor ecological condition. For the QMCI score, the same bands occur with scores > 6.0, and < 4.0 respectively. The number and percentage of sensitive mayfly, stonefly and caddisflies (EPT) were also calculated. Sites in good ecological condition generally have high values of the EPT indices, whereas values of these indices decrease in sites with lower ecological condition.

Another biotic metric, the Bay of Plenty Index of Biotic Integrity (BoP_IBI) was also calculated for all waterways in the Kaituna and Rangitaiki Plains. This metric was developed to describe the condition of invertebrate communities at sites throughout the region (Suren 2017), and compares observed values of individual metrics at a test site to those at appropriate reference sites, selected to represent the least disturbed condition. The BoP_IBI has been divided into four classes to describe the overall ecological conditions at a site (Suren et al. 2017), and responds in a more predictable manner to stressors associated with land use change and other metrics such as the MCI or QMCI. Following the calculation of these metrics, the following analyses were done on the data.

Comparisons to other waterways

Firstly, the ecological condition of the invertebrate communities found within the lowland drainage network of the Kaituna and Rangitaiki Plains was compared to that of other waterways draining catchments dominated by agricultural or urban land use throughout the region. A total of 168 sites were selected for this analysis (Table 12). General Linear Model (GLM) analysis was used to determine whether the calculated metrics differed between the different waterway classes.

Table 12 Summary breakdown of the number of sites of different water quality classification in catchments dominated by agriculture or urban landuse.

WQ Classification	Agriculture	Urban	Total
Aquatic Ecosystem	79	7	86
Drain Water Quality	16	6	22
Modified waterways with Ecosystem Values	21	2	23
Regional Base Line	20	8	28
Other	9	0	9

Assessment against recommended attribute state bands

Following this regional comparison of waterway health, the data from the lowland waterways of the Kaituna and Rangitaiki Plains was examined to determine the appropriate banding level each waterway belonged to.

Again, this analysis was restricted only to sites in catchments dominated by draining agricultural or urban land cover. Carter et al. (2017) proposed numerical attributes for different water quality and ecological parameters, including biotic metrics to summarise invertebrate communities. Bands were developed for MCI, the number of EPT taxa, and the BoP_IBI for streams flowing in each of the two biophysical classifications encountered; volcanic steep and volcanic gentle (Table 13). As with the water quality, the ecological attributes and their bands are not applicable to DWQ sites, only the MEV sites.

Table 13 Summary of the band categories for the invertebrate metrics used to summarise stream ecological health.

Annual monitoring	Band	Volcanic gentle	Volcanic steep	Narrative
MCI scores	A	> 124	> 115	MCI scores typical of healthy and resilient invertebrate communities, similar to natural reference conditions. Indicative of streams in excellent ecological condition.
	B	106 - 124	100 - 115	MCI scores show slight reductions, suggesting loss of some potentially sensitive taxa from what would be expected in a similar reference condition stream. Indicative of streams in good ecological condition.
	C	88 – 106	87 – 100	MCI scores show moderate impacts, with a more noticeable reduction in the majority of sensitive taxa from what would be expected in a similar reference condition stream. Indicative of streams in fair ecological condition.
	D	< 88	< 87	Reduction in MCI scores show large detrimental impacts, with a loss of all sensitive taxa from what would be expected in a similar reference condition stream. Indicative of streams in poor ecological condition.
EPT richness (Number of taxa)	A	> 11 EPT	> 9 EPT	The number of sensitive EPT taxa typical of those found in reference condition streams.
	B	7 – 11 EPT	6 – 9 EPT	Streams showing a slight reduction in the number of sensitive EPT taxa that are typically found in similar reference condition streams.

Annual monitoring:	Band	Volcanic gentle	Volcanic steep	Narrative
EPT richness (Number of taxa)	C	2 – 7 EPT	3 – 6 EPT	Streams showing a moderate reduction in the number of sensitive EPT taxa that are typically found in similar reference condition streams.
	D	< 2 EPT	< 3 EPT	Streams showing a large reduction in the number of sensitive EPT taxa that are typically found in similar reference condition streams.
BoP_IBI	A	> 47	> 18	Streams supporting a range of invertebrate species that are very similar to those found in reference condition streams.
	B	36 - 47	7 - 18	Streams supporting a slightly reduced range of invertebrate species that would be expected in similar reference condition streams.
	C	26 - 36	3 - 7	Streams supporting a moderately reduced range of invertebrate species that would be expected in similar reference condition streams.
	D	< 26	< 3	Streams supporting a greatly reduced range of invertebrate species that would be expected in similar reference condition streams.

Multivariate analyses

The third analysis examined what factors were responsible for the observed invertebrate community structure in the lowland drainage network, and other waterways throughout the region. Data summarising environmental factors such as habitat condition (using the RHA) and water quality were obtained from various sampling programmes, including the NERMN invertebrate and water quality programmes, as well as data gleaned from a gap filling programme identified in the Kaituna and Rangitaiki Water Management Areas (Suren et al 2015; Carter et al 2015). A total of 62 sites which had assessments of both RHA and water quality were consequently selected or this analysis.

This analysis was done in two stages, involving both environmental and ecological data. Firstly, to reduce the complexity of the environmental data (RHA scores and 10 water quality parameters from 62 sites), a Principal Components Analysis (PCA) was used to reveal any structure in the data and to identify what the major environmental differences were between sites. Prior to the PCA, all environmental factors were standardised so that measures with different units could be analysed together, and then log (x+1) transformed to ensure normality. 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 one and two scores.

Following the PCA, a similarity matrix was calculated to show the similarity of all sites to each other based on their environmental data. The euclidean distance measure was used for this analysis, which measures the straight-line distance between samples, and is appropriate for physical data. Thus, for example, consider three sites, A, B and C. If Sites A and B were small natural streams, far inland, and dominated by native bush, and Site C was a large modified drain in the Kaituna Plains flowing through a catchment dominated by pasture, then the Sites A and B would have a very small euclidean distance measure as all environmental factors would be similar. However, there would be a greater euclidean distance between sites A and C, and B and C, reflecting the fact that site C was different to the other sites. The resultant similarity matrix for all 62 sites thus summarised the similarity of all sites to each other, based on their environmental data. This similarity matrix was used to compare to a second similarity matrix that was created based on the invertebrate data (see below).

Having two similarity matrices allowed us to see how well relationships in the ecological data matrix matched up with the patterns in the environmental data matrix.

For the ecological data, invertebrate relative abundances were fourth root transformed to ensure that the data had a normal distribution, and ordination (non-metric multidimensional scaling: NMDS) used to examine and explore relationships and patterns in the data. Ordination is a statistical method used in exploratory data analysis to search for patterns in the data, such as being done here, rather than in testing specific hypothesis. It orders objects (in this case individual sampling sites) that are characterised by values of multiple variables (in this case the relative abundance of invertebrate taxa at each site) so that similar sites are located near each other on an x-y graph, and dissimilar sites are located farther from each other. The first step in this ordination analysis was to calculate a similarity matrix of all sites to each other. The Bray-Curtis similarity measure was used for this analysis. This measure results in scores ranging from zero (i.e., two sites having no species in composition) to one (i.e., two sites having exactly the same species composition).

An NMDS ordination was then run on this similarity matrix to examine relationships between all the individual sites. NMDS produces a statistical score (called stress) that indicates the strength of the resultant ordination. Stress values greater than 0.3 indicate the resultant sample configurations are no better than arbitrary (i.e., there are no underlying patterns to the invertebrate community composition at each site). This would occur where invertebrate communities did not differ greatly between the different streams. Under such a scenario, no differences to invertebrate communities would be expected between unmodified streams flowing through native forest or communities in the highly modified drains flowing through the Kaituna or Rangitaiki Plains. Generally speaking, sample configurations should not be interpreted unless the stress value is less than 0.2 (Clarke and Gorley 2001), while a stress > 0.3 is equivalent in having the points being randomly placed, and thus having no discernible pattern. The NMDS ordination thus identifies major gradients in the data, with the x-axis representing the greatest difference between samples, and the Y axis representing the second greatest difference. Analysing correlations of both species distribution and environmental variables against these axes allows us to determine which species and environmental variables were responsible for the observed gradients in the data.

Following the NMDS analysis, Analysis of Similarities (ANOSIM) was used to measure the degree to which the grouping of samples according to their water quality classification explained the observed variability of the invertebrate communities. Analysis of Similarities produces a statistic (called Global R) that is a measure of the degree to which a predictor variable (in this case, the water quality classification groups) explained variability to the invertebrate data. A high global R means that a particular grouping variable explains a lot of the observed variation in the data. Significant differences were set at a level of 0.1%.

Finally, the similarity matrices developed from the environmental and ecological data were examined to determine how well the relationships between sites matched each other. For this analysis we used the Relate command in Primer (Ver 6.0), which calculates the spearman rank correlation of the similarity matrices based on environmental or ecological data. If invertebrate communities were structured by the derived environmental variables, then we would expect a strong correlation between the two similarity matrices, whereas if these communities were responding to other non-measured environmental variables, then such strong correlations would not exist. Following this analysis, we examined relationships between environmental variables and invertebrate communities using the Best procedure. This procedure determines which environmental variables were responsible for any observed patterns to the invertebrate data. Because of the large number of environmental variables (13), we used a stepwise approach for this analysis, whereby the Best procedure iteratively added or removed variables and selected only those which explained the highest degree of variation to the macroinvertebrate communities.

This analysis was complemented by a regression analysis of environmental variables against the NMDS axis scores. Both the Best and regression analysis enabled us to determine which of the environmental variables were responsible for structuring invertebrate communities in the lowland drainage network in the Kaituna and Rangitaiki Plains. The inclusion of other sites was done to create a larger environmental gradient, as comparisons only amongst highly degraded systems such as the drainage network alone would be unlikely to detect what the dominant driving variables were, as all sites would be at the extreme end of a disturbance spectrum associated with land use change.

4.3 Results

4.3.1 The invertebrate communities

Invertebrate data was obtained from 45 low elevation sites in the Kaituna and Rangitaiki Plains; 23 of which classified as DWQ and 22 as MEV. A surprisingly diverse fauna of 100 taxa was recorded from these sites. This fauna was dominated by the common mudsnail *Potamopyrgys*, chironomid midges, and members of the shrimp-like crustacean family Tanaidicea.

Other common invertebrates found in the drains were the purse-case caddisfly *Oxyethira*, the water boatman *Sigara*, worms, leeches and flatworms, and ostracods (Figure 22).

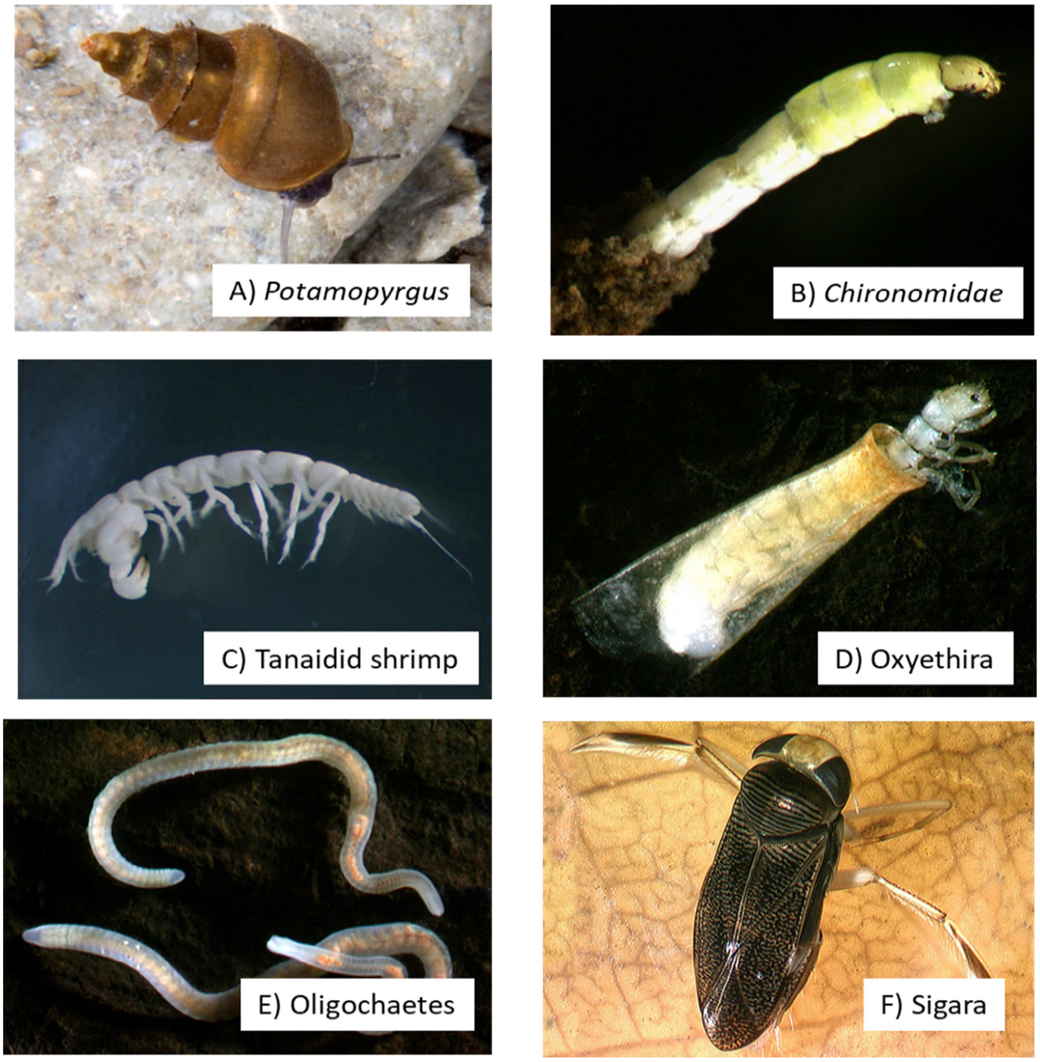

Figure 22 *Examples of aquatic invertebrates commonly collected from waterways throughout the Kaituna and Rangitāiki Plains. Photos courtesy of Landcare Research Limited.*

Calculated biotic indices from the sampled sites were all relatively low, and indicative of degraded conditions. Average MCI scores were only 74, and average QMCI scores only 3.4; both indicating degraded water quality conditions (Stark and Maxted 2007). Despite these low scores, high MCI scores were observed at some sites (two sites on the Raparapahoe Drain, and the Waioho Stream: Appendix 1). Both the number of EPT taxa, as well as the percentage of EPT taxa were also very low (Table 14) at most of the 45 sites, although there were some sites where many of these sensitive invertebrates were found.

Table 14 Summary statistics for the five biotic metrics calculated to describe invertebrate communities collected from the 45 low elevation MEV and DWQ sites in the Kaituna and Rangitaiki Plains.

Metric	Minimum	Average	Maximum
MCI	24	73.7	133
QMCI	1.2	3.4	7.5
EPT richness	0	3.0	26
% EPT richness	0	10.8	66.6
% EPT abundance	0	6.6	92.5

Calculated BoP_IBI scores were also very low at most sites, with 25 sites being classed as in poor ecological condition, and 14 sites as being in fair condition. As with the other biological metrics, five waterways had excellent BoP_IBI scores: the Waiari, Kaikokopu, Pongokawa, Raparapahoe and Puanene Streams.

4.3.2 Regional comparisons

The 45 sites sampled in the Kaituna and Rangitaiki Plains belong to the water quality classification of either DWQ or MEV. The ecological health of these waterways was compared to that in other waterways throughout the region when assigned to their appropriate water quality classifications. All of the seven biotic metrics differed between the water quality classifications (Table 15) and observed values were lowest for all metrics in both the DWQ and MEV waterways, often considerably. Although MCI scores were similar between DWQ and MEV waterways, MEV waterways had significantly lower QMCI scores than DWQ sites, or sites in other water quality classifications (Figure 23). The only EPT taxa found in small DWQ drains in the Lawrence Oliver Park near Te Puke, where the caddisflies *Polyplectropus*, *Triplectides*, *Hudsonema* and *Oxyethira* were found. *Polyplectropus* was found at all four sites here, and these larvae are common throughout the country in slow flowing soft bottomed streams with woody debris – typical of the habitats there. However, no EPT taxa were found in any of the DWQ waterways draining agricultural catchments. Both the number and percentage of EPT taxa was also very low in waterways assessed as MEV, and much lower than in other waterway classifications (Figure 23). The percentage of EPT to total abundance was also very low in MEV waterways: much lower than in other classifications (Figure 24). Both DWQ and MEV waterways supported the fewest number of taxa – often considerably (Figure 24). Finally, the calculated BOP_IBI scores were considerably lower in DWQ sites, but were of intermediate value in MEV sites, and highest in sites assessed as AE (Figure 24).

Table 15 Results of one-way GLM examining for differences in calculated biotic metrics between water quality classes in 168 waterways draining agricultural or urban catchments, less than order six.

