The Ōtūkaikino River: Factors Contributing to Apparent Macroinvertebrate Loss

A thesis submitted in partial fulfilment of the requirements for the Degree of Master of Water Resource Management

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Photo: Ōtūkaikino Creek in July 2019

Abstract

In recent years the Ōtūkaikino River catchment has had some of best water quality and stream health of all Christchurch rivers. However, the absence of stoneflies from macroinvertebrate surveys in 2017 indicates that all may not be well with the catchment. Given stoneflies are typically associated with high habitat and water quality, their decline or disappearance may signal stream health challenges that require further attention.

A 12-month monitoring programme was created for the Ōtūkaikino River catchment to determine potential sources of pollution and habitat limitation related to this apparent decline. A range of physical, chemical and biological parameters were investigated across ten sites in the catchment in 2019 and 2020.

The moderately pollution sensitive cased caddisfly *Pycnocentria* was a dominating taxon at many sites, though pollution tolerant *Potamopurgus* snails and pollution sensitive *Deleatidium* mayflies were also typically in high numbers. Four *Zelandobius* stoneflies were identified in catchment monitoring surveys in 2019, indicating that the catchment is still able to support populations of stoneflies, despite the apparent decline between 2008 and 2017. Metrics for ecological health generally increased downstream towards the middle reaches. Site scores ranged from poor ecological health (upper Waimakariri South Branch) to excellent (middle reaches). This differed to the generally good-excellent ecological health reported in 2017.

Low concentrations of trace elements suggested they were generally not a key contributor to changes in ecosystem health in the Ōtūkaikino River catchment. Key exceptions were dissolved arsenic, chromium, copper and zinc at some sites. In sediment, metal concentrations were generally low, except for two headwater sites of the Waimakariri South Branch. These two sites recorded high levels of most parameters analysed, with lead and copper exceeding ANZECC (2000) interim sediment quality guidelines.

Most other water quality parameters were within ANZECC (2000) water quality guidelines for ecosystem protection, with a few key exceptions. Dissolved oxygen reached low concentrations at several sites. Elevated levels of faecal coliforms were recorded in some samples, though *E. coli* was comparatively low. While nitrate-nitrogen concentrations were low, DRP (dissolved reactive phosphorus) was consistently elevated above ANZECC (2000) water quality guidelines at one site, as was ammoniacal nitrogen at several sites.

The main factors identified in this study that contributed to this variation in macroinvertebrate community health and water quality were differences in riparian and canopy cover. They were typically highest in the middle reaches, though there were some other areas of thick vegetation. In particular, much of the upper reaches had limited mature shading plants and sediment filtering plants. Localised inputs, such as trace elements in two Waimakariri South Branch sites, were also potential contributors.

Substantial planting efforts have occurred in the catchment in the last couple of decades. This study recommends that these efforts continue, with a focus on intercepting sediment and shading the waterway. Further monitoring and research in the vicinity of the two sites where high levels of trace elements were recorded is also recommended.

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1 Introduction

Globally, the degradation of waterways and its impact on aquatic ecology has been receiving increasing attention over the last couple of decades as one of the key issues facing our world today (Glasgow & Burkholder, 2000; Ramstack et al., 2004). Over the last 200 years, many waterways have undergone significant alternation as a result of human activities, including urbanisation and agricultural intensification, resulting in a reduction in the ecosystem functions and services that they provide (Booth et al., 2016; Glasgow & Burkholder, 2000).

However, despite decades of research, effective management and mitigation strategies have been slow to emerge (Harding et al., 1999). Some cases of water degradation can occur slowly over long periods of time and as a result of numerous small but cumulative inputs (Harding, 1992). This makes a decline in water quality difficult to recognise before its effects become severe (Harding, 1992). It is further made difficult when there is a lack of clear point-source contaminant discharges (Ongley et al., 2010). This means that long-term water degradation can be both difficult to detect and difficult to reverse. Consequently, there is a need to identify early warning signs of potential degradation. In this research, the apparent loss of stoneflies in a catchment were assessed as a warning sign and possible causes were investigated.

1.1 Urbanisation Effects on Freshwater Systems

Urbanisation is a growing form of land use change. Though this is not a new phenomenon, the immense scale and intensity of urbanisation is of increasing concern. As of 2008, more than 50% of the world's population were living in urban areas, with urban populations projected to hit 5 billion by 2030 and 6.4 billion by 2050 (Gaston et al., 2013; Heilig, 2012). This population growth is typically coupled with increased urban expansion and its resulting impacts. However, the degree to which this will occur globally over the next few decades is harder to predict than urban population growth, as it is dependent on the rate of change in urban densities, which is highly variable between cities (Gaston et al., 2013).

New Zealand has followed this trend (Figure 1.1). Near the end of the 19th century, about 60% of the country's population lived in rural areas (Statistics New Zealand, 2006). By 2016, this had changed dramatically, with 86% of the population living in urban areas (The World Bank, 2020).

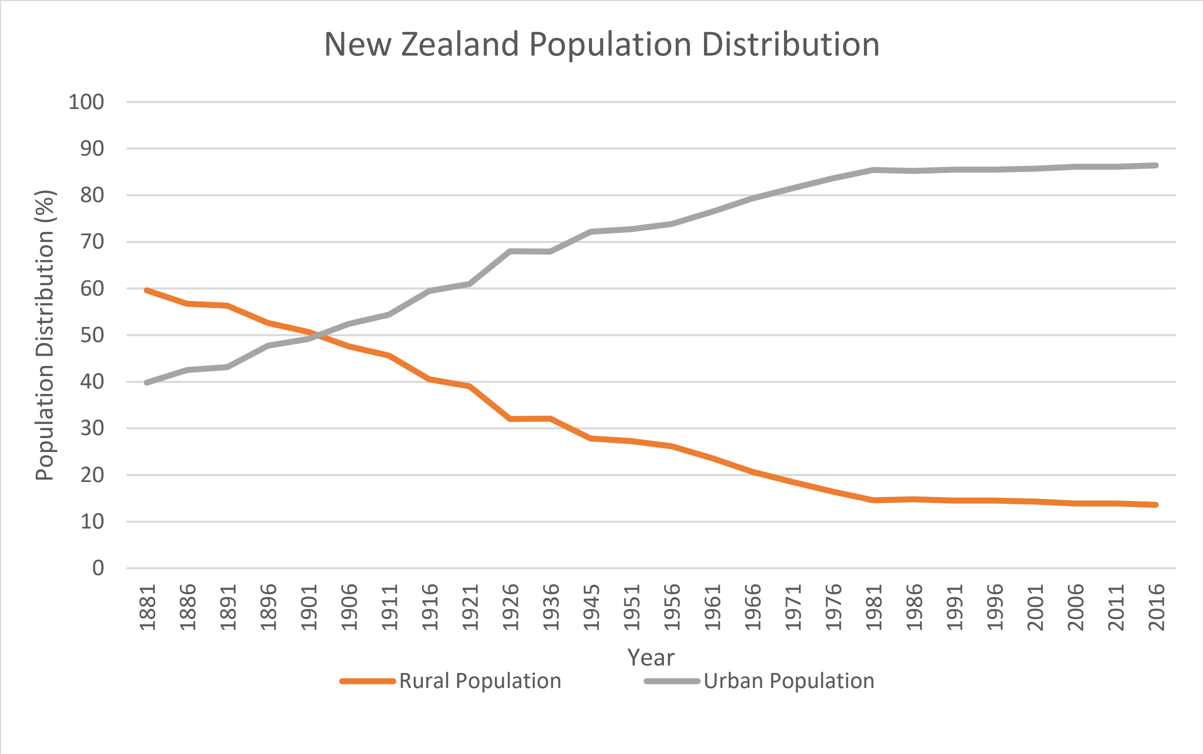

Figure 1.1: New Zealand urban and rural population distribution between 1881 and 2016 (adapted from Statistics New Zealand, 2006; The World Bank, 2020).

Urbanisation as a form of land use change has a substantial impact on the surrounding environment, particularly waterways. Streams are a common occurrence in modern urban settings, and impacts of increasing urbanisation on their appearance and function are well-studied globally (Beesley et al., 2016; Hale et al., 2016). The term "urban stream syndrome" was popularised by papers such as Meyer et al. (2005) and refers to the multiple impacts that urbanisation has on a waterway (Booth et al., 2016). Despite the similarities between urban streams, specifically with respect to degraded chemical, biological and physical factors, there are still meaningful differences in the drivers and intensity of impacts between catchments and in how they respond to the surrounding urban environment (Hale et al., 2016). Consequently, responses to urbanisation are typically catchment-specific.

1.1.1 Impervious Surfaces

A key driver of urbanisation effects on waterways is the increased proportion of impervious surfaces in the surrounding catchment. A typical forested catchment might have between 5 and 15% of its rainfall end up as overland flow, with the remainder either infiltrating into the ground or being evaporated and transpirated (Bolund & Hunhammar, 1999). However, impervious surfaces, such as roads and roofs, disrupt these natural hydrological regimes through reduced infiltration, causing increased surface runoff in rain events (Chithra et al., 2015). In areas where impervious cover exceeds 75%, for example, surface runoff might be 50% or more of the rainfall, with the majority of this runoff ending up in local waterways (Paul & Meyer, 2001). This in turn can impact groundwater recharge in the catchment, as well as causing increased peak surface discharges during flood events (Paul & Meyer, 2001). This altered hydrology increases the severity and frequency of high flow events, which is a key characteristic of the "urban stream syndrome" (Hale et al., 2016). Often, it is further compounded by the storm water systems, which funnel surface flows directly into waterways (Makepeace et al., 1995).