Biotic Index	Source	Sums of squares	DF	Mean Squares	F-ratio	p_Value
MCI	Water quality class	41 129.1	4	10 282.3	27.9	0.000
	Error	59 998.7	163	368.1		
QMCI	Water quality class	88.7	4	22.2	10.3	0.000
	Error	350.6	163	2.2		
EPT richness	Water quality class	3997.2	4	999.3	20.3	0.000
	Error	8032.8	163	49.3		
% EPT Richness	Water quality class	38145.7	4	9536.4	29.3	0.000
	Error	53040.5	163	325.4		
% EPT density	Water quality class	37482.8	4	9370.7	13.4	0.000
	Error	114349.7	163	701.5		
Taxon richness	Water quality class	13312.9	4	3328.2	14.3	0.000
	Error	37902.2	163	232.5		
BOP_IBI	Water quality class	8037.6	4	2009.4	15.2	0.000
	Error	21014.1	159	132.2		

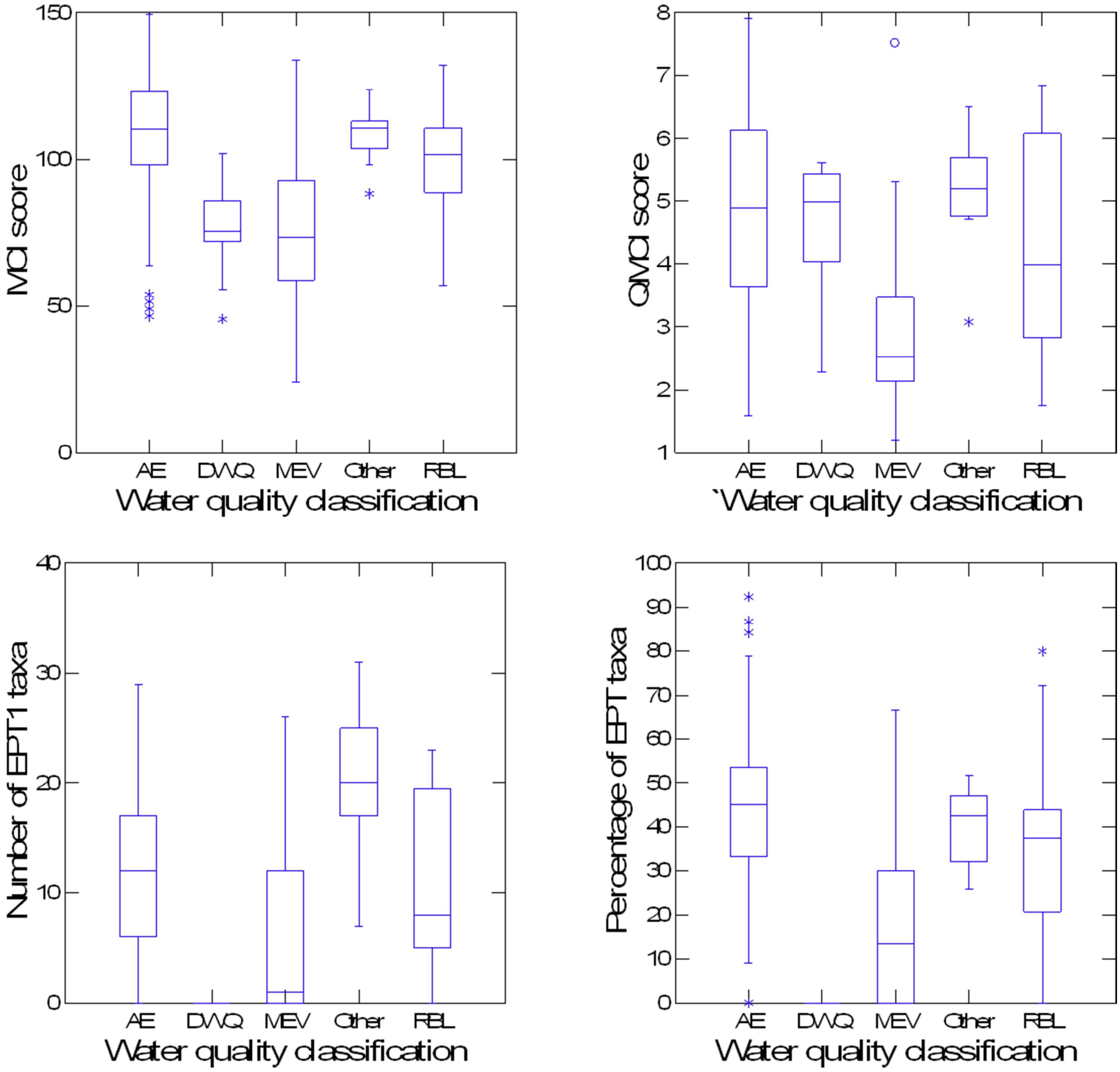

Figure 23 *Box plot of calculated MCI and QMCI scores, the number of sensitive Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa, and the percentage of EPT to total taxonomic richness collected from 168 waterways draining catchments dominated by either agriculture or urban land cover throughout the region and assigned to water quality classes as per the RWLP. AE = Aquatic Ecosystem; DWQ = drain water quality; MEV = Modified watercourses with Ecological Values; RBL = Regional Base Line; Other = a combination of Contact recreation, Fish spawning purposes, Natural state and Water supply.*

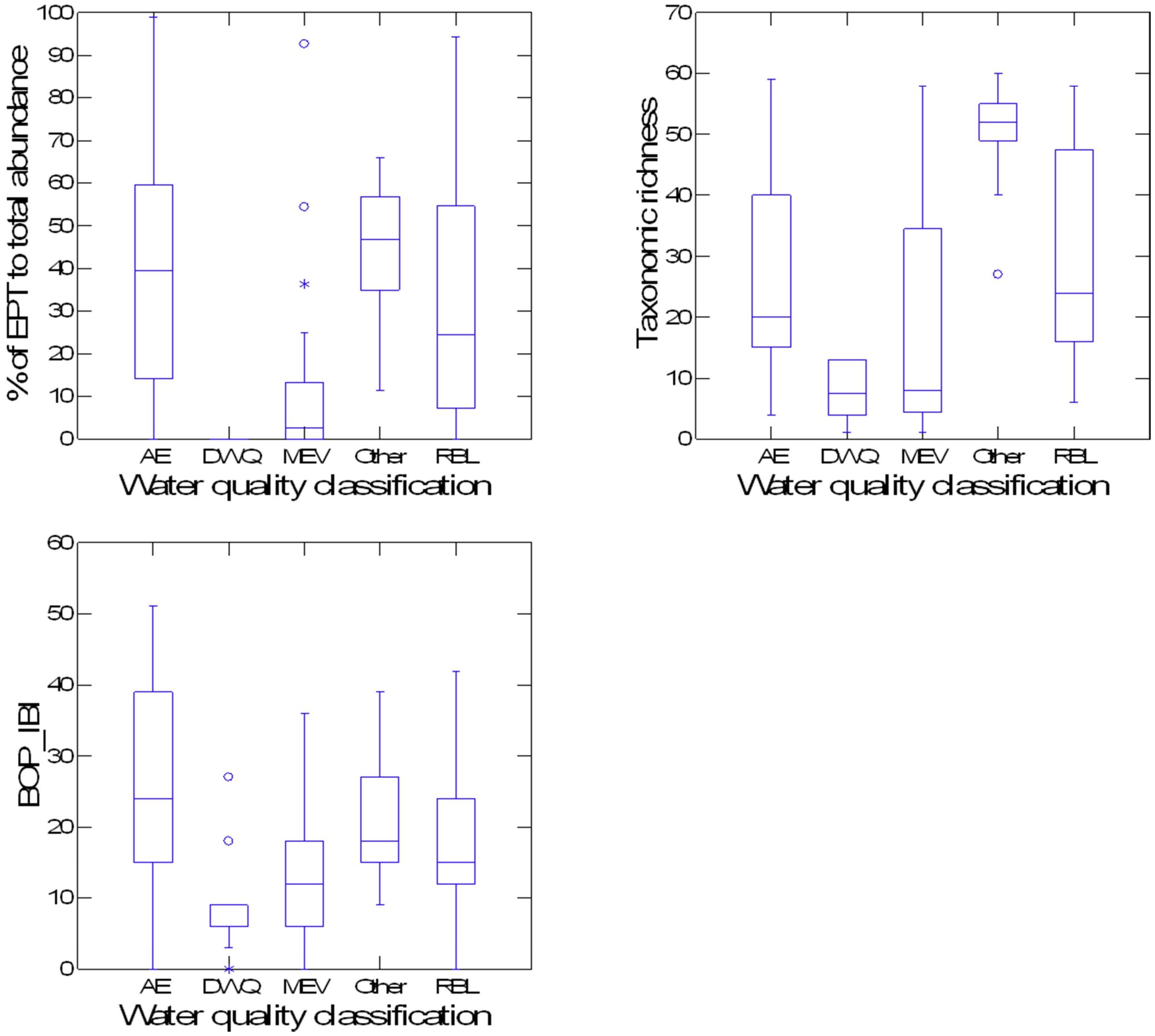

Figure 24 Box plot of the % of sensitive EPT to total abundance, the number of taxa in each sample (i.e., Taxonomic Richness) and the calculated Bay of Plenty Index of Biotic Integirty (BoP_IBI) in samples collected from 168 waterways draining catchments dominated by either agriculture or urban land cover throughout the region and assigned to water quality classes as per the RWLP. Conventions see Figure 23.

4.3.3 Assessment according to attribute state bands

Calculated biotic metrics for the MCI, EPT1-richness and the BoP_IBI were compared to the proposed ecological bands of Carter et al (2017). These bands were developed for streams in three different biophysical classes; non-volcanic, volcanic steep-gradient and volcanic gentle-gradient (See Snelder et al 2016 and Suren 2017 for further information). Only the volcanic steep gradient and volcanic gentle-gradient were considered below, as only a single non-volcanic site was sampled. The individual numerical values for MCI, EPT richness and the BoP_IBI were calculated for streams in each of these biophysical units and allocated to their appropriate band. The raw data for each site is presented in Appendix 1.

Large differences existed in the percentage of sites allocated to each of the four bands in the different water quality classifications. For all three metrics examined, the majority of DWQ and MEV sites were in the D band (Table 16), although the majority of sites classified as aquatic ecosystem (AE), or regional baseline (RBL) were also in the D band for the BoP_IBI. For both the MCI and EPT richness indices, the majority of sites were either in the A, B or C band for the non-drain other water quality classifications (Table 16).

Table 16 Summary of the percentage of sites when assessed according to ecological bands for MCI, EPT1 richness, and BoP_IBI (See Table 13) in each of the five water quality classifications. Shaded boxes show the band with the highest percentage in each water quality classification.

	WQ_Class2	A	B	C	D
MCI	AE	26.1	31.8	21.6	20.5
	DWQ	0.0	0.0	5.3	94.7
	MEV	4.8	0.0	33.3	61.9
	Other	12.5	37.5	50.0	0.0
	RBL	3.7	25.9	48.1	22.2
EPT_Richness	AE	50.0	12.5	22.7	14.8
	DWQ	0.0	0.0	10.5	89.5
	MEV	33.3	4.8	9.5	52.4
	Other	87.5	12.5	0.0	0.0
	RBL	33.3	25.9	29.6	11.1
BoP_IBI	AE	11.4	30.7	18.2	39.8
	DWQ	0.0	0.0	5.3	94.7
	MEV	0.0	9.5	9.5	81.0
	Other	0.0	25.0	50.0	25.0
	RBL	7.4	11.1	25.9	55.6

All three indices differed between the five water quality classifications found in the volcanic gentle gradient biophysical class (Table 17). Here, streams in the DWQ or MEV classes had lower scores for each of the three attributes that summarise ecological health (Figure 25). In contrast, only EPT richness differed between the four water quality classifications found in the volcanic steep gradient biophysical class (Table 17). Here, EPT richness was lowest in MEV and AE waterways, intermediate in RBL waterways, and highest in waterways classified as other(Figure 26). Median values for MCI scores and the BoP_IBI were not however significantly different between waterway classes in this biophysical unit.

Table 17 Results of one way GLM examining for differences in calculated biotic metrics between water quality classes in each of the two biophysical units for waterways draining agricultural or urban catchments, less than order six. Significant effects in highlighted in bold.

Biophysical Unit	Biotic index	WQ class	Sums of squares	df	Mean squares	F-Ratio	p-Value
VA_Gentle	BOP_IBI	Water quality class	11240.5	4	2810.1	**20.93**	**0.000**
		Error	14915.3	111	134.4		
	EPT_richness	Water quality class	2687.3	4	671.8	**15.48**	**0.000**
		Error	4815.6	111	43.4		
	MCI	Water quality class	34537.7	4	8634.4	**23.31**	**0.000**
		Error	41111.5	111	370.4		
VA_Steep	BOP_IBI	Water quality class	60.2	3	20.1	1.05	0.38
		Error	820.7	43	19.1		
	EPT_richness	Water quality class	486.8	3	162.3	**3.00**	**0.041**
		Error	2323.1	43	54.0		
	MCI	Water quality class	998.9	3	332.9	1.21	0.318
		Error	11854.1	43	275.7		

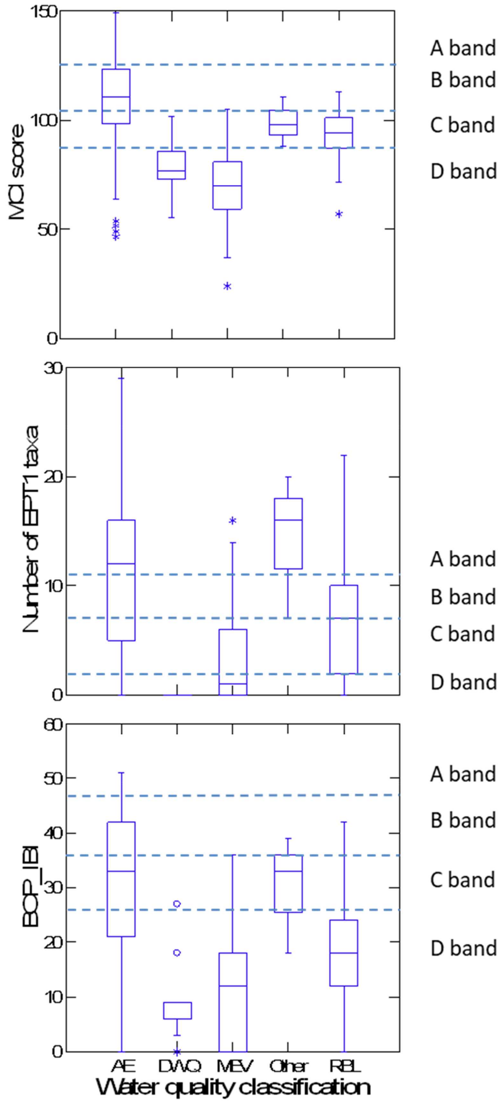

Figure 25 *Box plot showing differences in MCI, EPT richness, and the BoP_IBI between common water quality classification stream types in volcanic gentle gradient streams throughout the region. Also shown are the proposed bands for these metrics as outlined in Carter et al. (2017). Note the D band for MCI is at 80.*

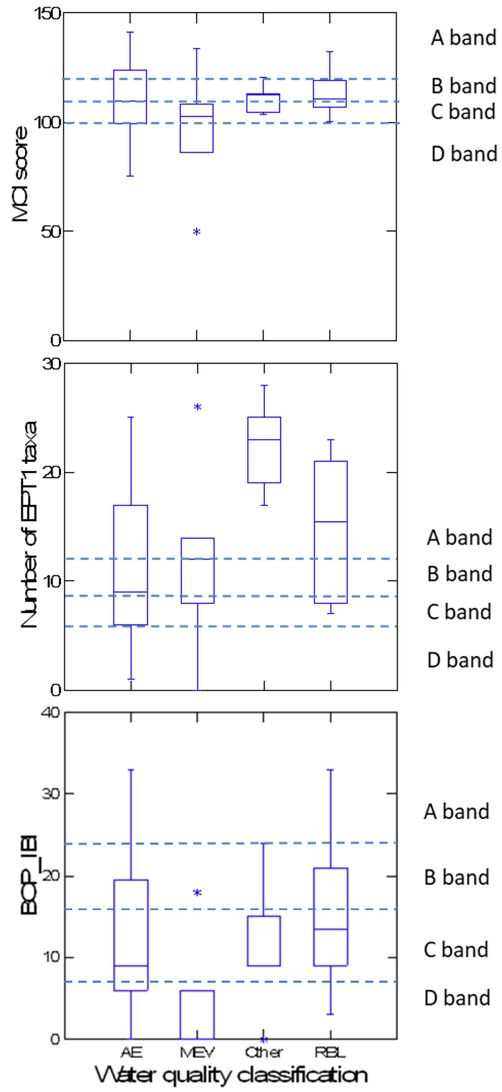

Figure 26 *Box plot showing differences in MCI, EPT richness, and the BoP_IBI between common water quality classification stream types in volcanic steep gradient streams throughout the region. Also shown are the proposed bands for these metrics as outlined in Carter et al. (2017).*

4.3.4 Multivariate analyses

The first two axes of the PCA explained 38.2% and 19.0% of the variation in the environmental data matrix, and thus collectively explained 57.2% of the total variation. The PC1 axis represented a gradient in % dissolved oxygen, total suspended sediments, ammonia_N and DRP (Figure 27). Thus, sites with high PC1 scores had high levels of dissolved oxygen, and low levels of total suspended sediments, ammonia_N and DRP, and vice versa. PC2 represented a gradient in NOx-N, RHA Habitat score and conductivity. Thus, sites with low PC2 scores had high concentrations of NOx_N, low conductivity and poor habitat conditions (Figure 27).