1.1.2 Water Quality

In urban environments, surface runoff typically contains high levels of contaminants, including trace metals. These are present in both the sediment and the water column of urban waterways, and are a key distinguishing feature of urban waterways, compared to waterways affected by other types of land use (Islam et al., 2015). In New Zealand, the trace metals of greatest concern are zinc, copper and lead (Abrahim & Parker, 2002). Nickel, chromium and aluminium, among others, can also be fairly common (Abrahim & Parker, 2002; Aryal et al., 2010). This concern stems from their toxicity to aquatic taxa, as well as their persistence and prevalence in the environment (Brown & Peake, 2006). Contaminant sources, and their levels, will vary greatly depending on the activities within the catchment, with road and building runoff being two key contaminant sources (Müller et al., 2019).

Traffic is a key contaminant source in urban catchments. The wearing of brake lining produces copper and zinc (O'Sullivan et al., 2012). Abrasion with the surface of roads creates tyre dust containing zinc, alongside low levels of lead, cadmium and copper (Brown & Peake, 2006). Lead was historically used in leaded fuel and legacy traces of this are still present today (Göbel et al., 2007). Lead is also still present in some current vehicle tyre weights (Müller et al., 2019). These can all potentially enter urban waterways during rain events through surface runoff.

Another key source of potential contaminants in an urban setting is building runoff. Trace metals are commonly used in roofing design: zinc for galvanising, copper for roof surfaces and guttering, and lead in solders and in older paints (Müller et al., 2019). The prevalence of lead in urban runoff, however, has been decreasing over time, partly due to increased regulation of lead-based paint (Kayhanian et al., 2012).

1.1.3 Physical Changes

The urbanisation of a catchment also brings many physical changes. Urban heat islands can develop, where air temperatures are noticeably higher in urban areas compared to the surrounding rural land (Mohajerani et al., 2017). Common causes include altered energy budget from increased dark surfaces, decreased evapotranspiration through removal of vegetation, and high levels of pollution (Mohajerani et al., 2017). One consequence of urban heat islands is a large increase in water temperatures (Müller et al., 2019). This can cause decreased dissolved oxygen levels, potentially to anoxic levels, which can have severe impacts on sensitive species like trout (Paul & Meyer, 2001).

As riparian vegetation plays an important role in bank stabilisation, its removal for urban development can contribute to an increased sediment load of waterways. This sediment may enter the waterway directly through soil erosion, breakdown of surfaces and from vehicle wear (Marsalek et al., 2014; Murugan et al., 2008) or indirectly through atmospheric deposition (Murphy et al., 2014). High levels of suspended sediment can reduce food and habitat quality, limiting hyporheic exchange between the benthos and the flowing water (thereby reducing egg survival in some fish species), and affecting light penetration and therefore the growth of macrophytes (Bilotta & Brazier, 2008; Nogaro et al., 2009). Additionally, as some trace metals, including lead, are poorly soluble, they tend to enter waterways entrained to sediment, further exacerbating the issue of contamination (Islam et al., 2015; Trujillo-González et al., 2016).

Altered channel geomorphology, including changed drainage density, is another common physical change from urbanisation. Natural springs and channels (especially small streams) are often paved over, while artificial channels can contribute to increased connectivity of waterways and consequently to increased flood velocity (Paul & Meyer, 2001). Additionally, bed aggradation during urban construction, compounded by subsequent erosion as increased imperviousness in turn increases flows, can lead to gradual channel incision and widening (Gurnell et al., 2007; Paul & Meyer, 2001).

1.1.4 Effects of Urbanisation on Aquatic Ecology

Water quality and quantity changes as a result of urbanisation can have significant impacts on aquatic communities. Despite the importance of many trace metals for maintaining aquatic life, including copper for metabolism, they can be acutely toxic at elevated levels (Beasley & Kneale, 2002). Elevated dissolved copper levels can lessen reproductive and growth rates in aquatic organisms, and potentially contribute to increased mortality (Charters, 2016). Increased zinc in waterways can, for example, have neurotoxic effects, including increased oxidative stress and altered energy metabolism (Koh, 2001). In invertebrates, chronic lead exposure impacts immune response, growth, reproduction, behaviour and development, while acute lead poisoning can lead to death (Ryan et al., 2019; Van Sprang et al., 2016). Elevated trace metals, including mercury, can also be a human health risk as they bioaccumulate in organisms and can subsequently be passed up the food chain (Murugan et al., 2008).

The general impacts of urbanisation on benthic invertebrate communities tend to be relatively consistent across waterways globally, hence their use as an indicator of water quality and land use change (Moore & Palmer, 2005; Purcell et al., 2009). Urban waterways generally have less diverse macroinvertebrate communities compared to local unimpacted sites, though the species richness can often stay relatively high (Paul & Meyer, 2001). Macroinvertebrate community composition of urban streams is typically dominated by pollution tolerant taxa like *Potamopyrgus* (Paul & Meyer, 2001). More sensitive taxa – in New Zealand, these are typically *Ephemeroptera* (mayflies), *Plecoptera* (stoneflies), and *Trichoptera* (caddisflies) (EPT) - tend to be less found at comparatively lower numbers, or may be absent completely (Thompson & Parkinson, 2011). However, the response of individual taxa is complex and varies greatly depending on other factors involved, including water hardness and levels of dissolved organic matter (Harding, 2005).

1.2 Agriculture

Despite this global urban shift, agriculture remains another key land use that is impacting waterways, particularly over the last century (Greenwood et al., 2012). As the global human population continues to increase, so too does the demand for food. Despite the increase in production efficiency through the so-called "Green Revolution" and technologies like chemical fertilisers, agricultural land expansion continues to occur (Gibbs et al., 2010). Some projections suggest that up to 10 billion hectares of additional land will be required by 2050 to meet growing global demands (Gibbs et al., 2010; Tilman et al., 2001). Though land intensification reduces the need for further land conversion to agriculture, it also introduces problems like increased nutrient leaching and the subsequent degradation of soil and water (Mózner et al., 2012).

The effects of this increased intensity and coverage of agriculture are heavily felt by waterways. In New Zealand, non-point source pollution from agriculture is acknowledged as a key driver of degrading water quality (Monaghan et al., 2008). With this has come increased public awareness of agricultural effects, though changes in agricultural practices will take time to implement (Cullen et al., 2006).

1.2.1 Nutrients

High nutrient applications can have significant consequences (Cullen et al., 2006). Both nitrogen and phosphorus are essential nutrients for life. However, increasing levels of agricultural use are having pronounced effects on the environment, including how people can interact with rural waterways. These can also be an issue in urban catchments, particularly through wastewater inputs. However, the levels of these are often not as pronounced as those from agriculture and inputs are frequently from outside the urban environment (Metson et al., 2017).

Some forms of nitrogen are highly soluble and mobile, so readily leach through soil to end up in groundwater and surface water (Casey et al., 2002). In dairy farms, the main source of nitrogen leaching is typically patches of animal urine in grazed pastures (Monaghan et al., 2008). Chemical fertiliser application and animal manure are also common contaminant sources of nitrogen to water from agricultural landscapes (Monaghan et al., 2008).

Phosphorus, however, tends to adsorb to sediment, so is more likely to enter waterways through erosion and surface flow (McDowell & Sharpley, 2002). Specific sources of phosphorus from agriculture can include effluent ponds, fertiliser application and dung patches (Monaghan et al., 2008).

The effects of nutrient enrichment of waterways are well understood, though numerous interactions between anthropogenic and natural factors make their severity unpredictable (Brooks et al., 2016; Wilkinson et al., 2018). Increased levels of nitrogen and phosphorus can stimulate growth in periphyton and phytoplankton, leading to decreased light penetration as well as depleted oxygen levels as the periphyton decomposes (McDowell et al., 2013). In the case of some cyanobacteria blooms, the toxins secreted can be toxic to humans and animals (Wilkinson et al., 2018).

1.2.2 Other Water Quality Impacts and the Response of Invertebrates

Faecal contamination is a common impact of agricultural activities. In New Zealand, key inputs of this are unfenced stock access, drains and surface runoff (McDowell et al., 2013). Faecal contamination can also be an issue in urban streams, particularly from avian sources and poor sewage management (Paruch et al., 2015). Faecal contamination from livestock, and other sources, can contain a range of pathogenic microorganisms, such as species of *Campylobacter* and *Salmonella*, that are harmful to human and animal health (Mateo-Sagasta et al., 2017).

As in urban waterways, increased sediment input can be another key component of water quality degradation in agricultural streams. Agricultural intensification typically involves the clearance of riparian margins to allow for channel straightening, ease of access and the use of additional productive land (Landemaine et al., 2015). This can leave exposed banks, heightening the risk of sediment erosion, as well as alter temperature and light levels in the waterway (Lange et al., 2014).

Pesticides are used widely by the agricultural sector (among others) for weed species control, as well as unwanted fungi and insect species (Allinson et al., 2016). While effects vary between different types of pesticides, a key concern is their environmental persistence, especially for those that can bioaccumulate and biomagnify (Kim et al., 2017). Many are not species-specific, so can harm non-target species (Carvalho, 2017). For example, DDT, which is now banned globally, contains persistent organic pollutants that allow pesticide residue to remain in the soil for years (Kim et al., 2017).

The general response of aquatic invertebrate and fish communities to increased agricultural intensity tends to be similar to their response to urbanisation; both result in a shift in species composition towards taxa that are more tolerant to pollution (Harding et al., 1999; Lange et al., 2014).

1.3 Ōtūkaikino Catchment Overview

One area that has been impacted by land use change is the Ōtūkaikino River catchment in Canterbury. Lying between 3 and 34 metres above sea level, the Ōtūkaikino River catchment covers an area of approximately 16 km^2 (Figure 1.2). It is located to the north of Christchurch and is a tributary of the Waimakariri River.

Until the 1930s, the Ōtūkaikino River catchment was known as the South Branch of the Waimakariri River. Cross Bank, near McLeans Island, was constructed in the 1930s as part of ongoing work since 1860 to control the movement of the Waimakariri River for reducing the flood hazard posed to early Christchurch (Hudson, 2005). This separated what is now the upper reaches of the Ōtūkaikino River from the Waimakariri River, forming the catchment that can be seen today. The change between 1865 and 1987 across the catchment is shown in Figure 1.3. The Ōtūkaikino River is now spring-fed from shallow groundwater seeping through gravels from the Waimakariri River. Numerous tributaries run through mostly rural land until their confluence with the main stem of the Waimakariri River at State Highway 1. The near-surface geology of the catchment is mostly composed of "grey river alluvium beneath plains or low-level terraces" (Forsyth et al., 2008). Due to the proximity of the Ōtūkaikino River to the Waimakariri River and the underlying geology, the water table is high.

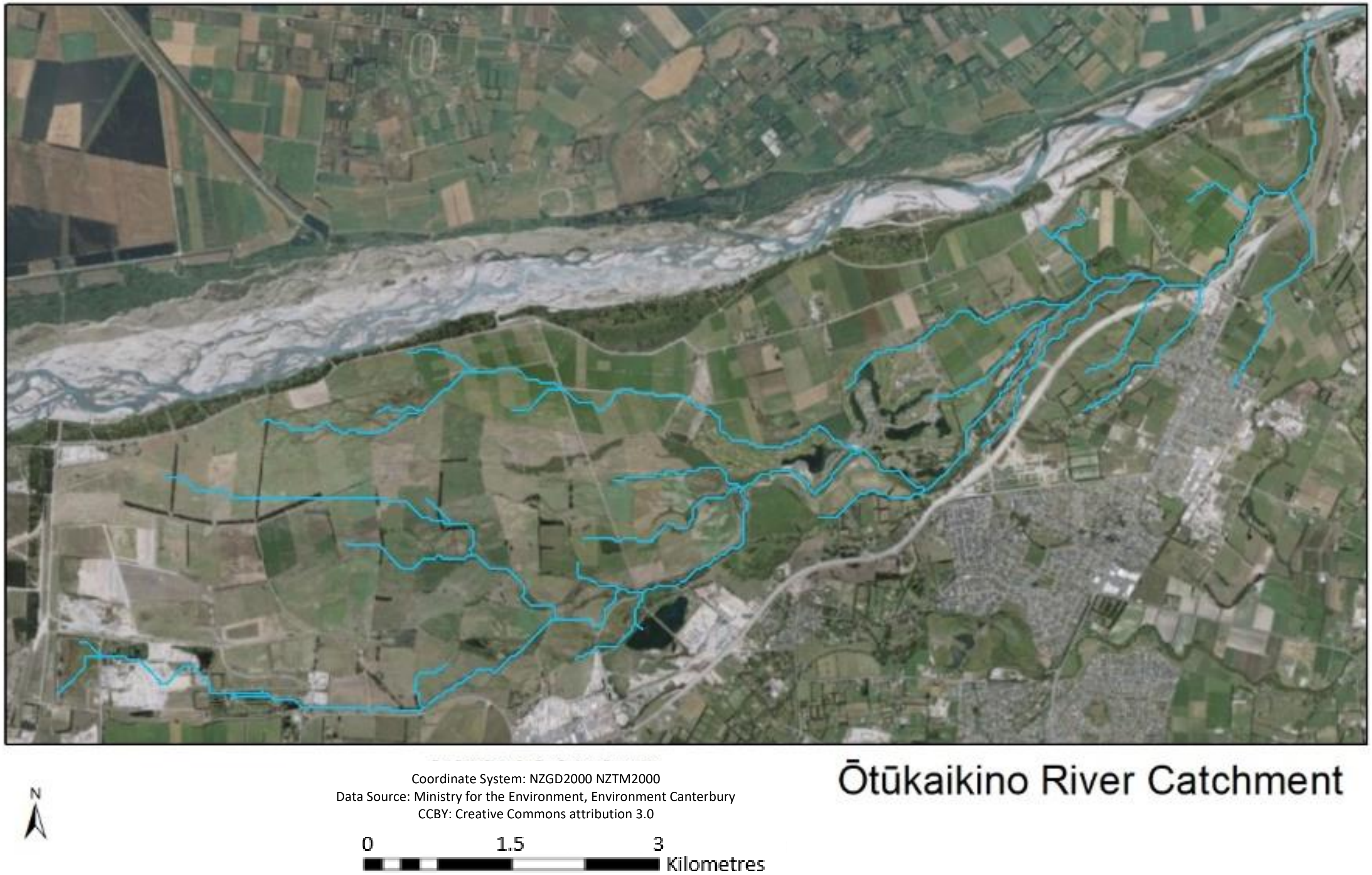

Figure 1.2: The Ōtūkaikino River catchment.

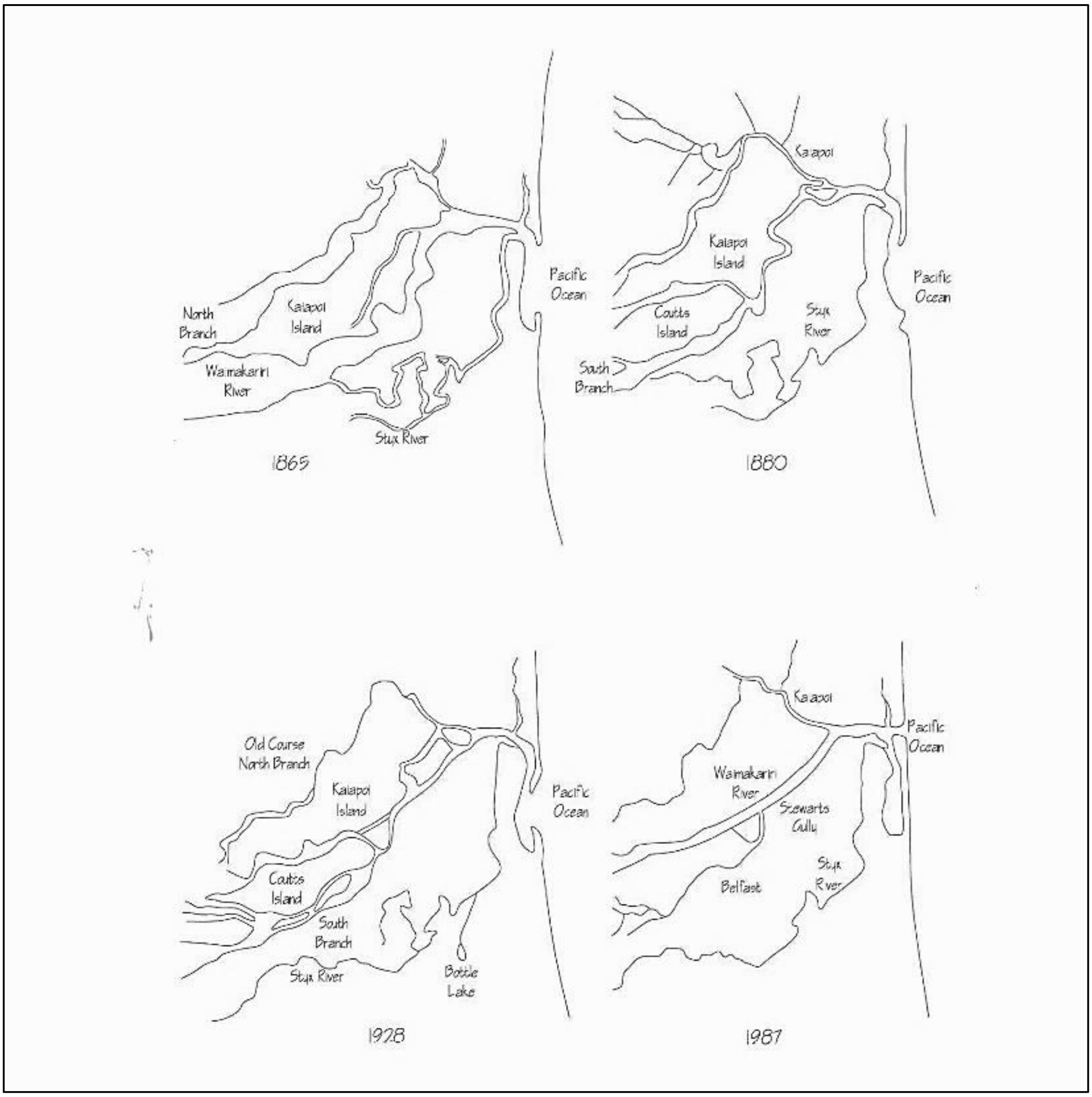

Figure 1.3: Lower Waimakariri River: 1865, 1880, 1928 and 1987 (Boyle, 2011).

1.3.1 Land Use

Historically, most of the Ōtūkaikino River catchment area has been rural and continues to be so (Figure 1.4). Exotic grasslands make up much of the catchment area, alongside some horticulture (Figure 1.4). The Groynes, a large recreational park, is situated in the middle to lower catchment (Figure 1.5).