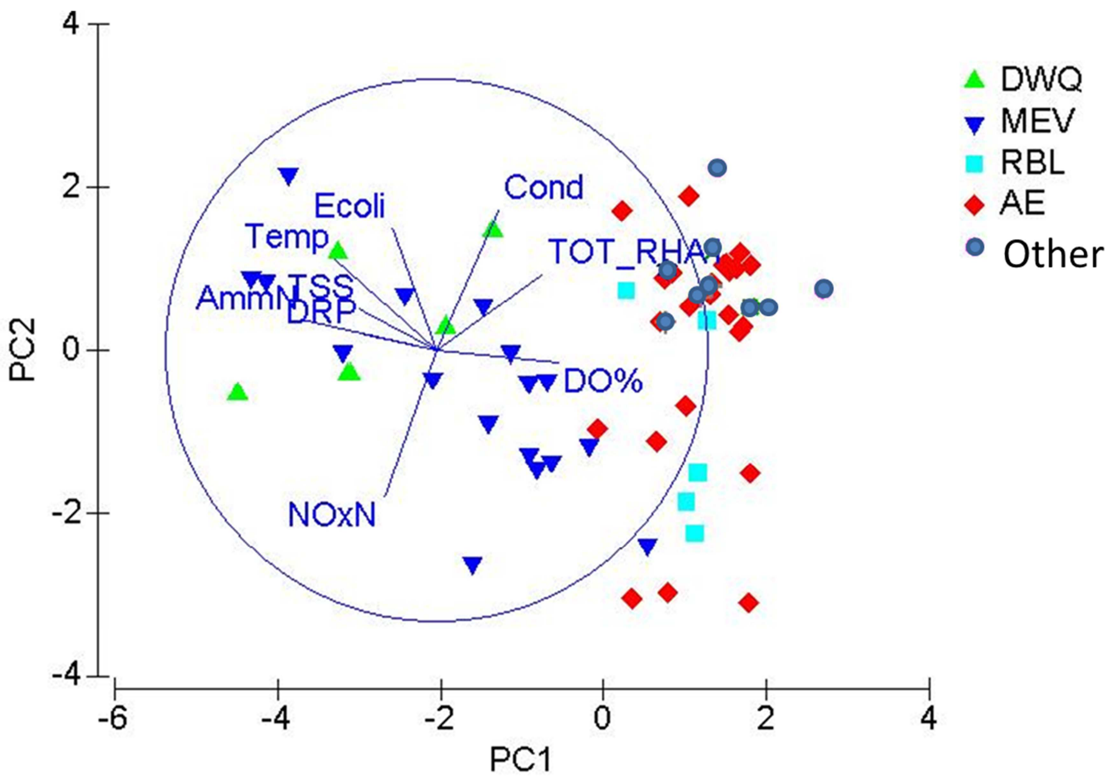

Figure 27 *Two dimensional PCA ordination of the 10 water quality and RHA scores for 62 sites collected from throughout the region, showing how these factors differed between sites when coded according to the Water Quality Classification.*

The NMDS ordination of the invertebrate data had a stress of 0.13, suggesting that there were very distinct patterns in the data. When coded by the WQ classification, there were distinct groupings according to the WQ classes in the NMDS plot (Figure 31). Thus, samples collected from both DWQ and MEV sites had much higher NMDS Axis one scores than samples form the other WQ classes. Samples from the MEV sites also had slightly lower NMDS axis two scores as well. Samples from all other waterways were generally clustered in the upper left quadrat of the graph (Figure 28), emphasising that they shared a common assemblage of invertebrate taxa which was generally quite distinctive to that from the lowland sites in the Kaituna and Rangitāiki plains.

These results were mirrored by the ANOSIM, which indicated that there was a significant difference between the five water quality classes (global R-statistic =0.508). Examination of the pairwise R-statistics showed that invertebrate communities from DWQ sites were the same as from MEV sites (R = 0.003, P = 1.4), but different to everywhere else (R-statistic >0.90). Communities from MEV sites were also similar to those from RBL sites (R = 0.293, P = 2.4), but differed to sites classified as AE and Other. These results highlight that the invertebrate community composition of both the DWQ and MEV sites is generally very distinctive to those from other waterways.

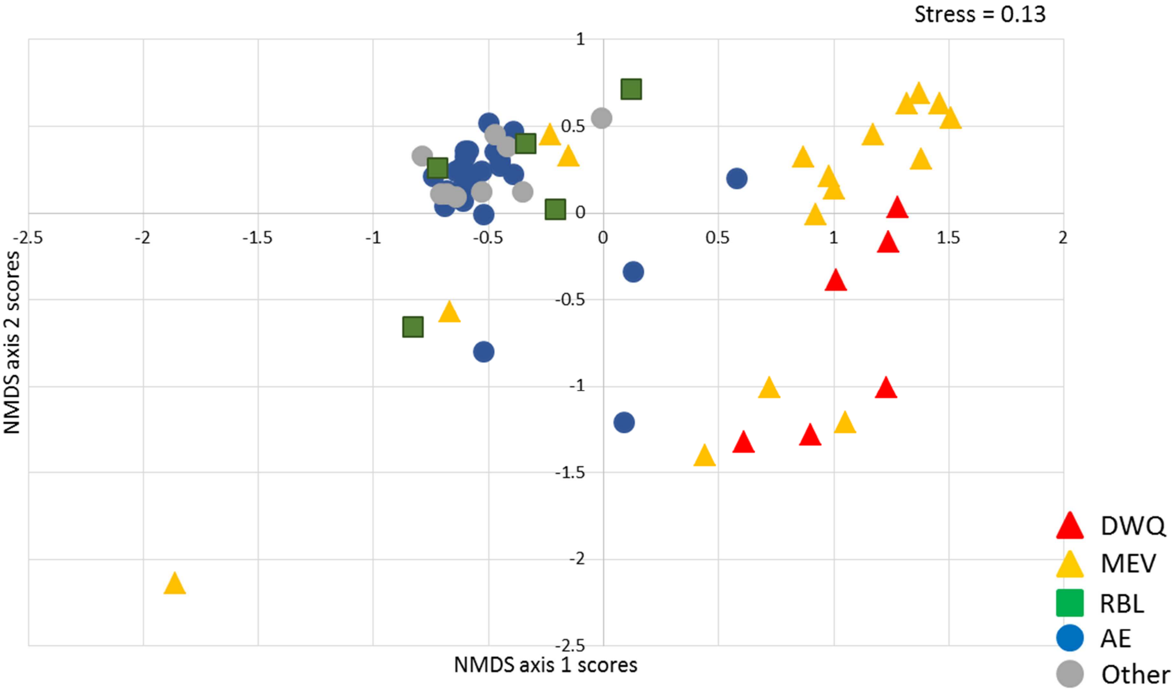

Figure 28 *Two dimensional NMDS ordination of the 62 sites collected from throughout the region based on their invertebrate community composition, showing how these factors differed between sites when coded according to the Water Quality Classification.*

Results of the Relate Analysis showed a significant Rho value (0.47, $P < 0.001$), suggesting relatively strong similarities between the resemblance matrices of the environmental factors and invertebrate communities. This means that the invertebrate communities were in fact structured by the measured environmental factors. Results of the Best Analysis showed that the main structuring variable responsible for observed patterns to the invertebrate data was concentrations of ammonia-N. This is consistent with the finding that ammonia-N concentrations were one of the main factors identified in the PCA analysis that separated DWQ and MEV sites from all the others along a the PC1 axis (Figure 27).

4.4 Discussion

Although a diverse invertebrate fauna was encountered in the waterways draining the Kaituna and Rangitaiki Plains, the ecological condition of these waterways as assessed using invertebrates was generally very low (See Appendix 1 for details of each site). For example, the average MCI score in the DWQ and MEV sites was only 74., well below the MCI score of 80 at which, for natural or modified natural watercourses, councils must investigate causes and take action to address (under the most recent amendments to the NPSFM). Observed metrics were generally in the D band based on the attributes in the Carter et al. (2017) report, especially in waterways classified as DWQ. Of interest was the observation that waterways classified as MEV were generally in the D band in the volcanic gentle gradient biophysical unit, whereas median scores for these waterways in the volcanic steep-gradient biophysical unit were generally higher and differed between waterways classes only for the BoP_IBI. This highlights the importance of putting observed biological metrics into their appropriate biophysical classification, as this classification provides a more context specific way to assess stream health.

The invertebrate fauna of both the DWQ and MEV waterways was dominated by a core assemblage of taxa which could tolerate the environmental conditions characteristic of these sites.

These conditions include a lack of instream habitat diversity, low flows, and lack of shade (see section 2), as well as poor water quality characterised by high water temperatures, low to very low dissolved oxygen levels, and high nutrient levels (see section 3). Many of the sensitive EPT taxa are absent from these waterways: indeed 28 of the 45 waterways examined supported no EPT. The small purse-case caddisfly Oxyethira was often the only EPT found in many of the sites. This animal is highly tolerant of degraded habitat and water quality conditions, and is often found in urban streams (Hall, Closs et al. 2001, Suren and Elliot 2004), so its occurrence in the waterways sampled is no surprise. However, what was surprising was the absence of even this highly tolerant caddisfly from 28 of the sites classified as either DWQ or MEV. This invertebrate is commonly found in urban streams which are characterised by a mix of highly degraded habitat and water quality conditions, so their absence highlights the degree to which these ecosystems are stressed.

This drain fauna is fundamentally different to that of other waterways throughout the region. Suren et al (2017) found that the invertebrate communities of the 130 sites monitored as part of the council's NERMN programme contained 135 different taxa, and that 90% of sites contained a core assemblage of just nine taxa. These included three mayflies (*Zephlebia*, *Deleatidium*, and *Austroclima*), and two caddisflies (*Aoteapsyche* and *Pycnocentrodes*). The three mayflies in particular are regarded as being relatively intolerant of highly enriched conditions, warm water, slow flow and silty substrates – characteristics common in the drains. The highly degraded nature of the invertebrate communities in the drainage network is likely to reflect many stressors, acting alone or in combination with each other. For example, the importance of habitat quality in structuring invertebrate communities was clearly evident in section 2 (see Figure 8), whereby sites with low RHA scores generally supported more degraded invertebrate communities. Other factors such as poor water quality (Section 3) may also be responsible for the degraded ecological condition of many of the waterways. Indeed, our analysis highlighted the importance of ammonia concentrations as one factor implicated in structuring the invertebrate community composition.

Other less obvious factors could also include a lack of suitable riparian vegetation around the drainage network to provide both shade and habitat for adult phases of aquatic insects. Riparian vegetation has numerous well documented benefits to waterways (Collier, Cooper et al. 1995, Parkyn 2004), yet many invertebrates only have limited ability to colonise small isolated plantings of riparian vegetation in catchments dominated by agricultural or urban land use (Suren and McMurtrie 2005, Parkyn and Smith 2011). The slow flowing and generally soft bottomed nature of the drainage network would also result in degraded invertebrate communities, as many aquatic insects have strong habitat preferences for non-silty substrates (Death 2000).

The above analysis was restricted only to streams draining catchments dominated by agriculture and urban land use. Stream ecological health is generally lower in these catchments than in catchments dominated by native bush or exotic plantation forest, so the analysis could be regarded as conservative, as streams with higher likely ecological health were omitted. Including these streams would further emphasise the low ecological health of the DWQ and MEV waterways.

In the 2017 amendment to the NPS-FM Policy CB3 states that regional councils need to establish methods to investigate the causes of declining MCI scores and scores below 80. Councils are required to attempt to halt these declining trends and/or improve to exceed 80, unless the low ecological health is caused by a naturally occurring process, or by infrastructure as listed in Appendix 3 (which is currently not populated). Although it could be argued that Policy CB3 does not apply to many of the waterways sampled as they are classified as DWQ, MCI scores below 80 were also found in many waterways classified as MEV. This finding may have important implications as to the continued management of these waterways if their condition is deemed to be unacceptable.

More work is thus needed to better determine causal features for the observed low ecological health in the MEV waterways, and determine what realistic mitigation measures can be employed to improve this.

Part 5:

Fish communities

5.1 Introduction

Freshwater fish are one of the important ecological values of waterways. Wetlands in the Kaituna and Rangitaiki Plains provided a bountiful food supply to iwi prior to land drainage and conversion into the current productive agricultural land. Important fish likely to be found within the original wetlands would have been tuna (both shortfin and longfin), inanga, and species such as banded and giant kokopu. Marine wanderers such as kahawai, mullet, parore and kingfish would also have occasionally foraged into the lower reaches of the main rivers in search of smaller freshwater fish for prey.

Extensive drainage, dredging and channel straightening occurred in the 1920s to the 1950s. This work resulted in loss of the original habitat, but also resulted in land use intensification that caused increases in nutrient and sediment inputs. Many native fish also require free access to and from the sea, and this has now been interrupted by a series of flood gates and/or pumping stations (e.g.,Figure 29; Figure 30). The former structures usually allow for only the one-way passage of water out from the drains on the receding tide, and are generally shut on the incoming tide. This means that upstream fish passage for migrating native fish in particular may be disrupted.

Pumps and pumping stations are common throughout the Kaituna and Rangitaiki Plains. These infrastructure devices have been shown to have adverse effects on the successful downstream migration of large, sexually mature eels in the Waikato (Vaipuhi Consulting 2017). Many of the pump stations in the Kaituna and Rangitaiki Plains have fish screens in front of them, so migrating eels may not suffer the same mortality when they pass through pump stations as observed in the Waikato study. However, eels still need to be able to pass around the screens to successfully migrate back to the sea to breed.

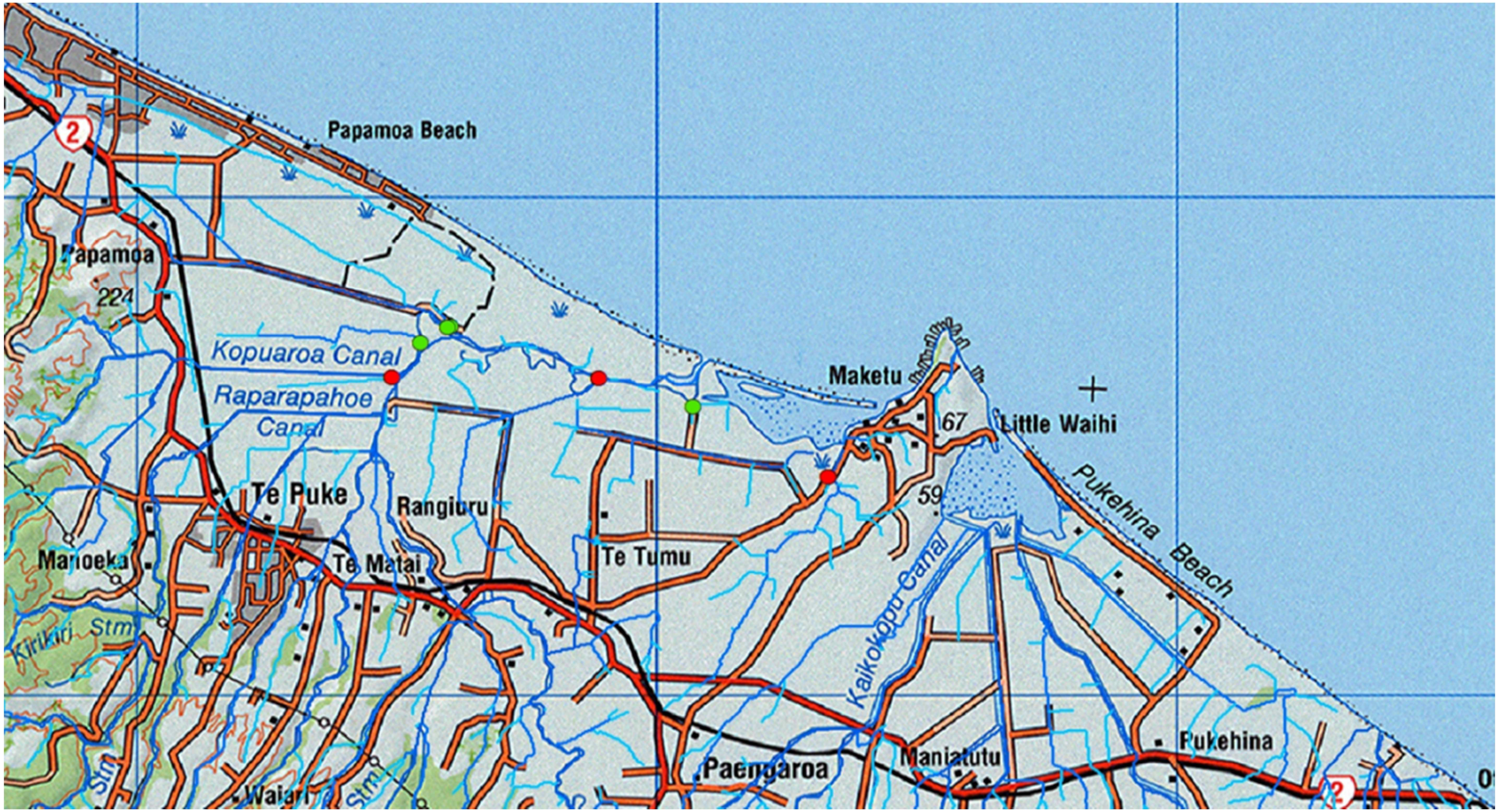

Figure 29 Location of pumps (green circles), and pump stations (red circles) managed by BOPRC in the Kaituna Plains.