There are several other important land uses in the catchment (Figure 1.5). The headwaters of the Ōtūkaikino Creek contain aviaries, run by The Isaac Conservation and Wildlife Trust. Captive breeding programmes are run for critically endangered species, including orange-fronted parakeet and black stilt. As these aviaries are cantilevered over the waterway, there is the potential for inputs of faecal contamination (Wither et al., 2005). This could have implications for users downstream.

Isaac Construction's quarry is also located within the headwater reaches of the Ōtūkaikino Creek. Though this is strictly managed to reduce its environmental impacts, minor effects on water quality – specifically on hardness, conductivity and alkalinity – and quantity have been noted (Cotterill, 2016).

The golf course at Pepper's Clearwater Resort, which finished construction in 2002, is in the middle of the catchment, and includes the convergence of the two main stems that feed the Ōtūkaikino River. There are potential effects of this on water quality in the catchment, both through pesticide and nutrient runoff from the golf course and through storm water inputs from the resort itself, though the effects of this will depend on the specific management practices used (Graves et al., 2004; Müller et al., 2006; Winter & Dillon, 2005). During 2019, there was significant bank landscaping undertaken within the golf course on the Waimakariri South Branch. This led to visible impacts on water clarity during high rainfall events, as well as during and directly after the work was undertaken.

In the lower part of the Ōtūkaikino River catchment, increasing urban sprawl effects are being felt. As of 2013, the Ōtūkaikino River catchment had 5193 permanent residents (Statistics New Zealand, 2013). Though Belfast township lies within the Styx River catchment, some sources estimate that up to half of the town's surfaces drain into the main stem of the Ōtūkaikino River through Johns Drain and Wilsons Drain, with the other half draining to the Styx River (Christchurch Engineering Lifelines Group & University of Canterbury Centre for Advanced Engineering, 1997). This is a rapidly growing part of greater Christchurch. The population of the Belfast area (including Belfast South) was 7809 in 2013, up from 6429 in 2006 (Statistics New Zealand, 2013). Before 2041, it is predicted that this population will have reached 15,000 people (Christchurch City Council, 2010). One of the main new residential areas to be developed lies in the lower Ōtūkaikino River catchment, around Johns Road and Main North Road (Christchurch City Council, 2010). Part of this construction has already been completed. If not managed well, the ongoing urban development could potentially lead to decreased water quality, exacerbated by the increased impervious surfaces and stormwater inputs.

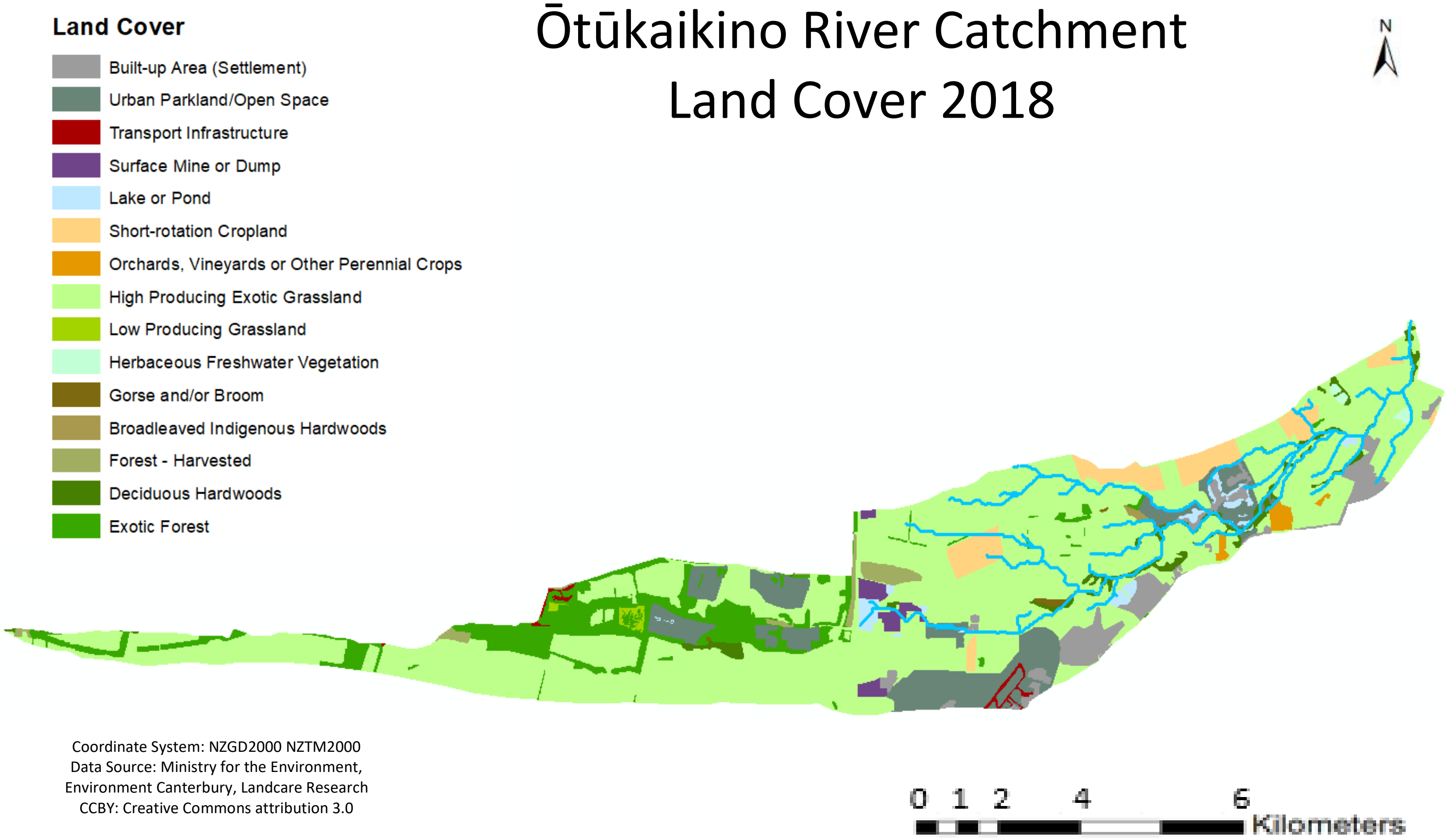

Figure 1.4: Key land cover in the Ōtūkaikino River catchment during 2018. Categories follow those used by Landcare Research (2020). Further explanation of these is included in Appendix 3.

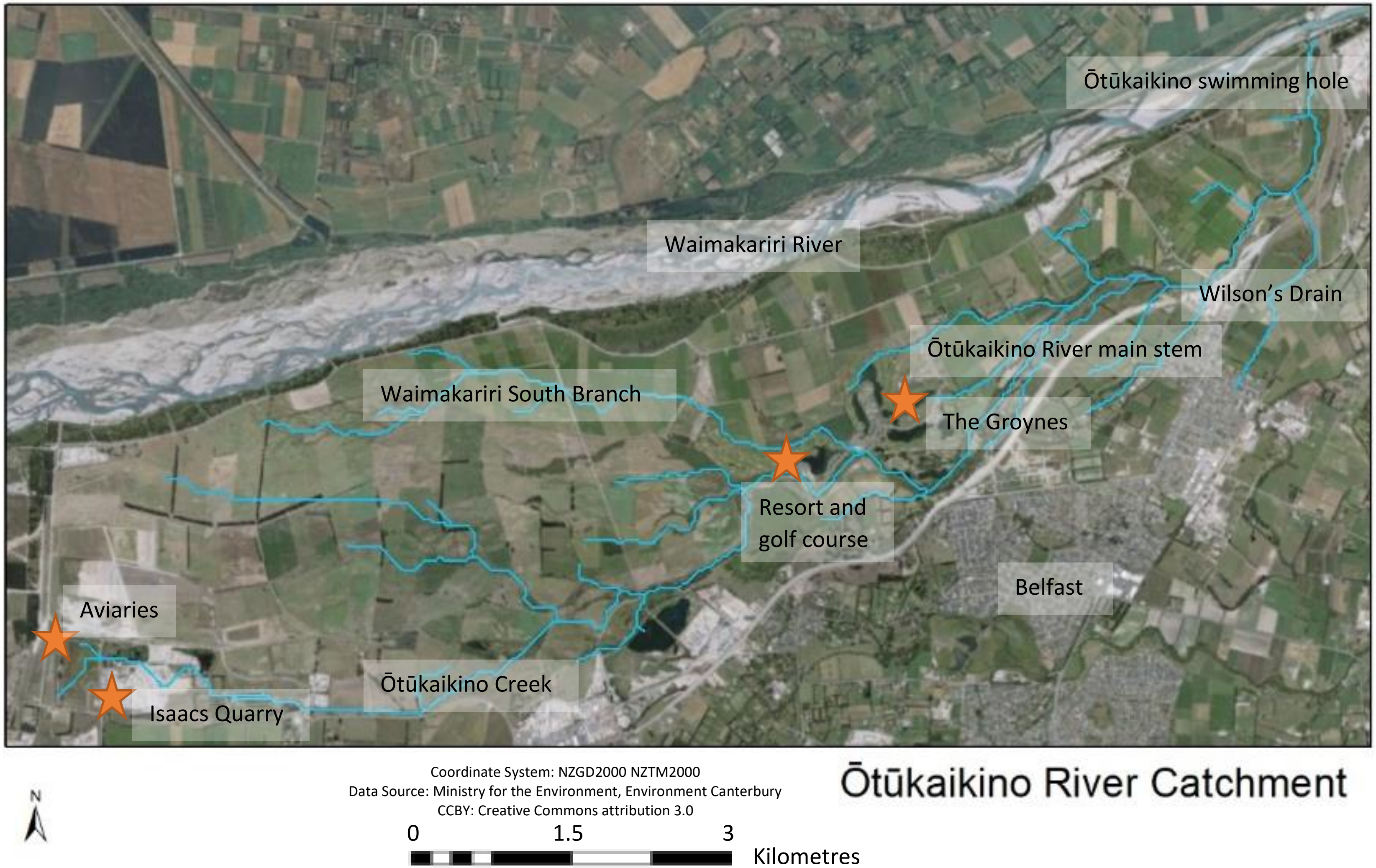

Figure 1.5: Key locations in the Ōtūkaikino River catchment.

1.3.2 Current Monitoring in the Catchment

The Ōtūkaikino River is part of a five-yearly monitoring programme for aquatic macroinvertebrates and sediment. This is run by the Christchurch City Council and covers the main waterways across greater Christchurch. Nine sites are monitored across the catchment, which have been sampled three times so far – in 2008, 2012 and 2017. Three separate sites are also sampled monthly by Environment Canterbury for annual State of the Environment reporting. The Christchurch City Council samples an additional three sites in the catchment, under the requirements of the Interim Global Stormwater Consent, South-West Stormwater Management Plan and the Styx Stormwater Management Plan (Margetts & Marshall, 2018). The Ōtūkaikino River swimming hole, just upstream from the confluence with the Waimakariri River (Figure 1.5), undergoes weekly sampling by Environment Canterbury over the summer to check its suitability for contact recreation.

In 2006, the Christchurch City Council completed a broad-scale habitat assessment of the Ōtūkaikino River, titled the Christchurch River Environment Assessment Survey (CREAS) (EOS Ecology, 2008). This involved recording habitat conditions and other site characteristics every 50 m across wadeable sections where there was permanent flow. Parameters recorded included bank attributes, channel attributes, river cover, and riparian vegetation. As of this thesis, the study has not been repeated in full.

The Ōtūkaikino River is the only major waterway in Christchurch that lacks a community group undertaking more frequent monitoring. This limits the spatial and temporal intensity of available water quality data. The Styx Living Laboratory Trust previously used one site just below Dickey's Road as a control for their invertebrate monitoring programme, though this programme stopped operating after the 2011 Christchurch earthquakes (The Styx Living Laboratory Trust, 2019).

There have been numerous restoration projects in the catchment over the last couple of decades. The majority of tributaries are now fenced and the catchment has been extensively planted in some locations, with an estimated 195,000 native and locally sourced plants introduced (Gorman, Nov 13 2018). This work is still ongoing.

1.3.3 Current State and Potential Areas of Concern

In recent years, the Ōtūkaikino River catchment has had some of best water quality and stream health of all Christchurch rivers (Margetts & Marshall, 2018). In fact, the river won Most Improved River at the 2018 New Zealand River awards, selected from all rivers across the country. This was primarily based on its phosphorus levels, with a 17.5 % decrease per annum over the last decade (Gorman, Nov 13 2018). Consequently, it hasn't been considered a focus area by the Christchurch City Council or Environment Canterbury until recently (Margetts & Marshall, 2018). However, the absence of stoneflies (aquatic invertebrates that are sensitive to pollution and land use change) in the 2017 macroinvertebrate surveys indicates that all may not be well with the catchment (Boffa Miskell Limited, 2017).

Macroinvertebrate communities in the Ōtūkaikino River catchment generally record increased diversity and abundance compared to other Christchurch waterways. Overall taxonomic richness, at 23 to 30 taxa per site, was similar across the catchment in 2017 (Boffa Miskell Limited, 2017). A significant number of these were sensitive EPT taxa, found at all sites, particularly *Deleatidium* mayflies and various caddisflies. Stoneflies (specifically *Zelandobius*), which are normally quite sensitive to pollution, were found at three sites in 2008, one site in 2012, and no sites in 2017 (Figure 1.6) (Boffa Miskell Limited, 2017; EOS Ecology, 2012). This is particularly concerning because the Ōtūkaikino River is the only Christchurch catchment where stoneflies have been found recently. The total number of EPT taxa was also generally lower in 2017 than in previous years, though this may be related to the altered methodology. Using MCI (Macroinvertebrate Community Index) scores, all sites were rated "fair" in 2017, indicating probable pollution.

As a result of the potential loss of a sensitive species from the catchment, a follow up intensive survey was conducted by Boffa Miskell of two of the three sites where *Zelandobius* were previously present (B. Margetts & M. Stevenson, personal communication, 13 April, 2018). No stoneflies were found during this sampling, though it was proposed that they could potentially still be present in other parts of the catchment, particularly near the headwaters.

This potential loss of one taxon from the sites monitored in the Ōtūkaikino River catchment is concerning for several reasons. Given stoneflies are typically associated with high habitat and water quality, their decline or disappearance may signal stream health challenges that require further attention. This in turn would indicate a need for increased management focus for the catchment to address the reasons for this degradation.

As mentioned, the Ōtūkaikino River is the only main waterway in greater Christchurch where stoneflies have been officially found in recent times. Surveys conducted in the 1980s and 1990s reported EPT taxa in numerous waterways across Christchurch, though these taxa are now typically not present at the same locations (Suren & McMurtrie, 2005). This is representative of a general trend in changing aquatic macroinvertebrate community composition across the city (Suren & McMurtrie, 2005). As this leaves the Ōtūkaikino River catchment as the last remaining stronghold of some of the more sensitive taxa in greater Christchurch, the potential loss of one such taxon is of concern.

Further investigation is also needed to understand whether similar trends can be detected in other EPT taxa through both their presence and their numbers across the Ōtūkaikino River catchment, and whether this is associated with any changes in water quality and habitat.

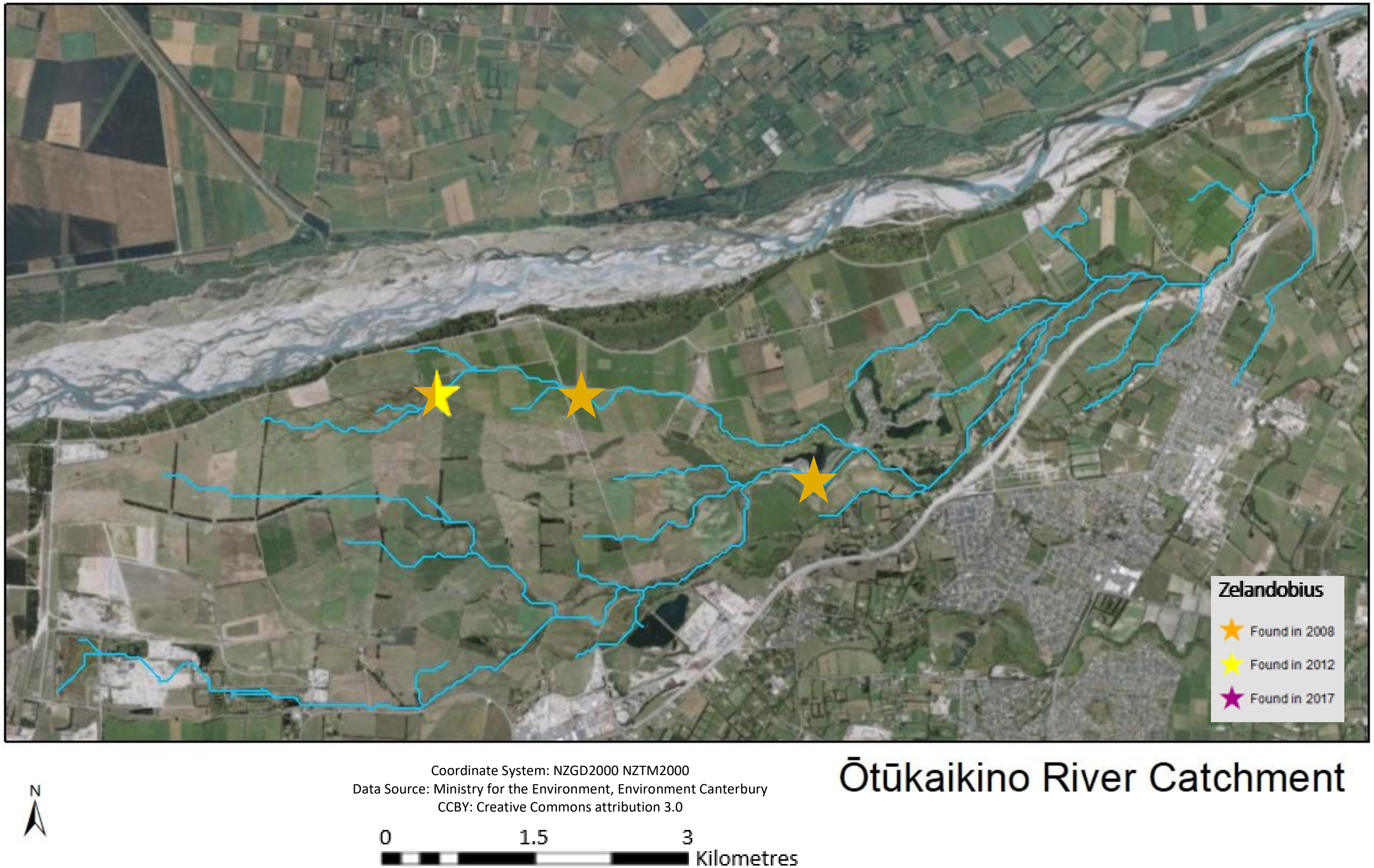

Figure 1.6: Christchurch City Council macroinvertebrate sampling sites where *Zelandobius* stoneflies were noted in 2008, 2012 and 2017 (Boffa Miskell Limited, 2017; EOS Ecology, 2012).