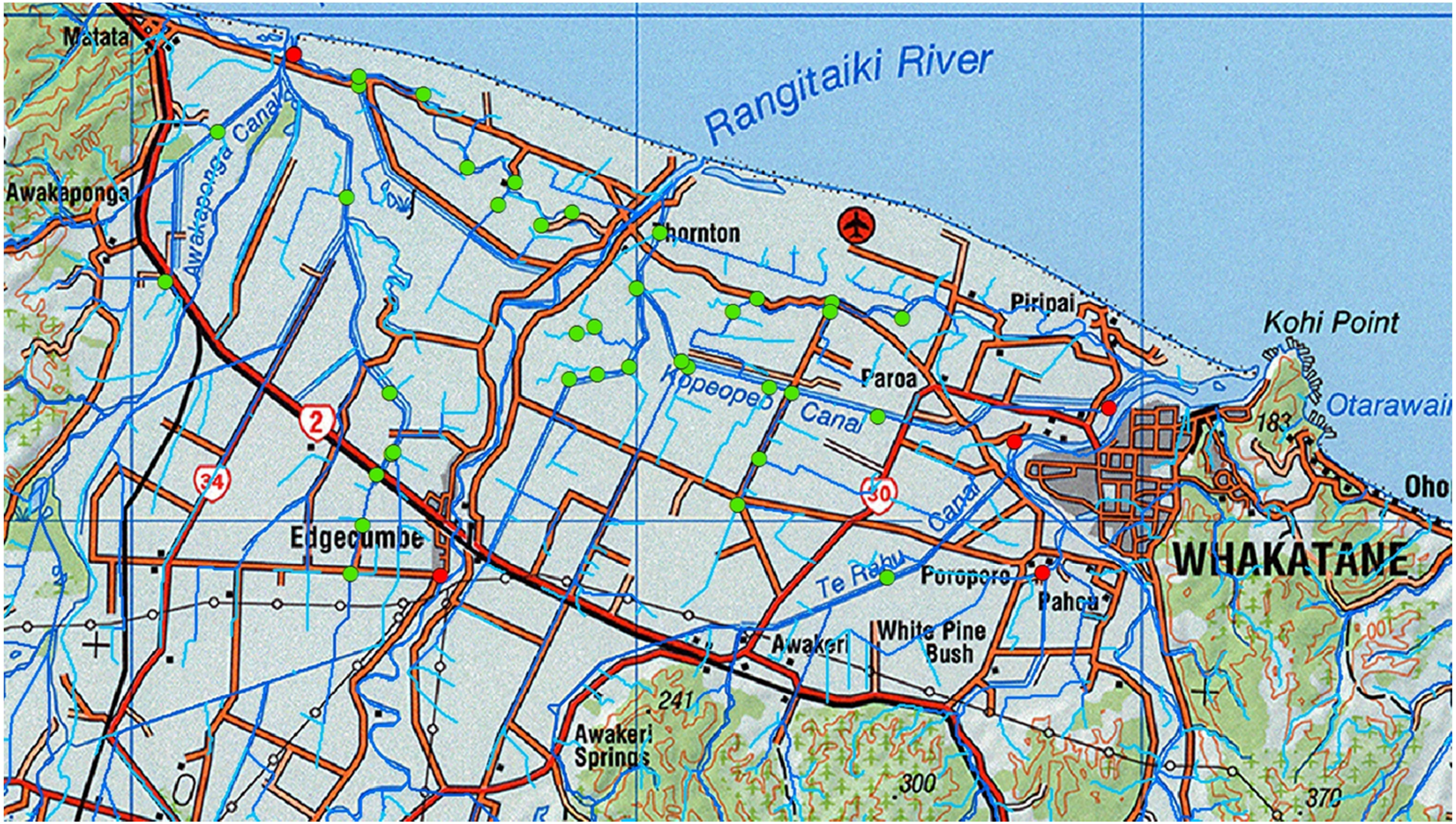

Figure 30 Location of pumps (green circles), and pump stations (red circles) managed by BOPRC in the Rangitaiki Plains.

Other pressures faced by fish in the drainage network include the low habitat quality characteristic of drains, particularly due to the low shade, lack of flow variability and poor riparian vegetation, all of which lead to lack of habitat heterogeneity. Water quality parameters such as low dissolved oxygen and high ammonia concentrations experienced in some drains may also have constrained where some fish species can live.

For example, Dean and Richardson (1999) measured the tolerances of seven New Zealand freshwater fish species to low levels of dissolved oxygen, and found many of the test species were intolerant of oxygen levels less than 1 mg/L. In this case, banded kokopu, torrent fish, common smelt, and juvenile common bullies displayed 100% mortality after a 48-hour exposure. Common smelt were the most sensitive species to low oxygen, dying after only 4 hours, while the other fish were slightly less sensitive, dying between six and 40 hours. The most tolerant fish were shortfin eels, which were not affected even at concentrations less than 1 mg/L. Examination of the water quality data showed that nine of the 20 drains monitored displayed minimum levels below 1 mg/L on at least one occasion. Concentrations fell below one mg/L on six occasions in Sutherland Road drain, four occasions in Secombe's Canal, and three occasions in Landenberger Drain. Given that water quality monitoring was done during the day when plant photosynthesis would be expected to be high, it is likely that these and other drains would display low oxygen levels during the night in the absence of photosynthesis. It is therefore likely that low oxygen levels may be playing an important role in structuring fish communities in many of the drains throughout the Kaituna and Rangitaiki Plains.

The purpose of this section was thus to undertake a fish survey of some drains throughout the Kaituna and Rangitaiki Plains, and to describe their fish communities.

5.2 Methods

Fish communities were assessed by deploying fyke nets in selected drains, which were dominated by fine substrates that made electric fishing impossible. Three unbaited replicate fyke nets (mesh size = 4 mm) were deployed overnight at sites that were unsuited for electric fishing. These nets all had an exclusion panel at the cod end to allow smaller fish to move past, while preventing larger fish from doing so. This minimised any predation of smaller individuals. All nets were retrieved the following morning, usually after a 10-12 hour period, and all fish caught were identified and measured prior to release. All collected fish were kept in buckets and anaesthetised using phenoxy-ethanol (diluted to about 5 ml per 10 l). Fish length was measured to the nearest millimetre, identified, and replaced into a bucket containing natural stream water to recover. All fish were subsequently released back into the stream.

In addition to data collected from the above survey, the New Zealand Freshwater Fish Database (NZFFD) was examined and all data from sites located on the Rangitaiki and Kaituna plains extracted. Data from the main stem of the Kaituna, Tarawera and Rangitaiki rivers were omitted from this analysis, as we were focusing purely on fish communities in smaller waterways draining these areas. This dataset was combined with the data obtained from the surveys, leading to a total of 53 sites from 29 individual waterways. The earliest NZFFD samples were collected in 1974, but most samples (35%) had been collected between 1990-2000 and a further 27% had been collected between 2000-2010. Where multiple surveys had been conducted at a specific site, the average fish community composition at that site was calculated.

Given that we were interested in species presence absence for this analysis, the effect of combining data that had been collected over a relatively long timeframe was thought to be small, especially as fish community composition appears to be stable in streams in the Bay of Plenty (A. Suren, unpublished data). Furthermore, this approach allowed us to maximise the spatial coverage of fish surveys by including all data.

The combined data set was used to characterise the fish community composition of waterways in the Rangitaiki and Kaituna Plains, including the number of different species found at each site and what the common fish species were. The fish index of biotic integrity (Fish_IBI) was calculated to describe the health of fish communities at each site (Suren 2016). This index is based on the number of native species, riffle dwelling species, pelagic pool species, intolerant species and the proportion of native species at each site. It has five narrative classes which are used to describe the integrity of fish communities found at a particular site, which takes into consideration the fact that fish communities respond strongly against a gradient of distance to sea and elevation. These gradients reflect the fact that many of New Zealand's native species need access to and from the sea to complete their life cycle, and that the different species have different abilities to penetrate inland. Scores are calculated based on the number of species expected to be found at a given site when compared to the number of species observed at the site. ANOVA was used to determine whether species richness and Fish_IBI scores differed between streams when allocated to different water quality classes.

Shortfin eels were the most commonly collected fish species in the drains surveyed, and so the length frequency distribution of the species was determined so that comments can be made of the overall population structure found within the waterways sampled.

5.3 Results

A total of 13 waterways were sampled in the Kaituna Plains and 26 in the Rangitaiki Plains. All sites were at low elevation (average = 6.3 m above sea level), and relatively close inland (average = 7.7 km). Of the 53 sites, 24 were from waterways classified as modified watercourses with ecological values, and 19 were from waterways classified as drain water quality.

A further 10 waterways were classified Aquatic ecosystem, Regional baseline, Fish spawning purposes, and Unspecified water bodies. For ease of comparison, these other waterways were grouped to a new class, Other.

A total of 18 fish species were found, the most widespread of which were shortfin eels, inanga, and mosquito fish. These were found at more than 50% of sites sampled. Other widespread species include common bullies, longfin eel and giant bullies (Table 18), which were found in more than 30% of sites sampled. The further eight species were found between two and eight sites, while four species were found only at one site (Table 4).

Table 18 List of the fish species found in the 29 sites sampled in the Kaituna and Rangitaiki Plains showing the number of locations where each species was found.

Common name	Scientific name	No of sites	% of sites
Shortfin eels	*Anguilla australis*	24	82.8
Inanga	*Galaxias maculatus*	18	62.1
Mosquito fish	*Gambusia affinis*	16	55.2
Common bully	*Gobiomorphus cotidianus*	13	44.8

Common name	Scientific name	No of sites	% of sites
Longfin eels	*Anguilla dieffenbachii*	12	41.4
Giant bully	*Gobiomorphus gobioides*	11	37.9
Gold fish	*Carassius auratus*	8	27.6
Redfin bully	*Gobiomorphus huttoni*	7	24.1
Smelt	*Retropinna retropinna*	7	24.1
Torrentfish	*Cheimarrichthys fosteri*	4	13.8
Giant kokopu	*Galaxias argenteus*	4	13.8
Yelloweye mullet	*Aldrichetta forsteri*	2	6.9
Rainbow trout	*Oncorhynchus mykiss*	2	6.9
Grey Mullet	*Mugil cephalus*	2	6.9
Banded kokopu	*Galaxias fasciatus*	1	3.4
Lamprey	*Geotria australis*	1	3.4
Crans bully	*Gobiomorphus basalis*	1	3.4
Cockabully	*Grahamina*	1	3.4

Species richness within the different waterways varied considerably, with the Waiari Stream and the Waikamahi Stream supporting the highest number of species (11 and 10 respectively). Seven sites only supported two species, one of which was shortfin eels.

Other fish commonly found at sites with low species richness were the introduced mosquito fish and goldfish, although common bullies and giant bullies were found only with shortfin eels at two of these sites.

Species richness was significantly different in streams when classified according to their water quality class (Figure 31) and was equally lowest in the DWQ and MEV classes, and significantly higher in the waterways classified as Other.

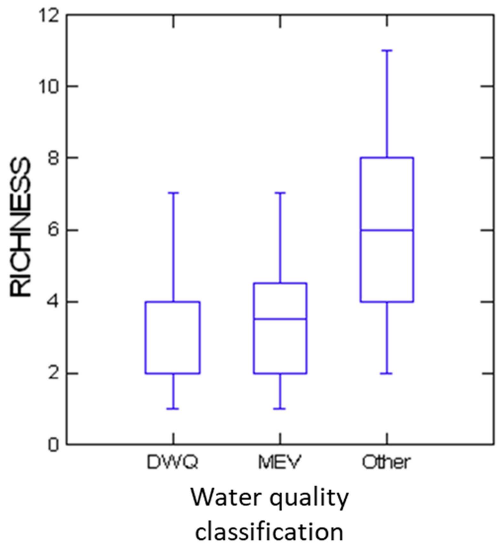

Figure 31 *Box plot showing the different number of fish species found in waterways in the Kaituna and Rangitaiki Plains when classified according to their water quality classification.*

No significant relationships were found between species richness and either distance to sea or elevation, although this result was not surprising considering the low elevation or distance gradients. No significant relationship was also found between species richness and habitat quality of the sites where fyke surveys were conducted.

Calculated Fish_IBI scores differed significantly between the different water quality classes, and were highest in streams classified as Other, and lowest in streams classified as DWQ Figure 32. Samples from MEV had intermediate fish IBI values. All of the waterways sampled had Fish_IBI scores characterised as poor, with the exception of samples collected from the upper site in Te Rahu Canal, which were assessed as being in moderate condition. Another site (Collins Drain) was assessed as being in good condition, with a Fish_IBI score of 43. Data from this site came from visual observations by BOPRC drainage staff as they were cleaning the stream from weeds. Three native fish (giant kokopu, inanga, common bully and giant bully) and goldfish were observed at this site. It is also highly likely that shortfin eels would also be found there. If this species were also included in the calculations of the Fish_IBI, this site would be rated as in excellent condition.

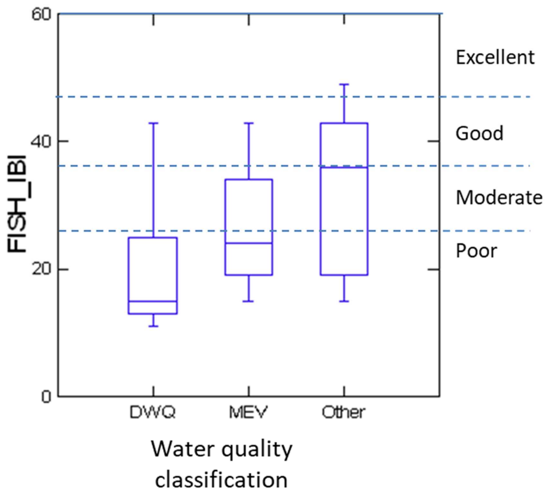

Figure 32 *Box plot showing calculated Fish_IBI scores (and the four narrative bands associated with these scores) that describe ecological integrity of fish communities found in waterways in the Kaituna and Rangitaiki Plains when classified according to their water quality classification.*

Examination of the distribution of the most common fish showed that both shortfin eels and inanga were found at all sites throughout both the Kaituna and Rangitaiki Plains (Figure 33, Figure 34). Gambusia was also widespread in both Plains, although it was absent from waterways draining Te Puke such as the Kopuaroa Canal, Raparapahoe Canal and the Ohineangaanga Stream (Figure 35). Unlike common bully, which appeared at sites throughout both the Rangitaiki and Kaituna Plains (Figure 36), giant bullies were found at only two sites in the Kaituna Plains but were widespread throughout the Rangitaiki Plains (Figure 37).

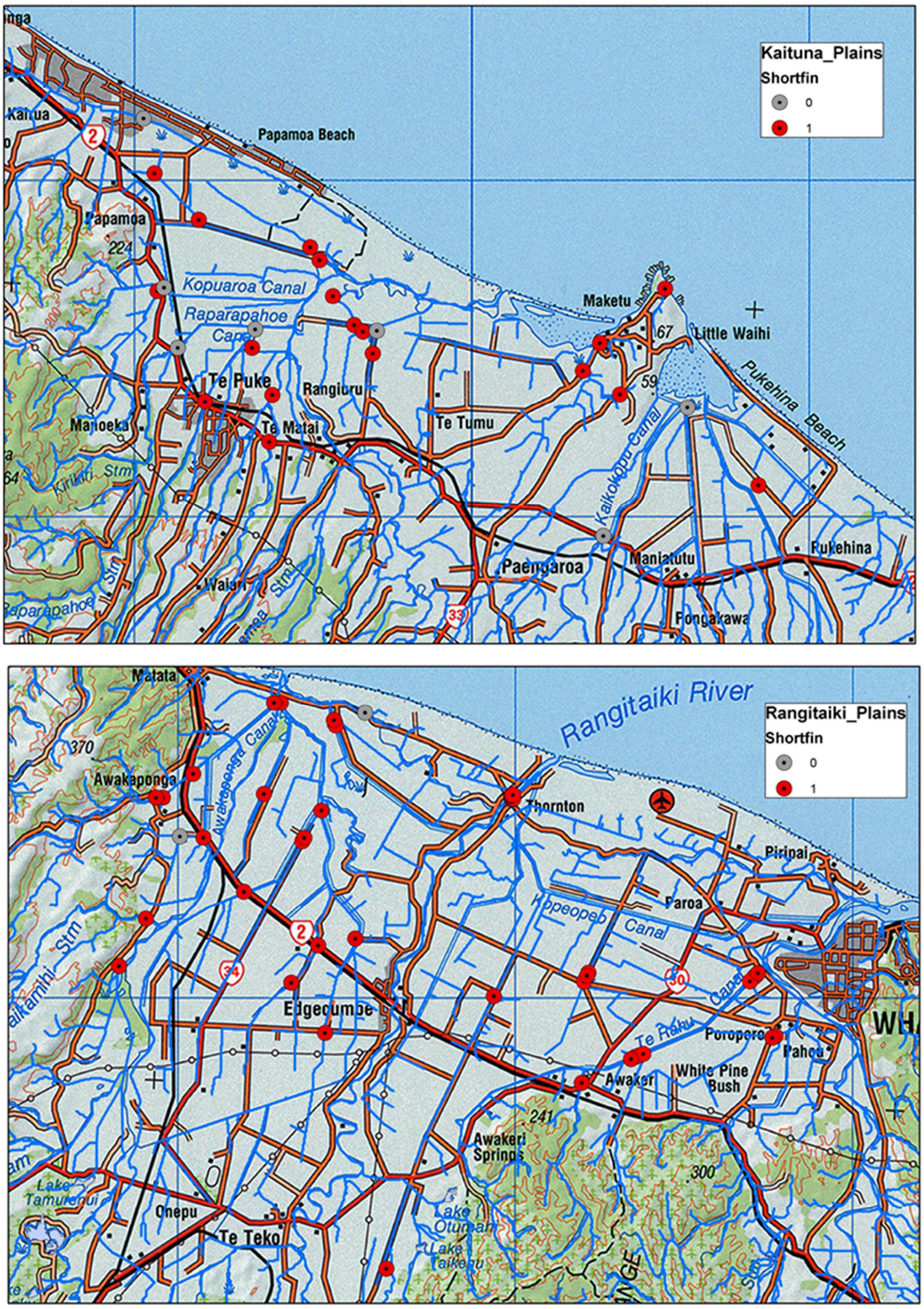

Figure 33 *Distribution of shortfin eels in waterways throughout the Kaituna and Rangitāiki plains showing sites where these fish were absent (grey circles) and present (red circles).*

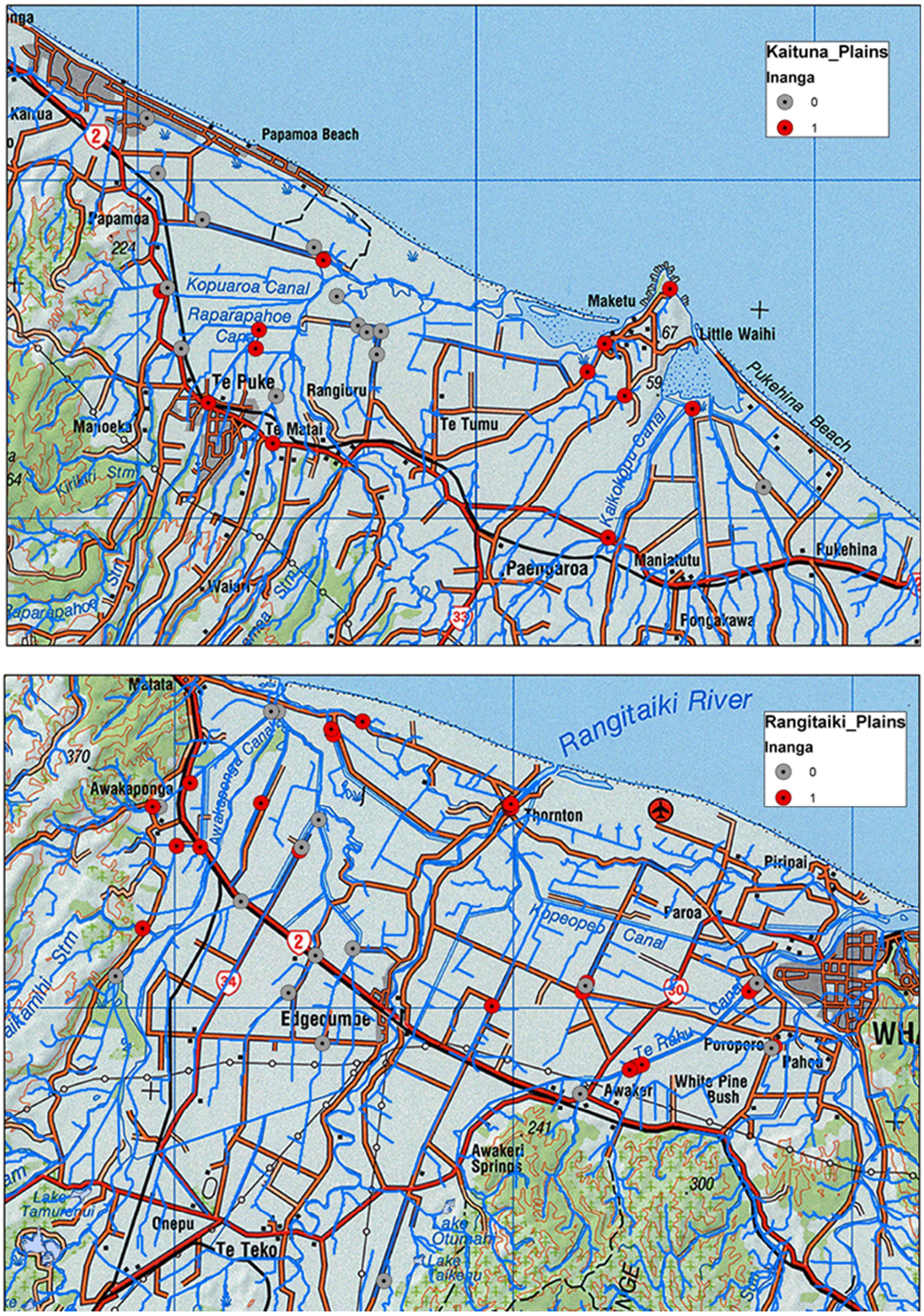

Figure 34 Distribution of inanga in waterways throughout the Kaituna and Rangitaiki Plains showing sites where these fish were absent (grey circles) and present (red circles).

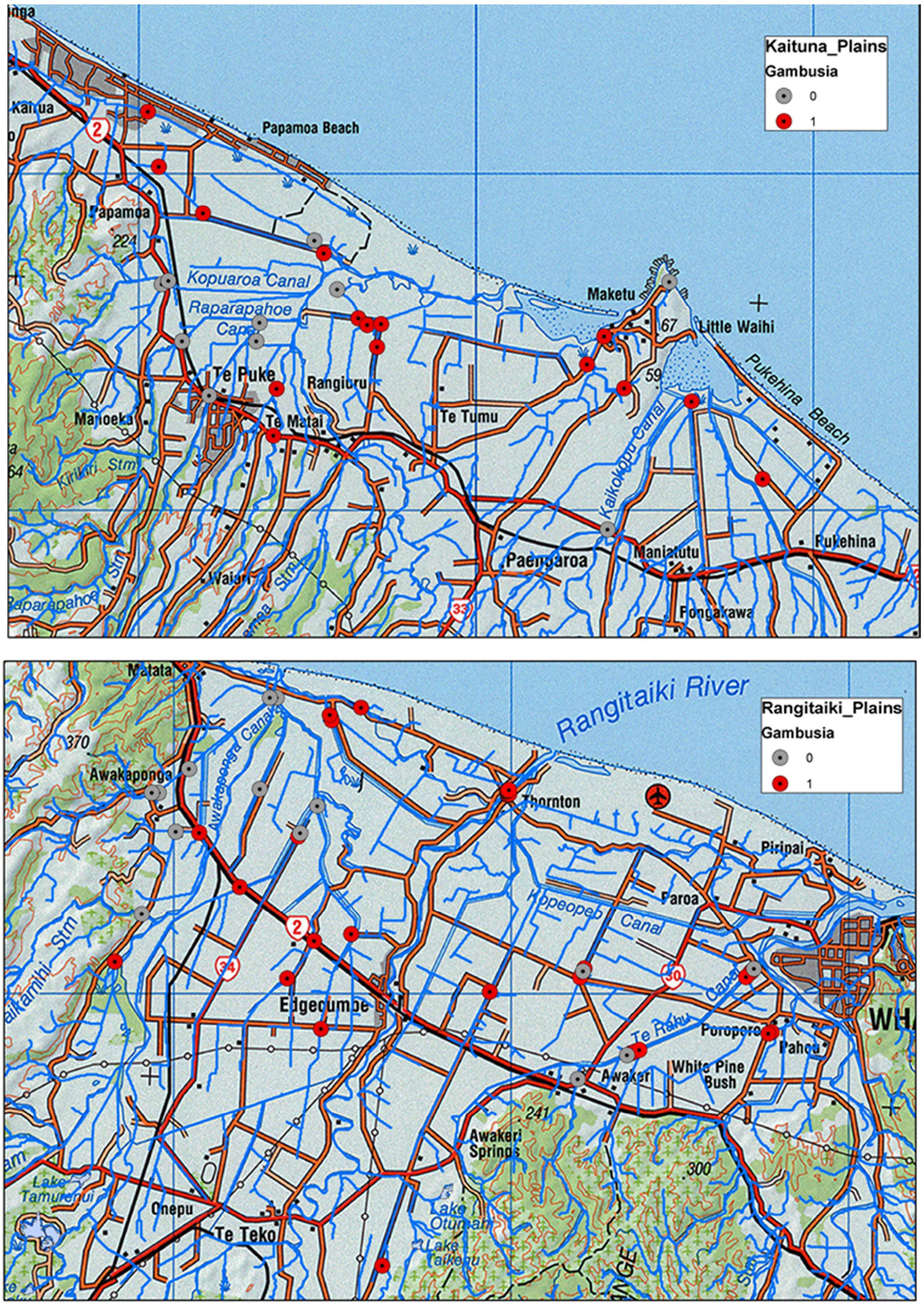

Figure 35 *Distribution of gambusia in waterways throughout the Kaituna and Rangitāiki plains showing sites where these fish were absent (grey circles) and present (red circles).*

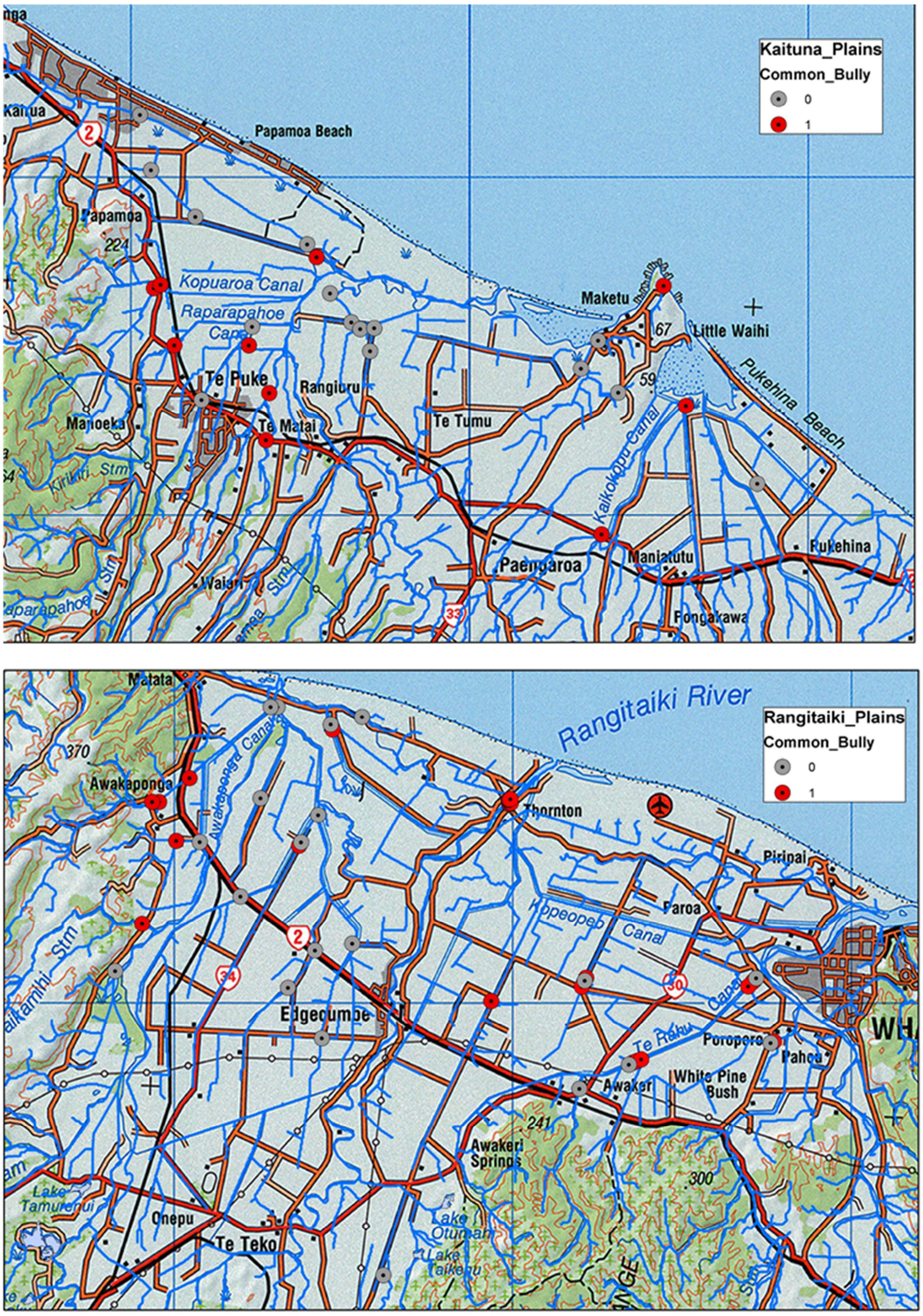

Figure 36 *Distribution of common bully in waterways throughout the Kaituna and Rangitaiki Plains showing sites where these fish were absent (grey circles) and present (red circles).*

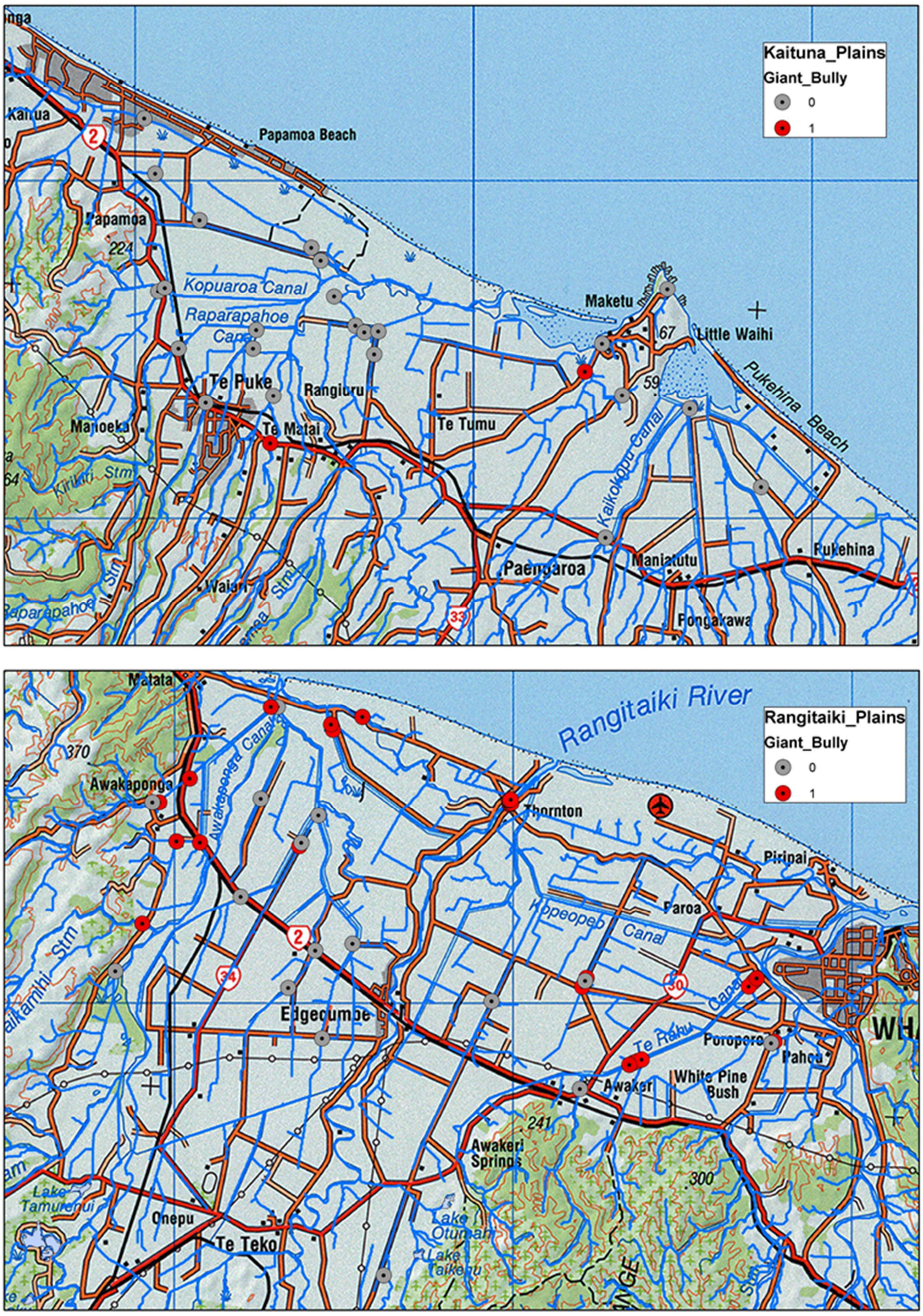

Figure 37 *Distribution of giant bully in waterways throughout the Kaituna and Rangitāiki plains showing sites where these fish were absent (grey circles) and present (red circles).*

The size frequency distribution of shortfin eels, the most commonly encountered fish, was examined. A total of 868 shortfin eels were collected from the six drains. The numbers of shortfin eels caught in the fyke nets differed significantly between sites and were highest in Landenberger Road Drain (296 eels collected), Awaiti Canal (257 eels collected), Seacombes Canal (220 eels collected), and lowest in Te Rahu Canal where only 10 eels were found in the four fyke nets. The average number of shortfin eels collected in each fyke net followed a similar pattern and was highest in Awaiti Canal (average of 86 eels per fyke net), and lowest in Te Rahu Canal (average of only three eels per fyke net) (Figure 38).