1.3 Research Aim and Objectives

The primary aim of this research was to determine potential sources of pollution and habitat limitation in the Ōtūkaikino River catchment related to the potential decline of a sensitive macroinvertebrate taxa.

There were four main objectives to this research:

- To design and undertake a 12-month monitoring programme for the Ōtūkaikino River catchment including water quality, habitat and ecological indicators
- To quantify temporal and spatial changes in the state of the Ōtūkaikino River, including macroinvertebrate community health
- To identify potential sources of pollution and habitat limitation related to the potential decline of a sensitive macroinvertebrate taxa
- To identify and propose practical solutions for remediation

2 Methodology

This chapter describes the methods required for the three main practical components of this study: visual assessment of the catchment's character, implementation of a surface water and habitat monitoring programme, and the identification of initial areas of concern.

2.1 Catchment Walks

The first component of this research project involved the identification of specific sources of pollution to the catchment. A visual assessment of the catchment was undertaken to indicate areas potentially of concern. This informed the subsequent water monitoring programme.

The visual assessment involved "Catchment Walks" and intensive observations and recording of catchment characteristics. Current sources of potential pollution, such as eroding banks and stormwater pipes, were noted and photographed. General changes in habitat characteristics across the catchment were also recorded.

This information was used to support conclusions drawn from later data analysis and interpretation. Additional analysis of current and historic aerial imagery from the Land Information New Zealand (LINZ) database was conducted to provide another perspective on current pollution sources and to identify potential historic pollution sources in the catchment.

2.2 Design of the Monitoring Programme

There is growing acknowledgement for the importance of reliable and comprehensive approaches to assessing the state and trends of a waterbody, particularly through their integration into a single holistic water monitoring programme. Though there is not one monitoring design that is suitable for all contexts, there are a handful of key components generally accepted to be important when planning a water monitoring program, including those proposed by Chapman (1996).

The desired objectives of the programme should be the first consideration, as they inform the type, intensity and frequency of sampling most applicable for the programme. Preliminary surveys might follow to test both logistical elements and any previous assumptions of the waterway (Bartram & Ballance, 1996). Out of this, an initial monitoring programme can be created. This should involve determining monitoring parameters most suitable to the context and objective, an appropriate frequency of sampling (informed by observed spatial and temporal water variation) and how to regularly evaluate the design. The water monitoring programme can then be put into practice, utilising protocols for sample collection and analysis such as those outlined in Bartram and Ballance (1996). After data collection, several areas still need to be considered: quality control, data storage and appropriate data interpretation and communication, to name a few.

For this project, a key aim was to develop a 12-month monitoring programme for the Ōtūkaikino River catchment. This involved 10 sites across the catchment and a variety of parameters to examine the current state of the water and sediment quality, habitat characteristics and macroinvertebrate community composition.

2.2.1 Site Selection

Water, sediment and habitat quality monitoring was undertaken to help determine potential sources of pollution to the catchment, supplemented by available data from the Christchurch City Council. Seven sites were identified across the Ōtūkaikino catchment as necessary for evaluating the current water quality in the catchment. These were all visited prior to the start of sampling. Site locations are described in Table 2.1 and their locations shown in Figure 2.1, with site photos shown in Figure 2.2. Criteria for site selection included:

- Site accessibility;
- The presence of any historical data; and
- Position in the catchment.

The seven sites used were taken from the five yearly invertebrate monitoring in the catchment (Boffa Miskell Limited, 2017). For this thesis, it was important to examine long-term temporal trends in invertebrate communities, hence the alignment of monitoring with past invertebrate sites instead of solely past water quality monitoring sites. Two of these sites (Sites 2 and 4) were also close to those regularly monitored for water quality by the Christchurch City Council and Environment Canterbury. These seven sites provided good coverage of the upper and middle part of the catchment, where more sensitive taxa have been found in the past (Boffa Miskell Limited, 2017). The lower catchment was not a key target for this monitoring programme. This was due to available time and resources, alongside the lower chances of more sensitive taxa like stoneflies being present in these reaches. However, ongoing monitoring and research is recommended due to urban stormwater inputs and future urban development plans.

Two sites from previous macroinvertebrate monitoring were excluded. Ōtūkaikino River at Dickey's Road was not included in routine sampling, as water quality at the location was already sampled monthly for State of the Environment monitoring and invertebrate health sampled yearly. Kaikanui Creek was also not included due to the lack of an additional site downstream of this tributary. A supporting site downstream would have been needed for the interpretation of water quality in Kaikanui Creek, particularly its contribution to the water quality of the main branch of the Ōtūkaikino River.

Three additional sites (Sites 8, 9 and 10) were used for supplementary macroinvertebrate collection, also shown in Table 2.1 and Figure 2.1. Their locations were chosen to provide a greater spatial intensity than the seven water quality sites, placed at relatively regular intervals in the upper and middle catchment.

For each site, a reach of approximately 20 metres was selected, containing at least one riffle and run, for invertebrate sampling. Water sampling was undertaken at a single site each time, within the 20 m reach, to ensure consistency.

Table 2.1: Water quality and invertebrate site locations.

Site Identifier	Site Description	NZTM East	NZTM North
Site 1	Ōtūkaikino Creek at McLeans Island Road	1563218.208	5186908.366
Site 2	Ōtūkaikino Creek at Scout Camp	1565846.780	5188138.762
Site 3	Ōtūkaikino Creek at Clearwater Resort	1567024.866	5188932.045
Site 4	Ōtūkaikino River main stem at the Groynes	1568568.775	5189683.742
Site 5	Waimakariri South Branch at Clearwater Resort	1566921.857	5189395.770
Site 6	Waimakariri South Branch off Coutts Island Rd, north of Scout camp	1564782.068	5189758.538
Site 7	Waimakariri South Branch off Coutts Island Rd, north of McLeans Island	1563505.641	5189634.692
Site 8	Ōtūkaikino Creek within QEII covenant	1565163.015	5188010.874
Site 9	Ōtūkaikino River main stem upstream of Kaikanui Creek	1569101.003	5190526.700
Site 10	Waimakariri South Branch headwaters	1563120.718	5189410.476

Figure 2.1: Water quality and invertebrate monitoring sites in the Ōtūkaikino catchment.

Figure 2.2: Photographs of the 10 sites monitored during 2019 and 2020. Sites 8, 9 and 10 were not included in the sediment sampling or seasonal water quality sampling.

2.2.2 Parameters and Frequency

For each reach, samples and measurements of a variety of chemical, biological and physical parameters were taken. The parameters were chosen based on previous sampling, as well as those that provided an overview of water and sediment quality in the catchment. These included nutrients, trace elements, total organic carbon (TOC), *E. coli*, stream flow, grain size, total suspended solids (TSS) and turbidity, alongside measurements of temperature, conductivity, dissolved oxygen and pH. Parameters are further detailed in Table 2.3. Observations were also taken of substrate, land use, macrophytes, riparian vegetation composition, fine sediment, water clarity/cover, weather and canopy cover (outlined in Appendix 1). Major ion concentrations were only measured in the second round of sampling, with conductivity used as a proxy thereafter. Major ions were assumed to remain fairly constant across rounds of sampling if the conductivity stayed consistent. Major ion concentrations were measured to gauge the chemical character of the waters, and for use in trace element modelling, if this was required.

Routine water quality sampling occurred approximately quarterly during this study: 22 May, 15 July, 10 October and 9 January. This provided an overall idea of the catchment over the course of a year, capturing any seasonal variation in quality while not exceeding resource and time limitations. A single round of sediment samples was taken from each site as deposited sediment quality tends to show little variation in the short term (Chapman, 1996). Major ions were sampled once, in July 2019, and other trace elements were sampled twice, in May and July 2019.

Macroinvertebrate samples were taken to assess the waterway community's response to current and historical water quality. Sites were sampled once, in July 2019. This spatially intensive, rather than temporally extensive, monitoring programme allowed the inclusion of new sites observed to be potential refugia for sensitive species. This meant that any stoneflies or other similarly sensitive taxa still present in the catchment were more likely to be observed.

2.2.3 Additional Data Sources for Macroinvertebrates and Water Quality

Alongside the data collected during the 12-month monitoring programme, several other data sources were used for water quality and macroinvertebrate community changes. For water quality, two previous surveys of the catchment by the Christchurch City Council were used: Margetts and Marshall (2018), and Marshall and Noakes (2019). For macroinvertebrates, the three previous catchment surveys were used: EOS Ecology (2008b), EOS Ecology (2012), and Boffa Miskell Limited (2017). Monthly data from Environment Canterbury's State of the Environment monitoring was not included. This is because the three sites located in the Ōtūkaikino River catchment that are included in this programme did not align with the locations of invertebrate sampling in 2019. In addition, they provided little additional information to the Christchurch City Council monitoring programmes. Data from the CREAS monitoring programme previously undertaken in the Ōtūkaikino River catchment (EOS Ecology, 2008a) was not available during this study.

2.2.4 Sample Collection and Treatment

For each reach, a range of habitat, riparian vegetation and land use were recorded in the field sheet shown in Appendix 1. *In situ* measurements were taken of dissolved oxygen, temperature, conductivity and pH using a recently calibrated HACH 40d multi meter.