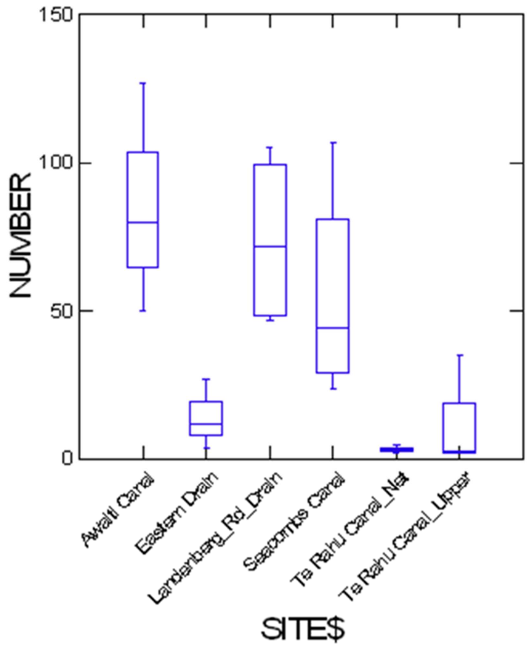

Figure 38 *Average number of shortfin eels collected in fyke nets deployed in each of the six drainage canals in the Rangitaiki Plains.*

Shortfin eel size ranged from a minimum of 85 mm to 850 mm, with an average of 383 mm. Size frequency distribution showed most eels were of intermediate range of between 250 and 450 mm, with much fewer smaller and larger eels (Figure 39). A normal healthy population of shortfin eels would be characterised by large numbers of smaller individuals, fewer large individuals. Such a healthy population distribution reflects an ongoing immigration of young elvers into sites, and a gradual decline in population density due to mortality. Lack of small elvers in the samples may suggest a barrier to the ongoing successful recruitment of Shortfin eels to many of these sites, or a bias against the collection of small individuals in the small meshed fyke nets.

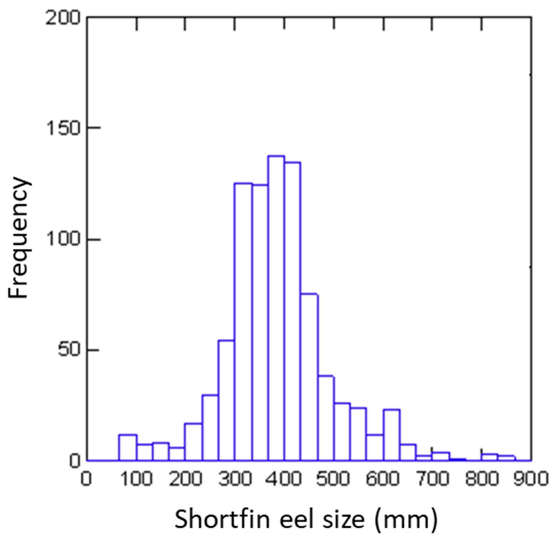

Figure 39 *Size frequency distribution of shortfin eels collected from six drainage canals in the Rangitaiki Plains. Note the very low numbers of both small and large individuals.*

5.4 Discussion

Some of the drains sampled supported large quantities of shortfin eels, despite the low habitat and water quality conditions, and despite the presence of pump stations below some of the drain sites. For example, two of the sites where the most shortfin eels were caught (Seacombe's Canal and Landenberger Drain) were above pump stations, while the Awaiti Canal was above a culvert with an aluminium floodgate. The presence of eels above these structures therefore implies that at least some individuals have been able to pass through or around these structures, possibly during periods when young elvers were migrating back upstream and were able to negotiate small gaps or leaks in these structures. Another potential pathway mechanism could have been movement of eels into these habitats during large floods if water was spilling over or around these structures. Indeed, large floods occurred across the Rangitaiki Plains in 2004 and 2017, and this may have allowed migrating eels to access many of the drainage canals above flood gates. A third explanation could of course be manual liberation of individuals at sites throughout the plains by commercial or recreational eel fishermen.

Irrespective of how eels had entered the drainage channels above pump stations or weirs, it is clear that the population structure found in the six sites surveyed was dominated by a cohort of medium size eels, with very few small elvers or large individuals. The fyke nets used in this study had a small mesh size (4 mm) and a series of baffles along their length that prevented large individuals from preying upon smaller individuals. The lack of small elvers in the study therefore is unlikely to represent unequal sampling efficiencies based on size. Instead, lack of small elvers implies that the mechanisms of colonisation to these upstream areas may be sporadic at best, as more young elvers would be expected in these areas if the colonisation pathways were always open.

That so many eels were found in drains such as Landenberger Road and Secombe's Canal where dissolved oxygen concentrations were extremely low was surprising. Eels are well known to survive in environments with low oxygen as they can take up oxygen through either their gills or their skin.

In addition to low oxygen levels, high ammonia concentrations in the drains may be affecting fish communities. Although native fish such as shortfin eels appear to be a very tolerant of high ammonia concentrations, other species such as inanga, smelt, and banded kokopu are less tolerant (Richardson 1997). As discussed above (Section 3), median ammonia concentrations in seven drains exceeded the lower toxic threshold for these species, all of which are likely to be common in lowland, coastal waterways. However, Richardson only determined the lethal concentrations of ammonia, yet a chronic exposure to sub-lethal concentrations can also affect fish, and alter their distributions through behavioural changes. For example, Richardson et al. (2001) assessed behavioural responses to combined stressors of both DO and ammonia on three fish species characteristic of lowland waterways (inanga, smelt, common bully) and the shrimp, (*Paratya)*. They found that smelt showed very strong avoidance behaviour to both low DO and high ammonia concentrations, both alone and in combination. This highlights a fundamental issue with assessing only acute effects of stressors such as DO and ammonia, as Hickey et al (1993) showed that *Paratya* was one of the least sensitive species to acute toxicity tests for ammonia.

The finding that smelt display such a strong avoidance behaviour to both low DO and elevated ammonia concentrations lead Richardson et al (2001) to suggest that smelt could be a good indicator organism to assess ecological health of lowland waterways. The fact that relatively few drain sites in the Kaituna and Rangitāiki Plains support smelt may indeed suggest that the ecological health of these waterways is relatively poor.

A total of 18 fish species were recorded from sites throughout the Kaituna and Rangitaiki Plains. Of these 18, three species (yellow eyed mullet, grey mullet, and cockabully) could be regarded as marine wanderers that regularly come into the lower estuarine part of rivers for feeding.

Twelve other native species were found, all of which require access to and from the marine environment as part of their life cycle. The relatively high number of native species found in his low elevation sites emphasises the fact that such environments are important habitats for many native fish, or act as migratory corridors for other species to gain access to upper reaches of waterways. Five of the fish species recorded in waterways in the plains (inanga, longfin eel, redfin bully, torrent fish and giant kokopu) are regarded as at risk, declining, and lamprey are regarded as threatened, nationally vulnerable (Goodman et al 2014). All of these species require access to and from the sea to complete their life cycles, so the importance of maintaining unrestricted access cannot be over emphasised.

That water quality and habitat conditions in some drains are likely to be detrimental to the long-term survival of larval fish that enter these systems means that they may in fact be acting as population sinks, as any juvenile fish entering them may not survive to breed. This finding adds a new dimension to work by Hickford and Schiel (2010), who found that large rivers had little or no spawning habitat and very little egg production and effectively became sink populations despite supporting large adult populations of inanga. In contrast, some of the smallest pristine streams they surveyed produced millions of eggs. However, many of the small waterways in the Kaituna and Rangitāiki plains currently have usually very poor riparian conditions for spawning, even if they had no flood gates on them. Prior to agricultural development and the associated drainage activities in the area, these areas would have represented important breeding areas for many of these species.

Of interest was the finding of many fish in drains above pump stations or floodgates. This shows that these structures are not complete barriers to upstream movement. What proportion of the total migrating population can pass these structures is, however, unknown. A similar finding was reported by Franklin and Hodges (2015), who found migratory fish including eels, banded kokopu, common bullies and inanga above a large floodgate in the Kurere Stream, in the Coromandel. They attributed the presence of these migratory fish above the floodgate due to the ability of fish to pass through the gates drain at low tide when the gates were opened by catchment flow. Furthermore, a small (3 cm) gap was identified between each gate and the concrete headwater structure which may also have facilitated fish passage. Despite this, Frankston and Hodges (2015) emphasise that for much of their life cycle, fish passage would have been restricted due to gate closure, and they concluded that the abundance of native migratory fish in the catchment above the gates would have been higher if the gates were not there.

Of greater concern, is the effect of these structures on the downstream migrating behaviour of fish, and in particular the effect of pump stations on large migrating fish such as eels. Both longfin and shortfin eel can reach large sizes (over 700 mm in length) before migrating and these large migrant individuals may be particularly susceptible to mortality or pump stations. Such mortality has recently been assessed at the Orchard Road pumping scheme in Te Kauwhata, Waikato (Vaipuhi Consulting 2017), and found to be very high. Conditions in the Bay of Plenty drainage system differ to Waikato, for example many pump stations have fish screens installed. However further work on the risk of eel mortality due to pump stations is warranted, or to investigate what practical steps can be taken to allow migrating eels to bypass pump stations that are fitted with screens.

Duirs (2017) reviewed the effects of land drainage and flood control schemes on native fish passage and highlighted the adverse effects of these schemes on fish migration pathways. Although they acknowledged that there is limited information to quantify these adverse effects, there appears to be enough anecdotal evidence to suggest that such adverse effects are relatively widespread throughout land drainage schemes throughout the country. They note that, to date, there appears to have been only limited efforts by land drainage and flood scheme managers to address such adverse effects, although they acknowledged that most councils managing and maintaining such schemes now acknowledged the issue, and are considering implementing remedial measures.

Of relevance is the fact that many overseas countries have developed stringent policies to address adverse impacts of land drainage activities on eel populations, and Duirs highlights that the sustainable management provisions of both the RMA and various regional policy statements indicate a requirement for fish passage exists.

The Duirs review also highlighted a range of potential remedial options to address fish passage that councils, including BOPRC, have used. Their report showed that BOPRC has fitted small transformers to impart electrical charges through debris screens to deter fish from entering pump stations although the efficacy of this is unknown. Although these measures may prevent eels from entering pumps with lethal effects, they still provide no easy way for migrating eels to pass below these pumps. Other options would include fish friendly pumps such as Archimedes or hidrostil pumps, as well as fish passes and trap and transfer operations. Although we know that eels migrate at the first significant rainfall in March and April (Boubee, Mitchell et al. 2001), the timing of such rainfall events is highly unpredictable and therefore a trap and transfer operation will, by necessity, need to be implemented at short notice. Furthermore, most eel migration occurs at night, and this is likely to place further logistical constraints on the success of such operations. More robust and longterm solutions are thus needed to solve the somewhat vexed issue of the need to maintain the ability to pump excess water from the drainage network into downstream rivers whilst minimising eel mortality.

Another potential stress on fish communities in the Kaituna and Rangitiaki plains concerns the process of macrophyte removal. This activity can have large effects on water quality (Ballantine and Hughes 2012), resulting in increases in total suspended solids, and total phosphorus loads in drain water following drain clearing. Drain clearing can also significantly affect channel morphology, bank vegetation and instream physical conditions such as temperature (Ballantyne and Hughes 2012). Drain clearing may also adversely affect fish communities, although these effects are not clear cut. For example, Ward-Campbell et al. (2017) examined the effects of drain maintenance on fish assemblages in eight drains in Ontario, Canada, and found no evidence for short or long-term adverse effects on fish communities, despite clear changes in physical habitat. They thus suggested that the fish assemblages in the drains were resilient to drain maintenance.

Such resilience could be explained by the fact that the fish communities in these drains were dominated by cyprinids, which have behavioural and reproductive strategies that can facilitate rapid recolonization following drain maintenance, and therefore display a high degree of resilience to disturbance. These results are very different to work by Greer et al. (2012), who showed that macrophyte removal to maintain drainage performance had negative effects on resident fish communities in a small Southland stream. They found that that complete macrophyte removal reduced the abundance of giant kokopu (*Galaxias argenteus*), presumably due to loss of habitat for this species, as well as removal of invertebrate prey from the water column. Giant kokopu are commonly found in lowland waterways amongst dense macrophyte vegetation, and are regarded as at risk, declining (Allibone, David et al. 2010). Drain management practices associated with macrophyte removal may thus be having adverse effects on this species.

However, the study by Greer et al (2012) involved mechanical drain clearing using an excavator, a practice which is not extensively used throughout the Bay of Plenty. Instead, BOPRC manage weeds using a combination of methods, including mechanical drain clearing, spraying with herbicides, and use of a specially built plant cutter boat to remove excessive macrophyte growth from drains to a specified depth that is controlled by the boat operator. Such activities are also controlled by clear guidelines for river drainage maintenance activities (Crabbe and Ngapo 2001), which among other things, specify the timing of weed cutting as well as what to do with the quantities of cut weed produced. The use of the plant cutter boat on fish communities in particular is likely to result in having less adverse impacts on fish than other methods, as they presumably may be able to swim away from the boat and cutting blades. More importantly, some macrophytes can be retained along the bottom and riparian margin of the drains, and this remnant plant material would represent valuable habitat for species such as giant and banded kokopu that require fish cover. Giant bully, giant kokopu, and koura have been observed in Collins Drain during past weed cutting operations, highlighting that even apparently small drains that are covered with weeds can support important fish species. Targeted fish monitoring could be undertaken to ascertain the relative impacts of different macrophytes removal methods on native fish species, to help select a suitable method which efficiently removes macrophytes with minimal impact on resident fish populations.

Another control method for managing excess macrophyte growth in some drains in the Rangitāiki Plains is based on using grass carp (*Ctenopharyngodon idella*) in drains. Grass carp are native to parts of central Asia and can reach sizes in excess of 30 kg. These herbivorous fish are highly unlikely to successfully breed in New Zealand waters (Rowe and Shipper 1985), as their fertilised eggs need to be continually entrained in large, fast flowing rivers, conditions highly unlikely to be encountered in the drainage network.

Furthermore, the grass carp that are used to control macrophytes are triploid animals and are consequently sterile. These fish have been shown to successfully reduce macrophyte cover in a number of drains throughout the country (Rowe and Schipper 1985), while minimising adverse effects associated with mechanical or chemical control for macrophytes. Despite these advantages, grass carp appear to be used in only a few locations throughout the country, presumably reflecting a high degree of public concern at their introduction. McDowall argues that this concern is simply not warranted, given the overall lack of adverse effects of grass carp on other native New Zealand species. He also highlights the irony that relatively benign fish such as grass carp have been somewhat maligned whereas fish such as trout, which have documented large adverse effects on native biodiversity, are actively managed and liberated throughout the country.

Within the Rangitaiki Plains, grass carp are used in a number of drains including for drains to the West of the Rangitaiki River (Secombe's Canal, Crystals drain, Awaiti East Scheme Drain, Mahys Outlet and Pearces Outlet), and drains to the East (Law's Scheme Drain and Section 72 Outlet; Figure 40). Given the comments of McDowall (1990) and the fact that the positive outcomes of the biological removal of excess macrophyte growth using grass carp may well outweigh their perceived risk, there could be a strong case to introduce these fish into more drains in the Kaituna and Rangitaiki Plains. Any introductions of grass carp into more drains would need to consider the suitability of these drains to support introduced fish, especially given the potentially low dissolved oxygen levels characteristic of many of the drains. However, grass carp have been successfully used in Secombe's Drain for a number of years, and this drain was noted for its very low oxygen levels during the monitoring period. On one occasion (13 March 2017), grass carp were seen congregating just above the pump station in large masses where they were gulping for air to help oxygenate their gills. Just prior to this time (7 March 2017), monthly monitoring recorded an oxygen concentration of only 0.52 g per cubic metre, or 5.7% saturated, one of the lowest values records during the sampling period. This sample was collected just prior to a relatively heavy rainfall event, when a total of 165 mm was recorded at the Edgecumbe metrological station over this six-day interval.

It is highly likely that this rainfall resulted in a further reduction to the oxygen concentration in this already stressed system as nutrient rich material was brought into the drain and potentially mobilised the anoxic bed organic matter. Notwithstanding this event, the fact that grass carp have survived in both Secombe's Canal and the neighbouring Crystals Drain, where oxygen concentrations can often reach very low levels suggests that grass carp may be relatively tolerant to periods of low oxygen, and could therefore survive in other drains throughout the plains. There is consequently an option of using these fish to control excessive macrophyte growth in other waterways, pending approval from organisations such as the Department of Conservation and Fish and Game.

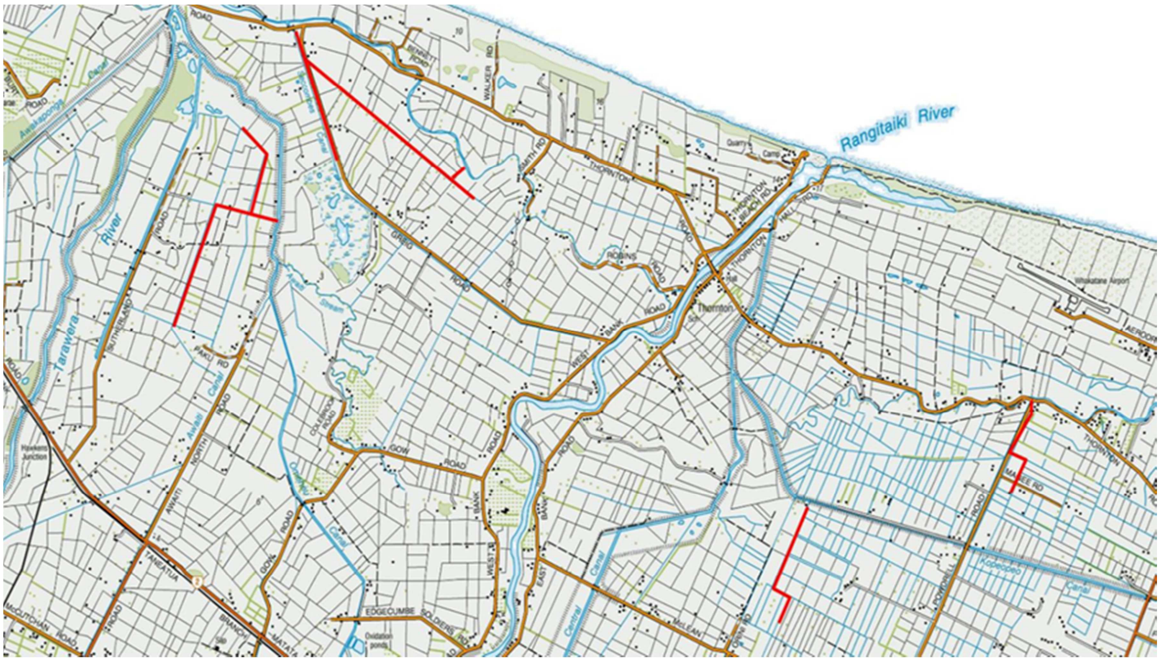

Figure 40 *Location of drains where grass carp have been liberated to control macrophytes (red lines). (Source, Andrew Pawson, Works Co-ordinator, Rivers and Drainage).*

Part 6:

General discussion

6.1 The current status

The drainage network in the Kaituna and Rangitaiki Plains is characterised by poor instream physical habitat conditions, with high amounts of fine sediment, a large degree of channel alteration, a lack of bank vegetation and no riparian shade.

Water quality typically has high to very high levels of nutrients, turbidity, and extreme levels of dissolved oxygen - both high and low. Such conditions are typical of other lowland waterways flowing through productive agricultural landscapes throughout the country (Wilcock et al. 1999, Larned et al. 2004), although the low DO levels and high ammonia levels characteristic of the drains we sampled were generally much greater than observed in other catchments. Catchment loads of nutrients are also very high, especially for ammonia, despite the relatively small size of these drains.

These high catchment loads reflect the intimate contact of the drainage network to the surrounding agricultural land. The lack of riparian vegetation has major effects on both physical habitat and water quality of the drainage network, and results in excessive primary production (usually in the form of macrophytes), warm temperatures, high bacterial respiration and low DO levels. Other characteristic features of many of the drains are the presence of either floodgates or pump stations in their lowermost reaches.

These structures are likely to affect both water quality conditions above them (Franklin and Hodges 2015), and the ability of fish to successfully migrate up and down these waterways (Vaipuhi Consulting 2017). The inability of fish to freely migrate into many of the drains, combined with the often-poor habitat and water quality conditions will affect the fish communities in many of these lowland drains.

Many of these physical and water quality features represent either direct or indirect stressors that can affect fish and invertebrate communities (Figure 41). For example, the small riparian zones around drains are likely to result in large inputs of sediment, organic matter and nutrients. These inputs, combined with a lack of shade that results in high temperatures and high photosynthetic rates can often lead to excessive plant growth (Wilcock and Nagels 2001). Although these plants can be removed by chemical (use of Glyphosate), physical (drag lining or use of the weed cutter boat) or biological means (grass carp), excess organic matter in the drains will often accumulate within the stream bed. Microbial decomposition of this organic matter under high temperatures can lead to very low dissolved oxygen concentrations in some drains, which can directly affect both fish and invertebrates.

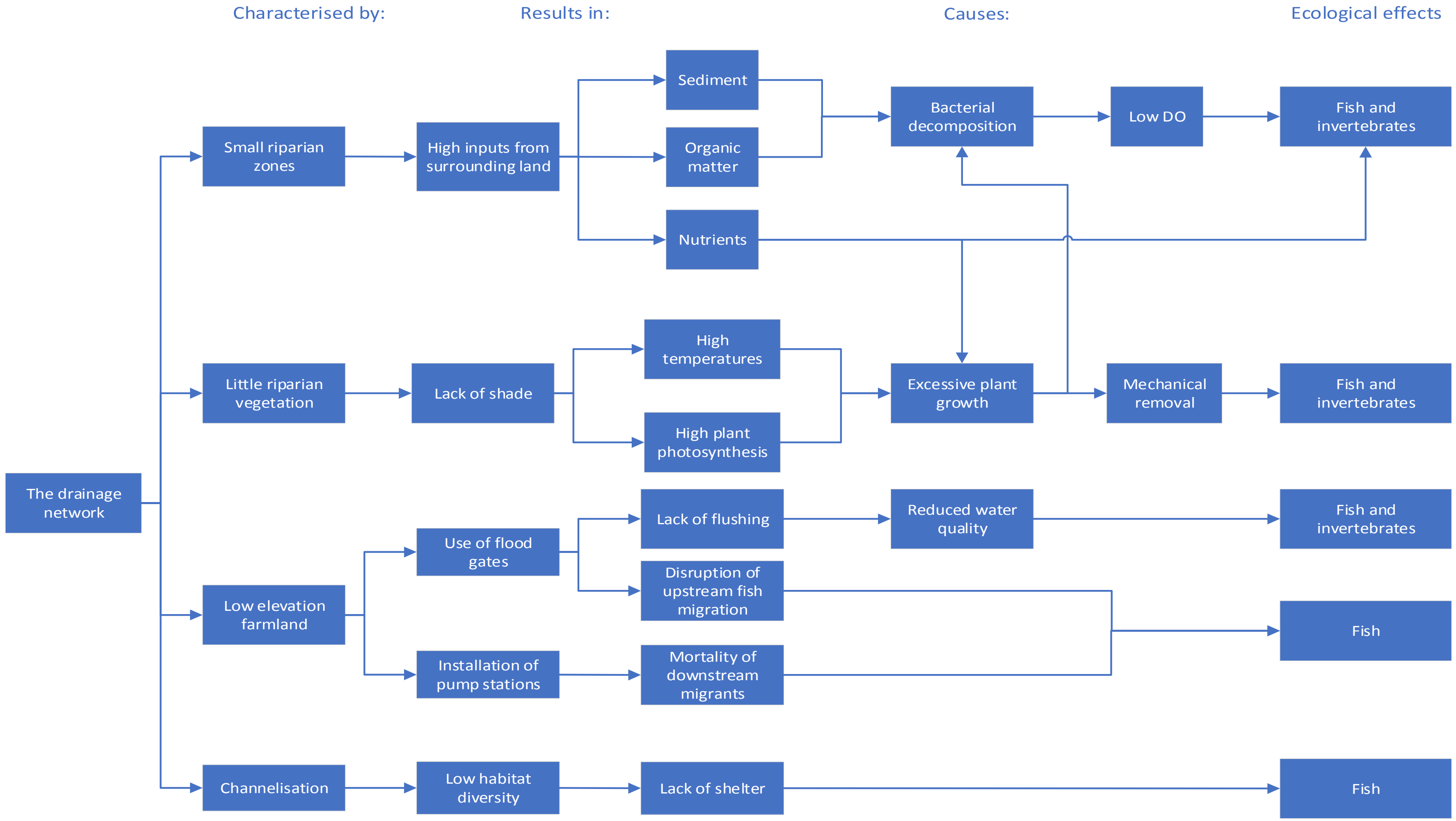

Figure 41 Conceptual diagram showing the dominant physical features of drains, the net result of these features and their ecological implications. Also shown are the dominant ecological groups that are affected by changes in the physical, water quality and hydrological features.

Most of the drains surveyed are found in low gradient areas of the Kaituna and Rangitaiki Plains and are often characterised by presence of floodgates or pump stations. Presence of floodgates can have large effects on water quality as the normal bidirectional movement of water during a tidal cycle has been disrupted. This can often result in stagnation of freshwater above the tide gates, resulting in sedimentation increased water temperatures, and reduced oxygen concentrations (Franklin and Hodges 2015). Furthermore, presence of floodgates in the lower reaches of the drainage network may prevent the normal gradual transition in salinity, and it is this salt wedge where species such as inanga spawn (Hickford and Schiel, 2011). Consequently, adult whitebait living in rivers below floodgates would be unable to access potential spawning locations in the small side tributaries/drains, if conditions above these floodgates were conducive to spawning. Such populations would thus represent a population “sink”, as these individuals would be unable to successfully breed.

6.2 Mitigation measures

This report would not be complete without some discussion on potential ways to mitigate the adverse effects of the current conditions in the drainage network. This discussion is especially pertinent given the need under the NPS-FM for regional councils to improve WQ and ecological conditions to achieve desired outcomes. Although community expectations for conditions in the drainage network would not be expected to be as high as expectations for natural waterways, the fact that MCI scores in many MEV waterways are lower that the threshold for action in the NPS-FM suggests that some form of mitigation may be required to improve ecological condition in the drains. This is also relevant given that nutrient, sediment and bacterial loading of the drainage network is likely to have detrimental impacts on sensitive receiving environments, particularly estuaries.

Many larger drains are maintained by BoPRC, and many are characterised by steep edges and lack riparian vegetation. Spraying is also commonly done along the margins of these drains, presumably in an effort to maximise hydraulic efficiency (Figure 42). There may be an opportunity to investigate alternative management practices in these larger drains, which receive water from the smaller drains, and which then discharge into sensitive environments such as estuaries.

A) Lateral drain

B) Kaituna Rd drain

C) Bell Rd drain

Figure 42 *Examples of current drainage management of both a small lateral drain (A) that feeds into the larger Kaituna Road Drain (B). Also shown is the Bell Road Drain (C), where all riparian vegetation has been sprayed. Note the total lack of shade along these drains, plus the recently sprayed grass. Note also the line of small shrubs along the left (north) side of the Kaituna Road Drain (B) as part of a trial of to increase riparian shade by BOPRC.*

Implementation of a targeted programme of improved drain maintenance by BOPRC in the Kaituna Catchment could thus be a first step in dealing with issues identified by Park (2016) in reducing the input of nutrients, sediment and *E. coli* into both the Kaituna-Maketu and Waihī estuaries. They could also serve as impetus for more work to be done on the many smaller lateral tributaries that flow into the council-maintained drains as well. More importantly, there could also be demonstrable ecological benefits in the larger drains through increased shading and reduced macrophyte growth, as well as potential reductions in stream temperature. These two factors may lead to increases in dissolved oxygen levels.

The following discussion is an initial exploration of potential options that could be considered to mitigate the effects of intensive farming practices on water quality and ecological conditions in DWQ drains and MEV land drainage canals. Some are more tried and tested than others, and it is these that require more extensive feasibility assessment to assess their value in improving the ecological condition of these waterways. Finally, reducing land use intensification is also an obvious way to potentially mitigate adverse effects on waterways, but this is outside the scope of this discussion.

Overall drain management

It may be possible to implement specific interventions such as those outlined in the Dairy New Zealand (2016) report to minimise some of the dominant stressors found within the drainage network (Figure 43). McDowall et al. (2013) also reviewed current mitigation strategies designed to reduce contaminant losses of nutrients, sediments and *E. coli* from productive agricultural land to freshwater. In their report, they highlighted the disproportionate effect of dairy farming on total nitrogen loads into streams. For example, dairy farming contributes approximately 38% of total nitrogen load into streams despite covering only 6.8% of land area. Although dairy farming is not the only activity undertaken in the Kaituna and Rangitaiki Plains, it nevertheless makes up a significant proportion of this area. A number of potential mitigation measures identified in the McDowall et al Report could thus be used among the drainage network in the Kaituna and Rangitaiki Plains, including:

- Constructed wetlands.
- Natural seepage wetlands.
- Stream fencing.
- Vegetated buffer strips.
- De-nitrification beds.
- Dams and water recycling.
- Weed harvesting.
- Floating wetlands.

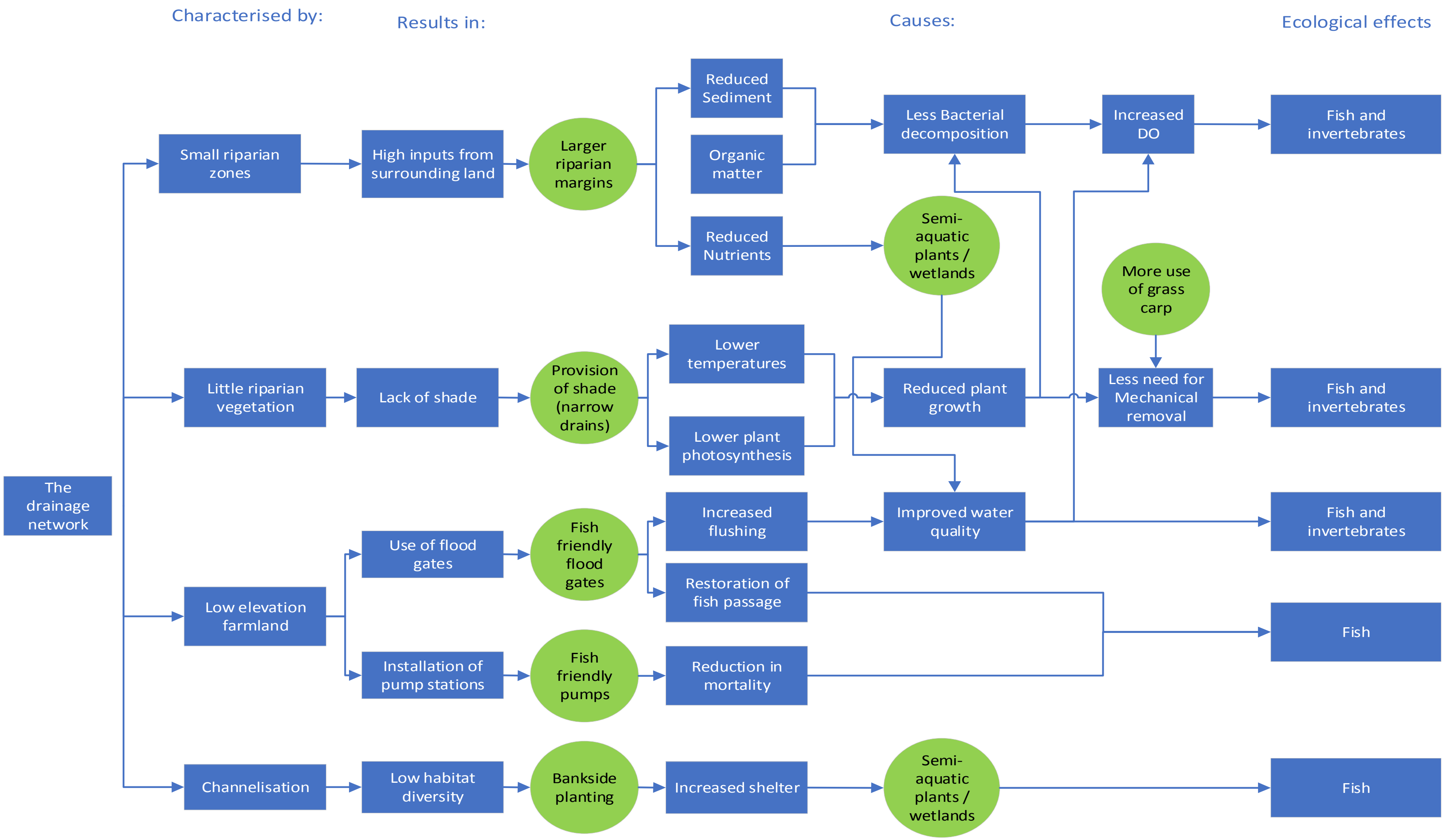

Figure 43 *Conceptual diagram showing potential mitigation measures (green circles) that could be employed to mitigate the adverse effects of the current management practices of the drainage network.*

The McDowall et al (2013) report clearly outlines the different effectiveness, costs, and co-benefits of each of these potential mitigation measures. Measures such as using natural seepage wetlands and denitrification beds have high efficiencies at removing TN, but also have very high relative costs. Constructed wetlands have slightly lower relative costs for the same efficiency, but often require large areas to be effective, which may not be practical around many of the drains, unless suitable land can be bought at the base of selected catchments. They are also mainly effective only during base median flows, when load contribution is typically low. Constructed wetlands also decrease flow rates and increase the contact time of water with vegetation, allowing for nutrient uptake by plants and biofilms. This, however, is counter to one of the main purposes of drains, which is to quickly and efficiently remove excess water from the catchment. However, wetlands can also serve as useful flood mitigation devices by holding back excess water, so they may have benefits in some areas.