Visual estimates of percentage cover of different substrate size classes were taken across the 20 m sample reach to determine substrate size and composition. These size classes followed those outlined in Wentworth (1922) and modified from Harding et al. (2009) (Table 2.2).

Table 2.2: Size classes used for percentage cover of substrate size classes.

Size	Aggregate name
>=4000 mm	bedrock/artificial hard surfaces
256<4000 mm	boulders
128<256 mm	large cobbles
64<128 mm	small cobbles
16<64 mm	pebbles
2<16 mm	gravels
<2 mm	silt/sand

Visual estimates of percentage cover were taken for macrophytes (emergent and submergent), periphyton, fine sediment, and riparian vegetation. These were determined by visual percentage cover over a 20 m sample reach at each site.

Additional water samples were collected for subsequent lab analysis. Nutrient samples were filtered in the field with a 0.45 µm membrane filter, then immediately chilled for transport and later frozen at the laboratory. Two samples for trace elements (including for major cations) were collected; one was filtered through a 0.45 µm membrane filter (for "dissolved" concentrations) and one remained unfiltered (for "acid soluble" concentrations). Both samples were acidified to 0.2% concentrated HNO_3, then chilled in a fridge while awaiting analysis. Anion samples were taken from filtered water samples. TOC samples were stored in amber glass containers while awaiting analysis to reduce light penetration. Water samples were also taken for faecal contamination and turbidity.

Sediment from the top 3 cm (approximately) of the bed sediment was collected by scraping a plastic spoon along the stream bed and transferring the wet sediment to the sample container. To provide enough material for trace element analysis, areas of fine sediment were targeted at each sample site during sediment collection.

Two sediment samples were taken from Site 6, one at the margin and one midstream. The texture and smell of the marginal sediment was very different to the instream sediment, so it was thought that this could serve as a potential comparison between local sediment and sediment carried from further upstream.

Six invertebrate samples, each over an area approximately 0.1 m^2 and across a range of microhabitats, were collected from each sample location with a kicknet (500 µm mesh), before being combined into a single sample. Sampling protocols followed Stark et al. (2001), specifically the C1 (hard-bottomed, semi-quantitative) protocols, to provide temporal consistency with previous monitoring. Invertebrate samples were preserved in 70% ethanol.

Stream depth and velocity was taken using a Global Water flow probe and a cross section created to give estimates of stream flow, following the mid-section method. Wetted width, as well as average and maximum depth, was taken from the stream flow transect. Macrophyte cover was consistently high at Site 7 (Waimakariri South Branch), so stream flow was estimated from the volume and speed of water flowing through two pipes located just downstream of the sampling site. As normal flow estimations were also not possible at Site 4 (Ōtūkaikino River main stem) due to safety concerns, flow was estimated from a cross-section of the river measured at low flow in January 2020, and the water level and velocity measured on every visit at a set location. The change in water level at this location was used to provide a rough estimate of stream flow for the first three site visits.

2.2.5 Analytical Methods

Table 2.3 outlines the parameters analysed during water quality and sediment sampling, including their detection limits.

Table 2.3: Parameters analysed during water quality and sediment sampling, including their detection limits. Major ions (Br, Ca, Cl, DIC, F, K, Mg, Na, NO_3, SO_4 and PO_4) were determined on samples taken during July 2019. All other trace elements were determined on samples taken during May and July 2019.

Parameter		Units of measurement	Detection Limits
Conductivity		µS /cm	0.01
Total organic carbon		mg/L	1
Dissolved oxygen (DO)		mg/L and % saturation	0.01 mg/L, 0.1 %
***E. coli* and faecal coliforms**		CFU/100mL	33
Stream flow		m^3/s	
Nitrate-nitrogen		mg/L	0.02
Ammoniacal nitrogen		mg/L	0.01
pH			1
Dissolved reactive phosphorus (DRP)		mg/L	0.002
Water temperature		°C	-10
Total suspended solids (TSS)		mg/L	1
Turbidity		NTU	0.5
Grain size profile		%	
Trace elements (water)	Al	µg/L	1
	As	µg/L	5
	B	µg/L	0.07
	Br	mg/L	0.02
	Ca	mg/L	0.00001
	Cl	mg/L	0.012
	Cd	µg/L	0.3
	Co	µg/L	0.5
	Cr	µg/L	0.5
	Cu	µg/L	0.6
	DIC as HCO_3	mg/L	5.0
	Fe	µg/L	0.4
	F	mg/L	0.003
	K	mg/L	0.0005
	Li	µg/L	0.1
	Mg	mg/L	0.00001
	Mn	µg/L	0.05
	Mo	µg/L	0.8
	Na	mg/L	0.0002
	Ni	µg/L	1.3
	P	µg/L	7
	Pb	µg/L	3
	S	µg/L	9
	SO_4	mg/L	0.05
	V	µg/L	0.4

	Zn	µg/L	0.3
Trace elements (sediment)	Al	wt%	0.00001
	As	mg/kg	0.53
	B	mg/kg	0.007
	Cd	mg/kg	0.03
	Co	mg/kg	0.05
	Cr	mg/kg	0.05
	Cu	mg/kg	0.06
	Fe	wt%	0.000004
	Mn	mg/kg	0.005
	Mo	mg/kg	0.08
	Ni	mg/kg	0.14
	P	mg/kg	0.74
	Pb	mg/kg	0.32
	S	mg/kg	0.95
	V	mg/kg	0.04
	Zn	mg/kg	0.03

Sediment samples were dried in a drying oven at 30 ^{0}C for one week, before being sieved using a Sefar PET 1500 77/195-55 PW screen printing mesh (pore size 67 µm) to extract some of the finest fraction. For trace elements and phosphorus, 100 mg of the sieved sediment was digested in 10 mL boiling concentrated nitric acid until almost dry. 45 mL of 0.1 N HNO_3 was added and samples were heated until the volume was less than 10 mL. This was then diluted to 10 mL with 0.1 N HNO_3 and then filtered through a 0.45 µm filter. The samples were submitted to the Lincoln University Analytical Services for trace element analysis by Suppressed Ion Chromatography. Trace element results were later multiplied by their dilution factor (mass extract/mass sediment) to convert the concentration in the digests into the concentrations in the sediment.

Trace elements taken from water samples, including cations (Ca, K, Mg, Na) and anions (F, Br, Cl, SO_4), were submitted to Lincoln University Analytical Services for analysis by Suppressed Ion Chromatography. Samples for TOC were also submitted to Lincoln University Analytical Services for further analysis by TIC/TC on a Vario TOC Cube.

For *E. coli* and faecal coliforms, three replicates of 1 mL water samples were plated on 3M Petrifilm disposable plates. These were incubated for 24 to 48 hours before counting. The average count was taken from each sample site and converted to CFU/100 mL (Eaton et al., 2005).

Dissolved inorganic carbon (DIC) was analysed using an IRGA (infrared gas analyser) in the Waterways Laboratory at Lincoln University. The water sample was acidified with 0.2 mL concentrated phosphoric acid, under a nitrogen headspace, to convert all DIC to CO_2. The CO_2 was then measured by IRGA. This gave a measurement of the DIC, which could be used to calculate HCO_3.

Nitrate-nitrogen (NO_3-N) analysis used the spongy cadmium method (Mackereth et al., 1979), with the absorbance measured at 543 nm. Ammoniacal-N (NH_4-N) concentrations were analysed using the phenate method (Eaton et al., 2005). After one hour, absorbance was read at 640 nm. Dissolved reactive phosphorus (DRP) was determined using the ascorbic acid method (Eaton et al., 2005), with absorbance measured at 880 nm. Absorbances for all nutrients were read on a DR3900 UV/VIS Spectrophotometer.

Total Suspended Solids (TSS) used a method adapted from Eaton et al. (2005). 500 mL of sample was filtered under vacuum using a mixed cellulose ester (MCE) filter. The filter was then dried in a desiccator until the weight stabilised. TSS was calculated from the difference between initial and final weight of the filter. Turbidity NTU was measured colourimetrically as light absorbance using an Orion AQ4500 meter.

For grain size profiles, the dried sediment was sieved with seven mesh sizes and the difference in weight calculated as percentages. The mesh sizes are shown below in Table 2.4 and follow the size classes outlined by Wentworth (1922). Sodium hexametaphosphate was used as a dispersant.

Table 2.4: Size classes used for grain size profiles.

Size	Aggregate name
>=2 mm	Gravel/pebble
1<2 mm	Very coarse sand
0.5<1 mm	Coarse sand
0.25<0.5 mm	Medium sand
0.125<0.25 mm	Fine sand
0.063<0.125 mm	Very fine sand
<0.063 mm	Silt/clay

Invertebrate samples were identified in the lab to a taxonomic level using the methods of Winterbourn et al. (2006). Taxonomic resolution was to genera where possible. P3 protocols (full count with subsampling option) were used in accordance with Stark et al. (2001), though subsamples were not completed for any site.

2.2.6 Data Analysis

A variety of local, regional and national guidelines were used to interpret the results of observed site characteristics and water quality samples. Guidelines from the Land and Water Regional Plan (LWRP) (Environment Canterbury, 2017) were used for macrophyte cover, temperature, pH, turbidity and nutrients. The Waimakariri River Regional Plan (WRRP) (Environment Canterbury, 2011) contained local guidelines for dissolved oxygen. ANZECC (2000) guidelines were used for trace element interpretation. ANZECC (2000) provides a range of trigger values for trace elements at different levels of ecosystem protection. For example, a 95% trigger value gives the level of contamination at which 95% of the freshwater ecosystem was estimated to be unaffected. Quality control was undertaken on the analysis data for major ions. The percentage difference in ion balance was less than 5% at all sites, with 5 of the 7 sites recording less than 2% difference in ion balance.