In contrast, other mitigation methods such as stream fencing or irrigation of the land with the drain water (i.e., water recycling), can have a large effect at mitigating phosphorus losses to water at the farm scale for a relatively small cost (McDowall et al 2013). It may be thus possible to encourage more farmers to irrigate with drain water during summer, as this may allow more uptake of nutrients by plants and require less fertiliser to be applied to the land.

Riparian planting

Stream fencing and the provision of larger riparian areas would also have benefits in terms of providing shade, thus reducing in stream plant growth (e.g., Dawson 1986, Johnstone 1986), and potentially reducing microbial decomposition rates, leading to increased oxygen levels (Figure 43). It may even be possible to provide incentives to farmers to increase the size of riparian margins around drains if they can plant some form of cash crop that can both help protect water quality and provide some form of financial return. Potential plants to be used for this could include flax, watercress or mānuka. Plants such as watercress in particular, would be expected to grow well under the high nutrient and high light conditions characteristic of the drains, and these could be used to generate some income from the relatively small area lost due to increased riparian margins, although care would be needed to minimise any bacterial contamination of any harvested material. These plants could also be utilised as part of constructing floating wetlands, which are also known to have beneficial effects on improving water quality in some systems. Note, any riparian planting around drains need not necessarily be done along their entire length; instead planting could be done in discrete patches associated with critical source areas, or along a single (north facing) side of a waterway. In this way, a drain will receive the benefits of shade, but access could still be maintained for mechanical removal of excess plants, or accumulations of sediment.

Extent of planting and planting density need to be matched to purpose, e.g., to achieve expected level of shade. Plant choices also depend on factors like bank instability and the need for bank battering. Ideally, any riparian planting and improvements by Council should be appropriate for purpose and a monitoring programme should also be implemented to assess the long term impacts on water quality and ecology. The results of such monitoring programmes would be important to feed into other further trails implemented to maximise riparian planting and minimise stream shade and macrophyte growth.

Providing shade to drains will not only decrease macrophyte growth, but also lead to less severe diurnal variations in oxygen concentration (Wilcock and Nagels 2001). Macrophyte beds also reduce near bed velocities in their midst (e.g., Riis and Biggs 2003; Bell et al 2013), and this leads to sediment accumulation (Lamsodis et al 2006). If macrophyte cover can be reduced through shading, then less sediment should accumulate within the drains. Moreover, provision of shade through riparian planting is also likely to reduce sediment inputs into the drains in the first place (McDowall, Wilcock et al. 2013). Thus, shade will serve multiple functions in reducing stressors such as high temperature, large diurnal oxygen fluctuations and sediment accumulation.

Macrophyte management

Macrophytes can be an important component of stream ecology, but excess macrophyte growth characteristic of many of the drains is a major stressor on their ecology. As such, macrophytes are regarded as keystone species in drains, in that their presence exerts large effects on ecosystem processes, ranging from increasing sedimentation rates (Lamsodis et al. 2006) through to lowering dissolved oxygen (Wilcock and Nagels 2001). On the other hand, macrophytes are beneficial in that they can provide attenuation of sediment within drains, a role that wetlands had before the plains were drained. Furthermore, they represent one of the few stable habitats in soft-bottom systems where wood is missing (e.g., Collier 1995; Bell et al 2013). Given this, it is important to manage macrophytes to reduce their biomass in drains, but not completely remove them. Improved management of macrophytes in the drains is therefore considered of key importance in any efforts to improve ecological conditions within the drainage network. However, any decisions to manage macrophytes needs to recognise conflicts between often competing values of the drainage network. The current management practices appear to be focusing purely on maximising hydraulic efficiency, leading to the current conditions of little (if any) riparian vegetation, minimal shade, and resultant high macrophyte cover.

Alternative management strategies may in future rely on strategies such as riparian planting to maximise shade, as well as minimise nutrient and sediment inputs into the drains. Under such conditions, macrophyte proliferations may not be as extensive and resultant dissolved oxygen fluctuations may be less. There may also be even less sediment entering the streams, as this is intercepted by riparian vegetation. Under such a scenario, it may be possible to achieve both drainage functions, as well as ecological functions.

Flow management

Another potential mitigation measure that could be employed to improve ecological conditions in the drains is to increase water movement. This could be done by a combination of installing fish friendly floodgates that can allow cleaner water to flow into the drains at high tide, and then flow out again at low tide (Figure 43).

Although this would only have a limited effect on improving water quality within the tidal prism of the drain (Franklin and Hodges 2015), this extra flushing would nevertheless be beneficial. It would also allow for movement of fish into many of the smaller drains, effectively recreating habitat for them. Retrofitting these with fish friendly floodgates would have potentially beneficial ecological effects, while minimising adverse effects to drainage.

Increasing water movement could also have beneficial results on DO levels through use of mechanical devices to aerate the water. Such devices could range from wind or solar powered paddle wheels to bubbling aerators and would be expected to greatly reduce the severity of anoxic conditions in some of the drains. Although the use of such devices would undoubtedly improve ecological conditions in the drains, such devices would be regarded as mitigation devices only, and more attention focussed on ways to avoid the development of anoxic conditions in the first place.

Fish passage

In addition to the need to ensure upstream fish passage on many of the drains throughout the Plains, alternative ways should be developed to allow migrating fish to pass through pump stations without suffering mortality. Duirs (2017), highlight that although fish friendly pumps such as archimedes screws exist, their cost is relatively high, and there is little documented evidence as to their efficacy. It may also be possible to design and implement some form of trap and transfer system in the drainage network that could be used during the autumn when eel migration is at its peak.

Work by Boubee et al. (2001) in the Rangitaiki River at the Aniwhenua Dam, showed that migrant eels commenced at downstream migration in early autumn when the catchment receives more than 20 mm of rainfall within a 12-hour period. That eels display such strong cues to local weather conditions means that it could be a relatively simple exercise to ensure that all pump stations have fish exclusion barriers in front of the pump intakes, and to then install fish traps during the first autumnal rainfall, where any downstream migrating eels could be trapped and transferred below the pump station.

Any work done to address fish passage issues either through flood gates or pump stations would also be best made on the basis of the detailed stocktake that has been undertaken of such structures throughout the Kaituna and Rangitaiki Plains, so that a prioritisation process can be made of what drains would be the most beneficial to fish communities if these mitigation measures were carried out. Such a process has been identified by Suren (2017) whereby sites identified on the basis of factors such as catchment size, distance to sea, upstream habitat conditions and water quality.

It is possible that with a few modifications to both land use practices and operational infrastructure, both water quality and ecological conditions of the drainage network could be considerably enhanced. This will not only benefit the drains themselves but is also likely to have large benefits to sensitive receiving environments into which these waterways flow.

6.3 Recommendations for further work

The proposed mitigation measures discussed above relate to three key areas:

- Increasing the amount of vegetation around drains to intercept nutrients and sediments, and to reduce macrophyte biomass in drains.
- Installation of fish friendly floodgates and either designing fish friendly pumps or transferring fish below pump stations.
- Investigating methods for macrophyte removal, including increased use of grass carp.

A large amount of targeted research is needed before any of these mitigation measures can successfully be employed. Some potential research questions are listed below under broad themes with the aim of prioritising these for further study.

Reducing macrophyte cover

For example, how much shade is required to reduce macrophyte cover in drains, and how much riparian planting is needed to reduce both nutrient and sediment inputs into the drainage network? If riparian planting was successful in reducing macrophyte cover, will this always result in a reduction in the frequency and magnitude of low dissolved oxygen periods? Will riparian planting or use of wetland plants help with the attenuation of dissolved nutrients such as nitrate-N and dissolved P?

Effects of macrophyte removal

Baseline data as to the effect of different macrophyte removal strategies (e.g., spraying, mechanical removal, use of the weed cutter boat, or use of grass carp) on native fish such as banded kokopu or shortfin eel is lacking in the Bay of Plenty. Studies investigating the movement of tagged fish under different macrophyte removal regimes could be implemented to determine what the long-term impacts of these strategies are on fish communities.

Finally, can a more focused and cohesive macrophyte removal strategy (e.g., Schwarz and Snelder 1999) be developed to ensure that such work maximises both hydraulic requirements and ecological requirements.

Improving water quality

Are there any ways of increasing the re-aeration of these generally slow flowing environments to increase oxygen levels, and subsequently reduce ammonia concentrations? What is the effect of installing fish friendly floodgates to increase the amount of water movement above existing floodgates, and will this increased water movement result in enhanced water quality outcomes? Some promising data from Franklin and Hodges (2015) has provided some insight on this.

Fish passage issues

There is an obvious need to develop an inventory of drainage infrastructure with respect to fish passage issues, and to prioritise areas where fish friendly floodgates could be retrofitted. Such an inventory is already underway in some parts of the region where road culverts and flood gates have been identified and prioritised for retrofitting with fish-friendly devices.

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Appendices

Appendix 1

List of invertebrate sampling sites in each Water Management Area (WMA), showing the calculated biotic metrics and their bands

MEV Sites with MCI scores < 80 are highlighted, as the NPSFM requires councils to investigate cause and take action where MCI is declining or where it is less than 80 (Note that such a requirement does not apply to DWQ sites).

RC_SID	WMA	Site name	Easting	Northing	Biophysical class	WQ Class	MCI Score	MCI Band	EPT1 taxa	EPT1 Band	BoP_IBI	BoP_IBI Band
BOP_DRAIN_02	Kaituna_Maketu	Bell Road Drain at Te Puke	1894698	5817897	VA/Gentle	DWQ	87	D	0	D	18	D
BOP_DRAIN_03	Kaituna_Maketu	Kaituna Drain at Pah Road	1896735	5815673	VA/Gentle	DWQ	45.5	D	0	D	9	D
BOP_DRAIN_04	Kaituna_Maketu	Kaituna Drain at Kaituna Road	1902169	5814595	VA/Gentle	DWQ	72	D	0	D	0	D
BOP_DRAIN_05	Kaituna_Maketu	Wharere Drain at Pukehina	1906387	5812911	VA/Gentle	MEV	72.3	D	1	D	15	D
BOP_DRAIN_07	Kaituna_Maketu	Pukehina Drain at Pukehina	1907565	5812877	VA/Gentle	MEV	67	D	0	D	0	D
BOP_DRAIN_08	Kaituna_Maketu	Pongakawa Drain at Cutwater Road	1906928	5813003	VA/Gentle	MEV	64.5	D	0	D	0	D
BOP_DRAIN_09	Tarawera	Awakaponga Canal	1930945	5797125	VA/Gentle	MEV	67.3	D	1	D	6	D
BOP_DRAIN_10	Tarawera	Section 109	1932924	5796827	VA/Gentle	DWQ	59.2	D	0	D	9	D
BOP_DRAIN_09	Tarawera	Awakaponga Canal	1930945	5797125	VA/Gentle	MEV	67.3	D	1	D	6	D

RC_SID	WMA	Site name	Easting	Northing	Biophysical class	WQ Class	MCI Score	MCI Band	EPT1 taxa	EPT1 Band	BoP_IBI	BoP_IBI Band
BOP_DRAIN_11	Tarawera	Awaiti Canal	1933519	5794286	VA/Gentle	DWQ	80	D	0	D	9	D
BOP_DRAIN_12	Tarawera	Omehue Canal upstream WWTP	1934562	5789921	VA/Gentle	MEV	55	D	0	D	3	D
BOP_DRAIN_13	Tarawera	Omehue Canal downstream WWTP	1935295	5791681	VA/Gentle	MEV	24	D	0	D	9	D
BOP_DRAIN_14	Rangitaiki	Reids Central Canal	1938326	5792045	VA/Gentle	MEV	42	D	0	D	0	D
BOP_DRAIN_15	Rangitaiki	Western Drain	1938530	5788430	VA/Gentle	MEV	51.5	D	1	D	15	D
BOP_DRAIN_16	Whakatane	Eastern Drain	1942045	5790393	VA/Gentle	DWQ	72	D	0	D	3	D
BOP_DRAIN_17a	Whakatane	Waioho Stream upstream of Drain_18	1948129	5788962	VA/Steep	MEV	55.6	D	0	D	0	D
BOP_DRAIN_17b	Whakatane	Waioho Stream downstream of Drain_18	1948129	5788962	VA/Steep	MEV	46.0	D	0	D	6	D
BOP_DRAIN_18	Whakatane	Langenberger Road Drain	1947950	5788941	VA/Gentle	DWQ	74	D	0	D	3	D
BOP_DRAIN_19	Whakatane	Te Rahu Canal	1947277	5790826	VA/Steep	MEV	50	D	0	D	3	D
BOP_DRAIN_21	Whakatane	Orini Canal off Thornton Road	1944957	5794132	VA/Gentle	MEV	78	D	0	D	0	D

RC_SID	WMA	Site name	Easting	Northing	Biophysical class	WQ Class	MCI Score	MCI Band	EPT1 taxa	EPT1 Band	BoP_IBI	BoP_IBI Band
BOP_DRAIN_22	Tarawera	Secombes Canal	1934753	5797959	VA/Gentle	DWQ	102	C	0	D	18	D
BOP_GAPS_13	Kaituna_Maketu	Raparapahoe at Above Drop Structure	1891609	5815319	VA/Steep	MEV	134	A	8	B	24	D
BOP_LOP_1	Kaituna_Maketu	Lawrence_Oliver_ Park_A	1893617	5812731	VA/Gentle	DWQ	76.5	D	3	C	12	D
BOP_LOP_2	Kaituna_Maketu	Lawrence_Oliver_ Park_B	1893656	5812881	VA/Gentle	DWQ	67	D	1	D	6	D
BOP_LOP_3	Kaituna_Maketu	Lawrence_Oliver_ Park_C	1893693	5812998	VA/Gentle	DWQ	81.1	D	2	C	21	D
BOP_LOP_4	Kaituna_Maketu	Lawrence_Oliver_ Park_Lower	1893905	5813230	VA/Gentle	DWQ	59.8	D	1	D	15	D
BOP_NERM_003	Kaituna_Maketu	Kaikokopu	1904961	5810715	VA/Gentle	MEV	94.7	C	14	A	36	D
BOP_NERM_005	Kaituna_Maketu	Raparapahoe	1891279	5814963	VA/Steep	MEV	115.4	A	26	A	18	D
BOP_NERM_026	Whakatane	Te Rahu Canal	1942463	5787849	VA/Steep	MEV	86.4	C	12	A	0	D
BOP_NERM_027	Tarawera	Awakaponga Canal at Matata Road	1930799	5794874	VA/Gentle	MEV	75.5	D	6	C	21	C
BOP_NERM_033	Kaituna_Maketu	Pongakawa at State Highway 2	1909234	5808797	VA/Gentle	MEV	99.0	C	16	A	36	D
BOP_NERM_034	Kaituna_Maketu	Puanene	1905579	5807979	VA/Gentle	MEV	105.2	C	12	A	33	D
BOP_NERM_049	Rangitaiki	Ngakauroa Creek	1936844	5784480	VA/Gentle	MEV	92.6	C	13	A	27	A

RC_SID	WMA	Site name	Easting	Northing	Biophysical class	WQ Class	MCI Score	MCI Band	EPT1 taxa	EPT1 Band	BoP_IBI	BoP_IBI Band
BOP_NERM_051	Whakatane	Waioho	1948048	5787392	VA/Steep	MEV	104.5	B	17	A	6	D
BOP_RES_083	Rangitaiki	Rangitaiki Canal Bennetts Road	1935430	5798490	VA/Gentle	DWQ	86	D	0	D	3	D
BOP_RES_085	Rangitaiki	Rangataiki Canal Smiths Road	1937525	5796832	VA/Gentle	DWQ	55.3	D	0	D	18	D
SENV_PPS_PD	Kaituna_Maketu	Ohineangaanga_DS1	1891688	5814008	VA/Gentle	DWQ	63.6	D	0	D	18	D
SENV_PPS_PD2	Kaituna_Maketu	Ohineangaanga_DS2	1892754	5814580	VA/Gentle	DWQ	46.5	D	0	D	3	B
SENV_PPS_PU	Kaituna_Maketu	Ohineangaanga_Upstream	1891549	5813697	VA/Gentle	DWQ	53.7	D	0	D	9	A
URS_MATATA_1a	Tarawera	Orini Stream_above Greig Road	1935793	5798382	VA/Gentle	DWQ	73.2	D	0	D	9	D
URS_MATATA_1b	Tarawera	Orini Stream_above Greig Road	1935793	5798382	VA/Gentle	DWQ	76.9	D	0	D	9	D
URS_MATATA_2a	Tarawera	Orini Stream_below LTS outlet	1934111	5798752	VA/Gentle	DWQ	85.3	D	0	D	9	B
URS_MATATA_2b	Tarawera	Orini Stream_below LTS outlet	1934111	5798752	VA/Gentle	DWQ	87.3	D	0	D	9	C
URS_MATATA_3a	Tarawera	Orini Stream_above flood gate	1933314	5799083	VA/Gentle	DWQ	84.3	D	0	D	9	C
URS_MATATA_3b	Tarawera	Orini Stream_above flood gate	1933314	5799083	VA/Gentle	DWQ	74.1	D	0	D	9	C