ANZECC (2000) Interim Sediment Quality Guidelines (ISQG) were used for the interpretation of sediment trace element concentrations. Contaminant concentrations below the ISQG-low value were unlikely to have harmful biological effects. For contaminant concentrations between the ISQG-low and the ISQG-high values, sensitive taxa are affected, while exceedances of the ISQG-high values were estimated to impact a range of biota. ANZECC (2000) was also used for dissolved oxygen guidelines. Guidelines by the Ministry for the Environment and Ministry of Health (2003) gave the recreational standards for *E. coli*.

A substrate index (SI) was calculated from the visual estimates of percentage cover of different substrate size classes. The SI was determined from a formula adapted by Harding et al. (2009):

$$SI = (0.03 \times \%silt/sand) + (0.04 \times \%gravel) + (0.05 \times \%pebble) + (0.06 \times (\%small\ cobble + \%large\ cobble)) + (0.07 \times \%boulder).$$

This gave a scale of SI values from 3 to 7, with lower SI values reflecting a higher proportion of finer substrate. This scale is unitless.

A range of macroinvertebrate indices were calculated. EPT and % EPT show the number of *Ephemeroptera* (mayfly), *Plecoptera* (stonefly) and *Trichoptera* (caddisfly) taxa. Macroinvertebrate Community Index (MCI) and Semi-Quantitative Macroinvertebrate Community Index (SQMCI) were both used (Stark & Maxted, 2007). These indices give each taxa a score depending on their sensitivity to contaminants, ranging from 1 (very tolerant) to 10 (very sensitive). An average score is then calculated, with a higher value typically indicating a healthier waterway. An explanation of how to interpret the results of MCI and SQMCI indices is shown in Table 2.5. MCI uses presence/absence data, while SQMCI uses coded abundances with five classes: Rare, Common, Abundant, Very Abundant, and Very Very Abundant. Total abundance and number of taxa were also recorded.

Table 2.5: MCI and SQMCI score interpretation for hard-bottomed and soft-bottomed streams (Adapted from Stark & Maxted, 2007).

Stream health	Water quality descriptions	MCI	SQMCI
Excellent	Clean water	> 119	> 5.99
Good	Doubtful quality/possible mild enrichment	100 - 119	5.00 - 5.99
Fair	Probable moderate enrichment	80 - 99	4.00 - 4.99
Poor	Probable severe enrichment	< 80	< 4.00

Analysis of variance (ANOVA) was used to determine whether there was statistically significant spatial and temporal variation in MCI scores. For spatial variation, MCI scores at each site over the four time steps (2008, 2012, 2017 and 2019) were aggregated and ANOVA performed, with the results displayed in ggplot (Oksanen et al., 2019). For temporal variation, MCI scores from each time step were instead aggregated.

Three sites were not included in the spatial and temporal analysis: Sites 8, 9 and 10. Sites 8 and 10 had not been sampled before 2019, while Site 9 had shifted location in 2012 and again in 2019. The exclusion of these sites provided more temporal consistency in the analysis.

Invertebrate sampling in 2008 and 2012 (EOS Ecology, 2008b, 2012) involved three replicate samples across a total of 1.8 m^2, compared to a single sample over 0.6 m^2 in 2017 (Boffa Miskell Limited, 2017) and 2019. To account for this, the average of the three replicates in 2008 and 2012 was used for each site. However, the greater area in the first two macroinvertebrate surveys potentially increased the probability of finding rare taxa, based on the idea of species-area curves (Boffa Miskell Limited, 2017).

Macroinvertebrate community data from 2008, 2012, 2017 and 2019 was classified into 6 taxonomic groups: *Crustacea, Diptera, Ephemeroptera, Mollusca* and *Tricoptera*, with the remainder classed as Other. Pie charts were used to show spatial variation in macroinvertebrate community composition in 2019. Stacked bar charts were used to show temporal changes in macroinvertebrate community composition between 2008 and 2019.

Non-Metric Multidimensional Scaling (NMDS) using Bray–Curtis dissimilarity was undertaken with the “vegan” package in R (Oksanen et al., 2019). NDMS was used to provide an indication of macroinvertebrate community similarity across time by summarising the number of taxa and the abundance of each in two-dimensional space. NMDS ordination was conducted twice to compare 2017 and 2019 macroinvertebrate community compositions as well as all four timesteps (2008, 2012, 2017, and 2019).

NMDS model goodness of fit was shown with the stress value. Low stress was thought to indicate a fair representation of the data, with a score of 0 suggesting a perfect model fit (Quinn & Keough, 2002).

Permutational multivariate analysis of variance (PERMANOVA), using the “vegan” package function “adonis”, tested whether the centroids of the distance matrices differed significantly between years. Analysis of similarity (ANOSIM), also in the “vegan” package, tested whether differences in community composition were greater between years than within each year. If both PERMANOVA and ANOSIM indicated a statistically significant difference, similarity percentage (SIMPER) analysis was performed using Bray–Curtis dissimilarity to indicate what taxa were contributing the most to the temporal differences.

2.3 Land Use and Population Change

2.3.1 Data Sources

Land use change, proposed by Boffa Miskell Limited (2017) to be a potential cause of stonefly disappearance, was assessed using historical satellite imagery and analysed using ArcGIS software. This utilised data available through the Land Cover Database, a project led by Landcare Research (formerly by Terralink). Population data was taken from Statistics New Zealand. River lines were adapted from the New Zealand River Environmental Classification database, while catchment boundaries were provided by Environment Canterbury.

2.3.2 Processing and Analysis

Population analysis of the Ōtūkaikino catchment involved five sets of census data: 1996, 2001, 2006, 2013 and 2018. Meshblock population data was cut to the Ōtūkaikino catchment boundary and the five census counts joined to the 2018 meshblock boundaries. Population counts were discounted from meshblocks that had more than half their area outside the catchment boundaries. This was done in order to help reduce overestimation of population numbers. This did result in potential inaccuracies in exact counts, but the available resolution and boundaries of the census data made a higher degree of accuracy difficult. Several meshblocks were also merged due to changes in meshblock boundaries between the 1996 census and the 2018 census. Absolute population numbers were classified and displayed for each census year.

At the time this research was undertaken, five different time steps were available through the Land Cover Database: summer 1996/97, summer 2001/02, summer 2008/09, summer 2012/13, and summer 2018/19. Historical aerial imagery was used to verify classifications.

The Land Cover Database contains 33 mainland land cover classes that have either been classified or manually digitised. To allow for more meaningful analysis, these were merged at three scales into a land cover classification hierarchy. For consistency, this followed the hierarchy previously developed for land cover reporting at regional and national scales in New Zealand: 33 detailed classes, 12 medium classes, and six broad classes. This hierarchy is shown in Table 2.6, with descriptions of the classes included in Appendix 3.

Table 2.6: Land cover classification hierarchy (LAWA, 2017).

Broad Classes	Medium Classes	Detailed Classes
Urban/bare/lightly-vegetated surfaces	Artificial bare surfaces	Transport infrastructure
		Surface mine or dump
	Natural bare/lightly-vegetated surfaces	Sand or gravel
		Landslide
		Gravel or rock
		Permanent snow and ice
		Alpine grass/herbfield
	Urban area	Built-up area (settlement)
		Urban parkland/open space
Cropland	Cropping/horticulture	Short-rotation cropland
		Orchards, vineyards or other perennial crops
Forest	Exotic forest	Forest - harvested
		Exotic forest
		Deciduous hardwoods
	Indigenous forest	Indigenous forest
		Broadleaved indigenous hardwoods
Grassland/other herbaceous vegetation	Exotic grassland	Depleted grassland
		High producing exotic grassland
		Low producing grassland
	Other herbaceous vegetation	Herbaceous freshwater vegetation
		Flaxland
		Herbaceous saline vegetation
	Tussock grassland	Tall tussock grassland
Scrub/shrubland	Exotic scrub/shrubland	Gorse and/or Broom
		Mixed exotic shrubland
	Indigenous scrub/shrubland	Manuka and/or Kanuka
		Matagouri or Grey scrub
		Fernland
		Sub-alpine shrubland
		Mangrove
Water bodies	Water bodies	Lake or pond
		River
		Estuarine open water

3 Results

All field data is presented in Appendix 2, with key parameters of interest graphed in this chapter to aid interpretation. Figure 2.1 shows site locality in the catchment. The site names within figures and tables in this chapter are marked according to their location within the catchment: "-OC" for sites within the Ōtūkaikino Creek (the main southern tributary), "-WSB" for sites within the Waimakariri South Branch (the main northern tributary), and "-OR" for sites within the main stem of the Ōtūkaikino River.

3.1 Site Characteristics

3.1.1 Vegetation and Periphyton

Percentage riparian cover over the sample reach varied greatly between the sites sampled in the Ōtūkaikino River catchment (Table 3.1). The lowest riparian cover was found in several of the headwater sites, with 5% riparian cover at Site 1 (Ōtūkaikino Creek), as well as at Sites 6 and 10 (Waimakariri South Branch). Riparian cover increased further downstream along both main tributaries, reaching 90% at Site 9 and 100% at Site 4 on the Ōtūkaikino River. Similarly, average canopy cover was generally lower in the headwater sites, with 0% at Sites 1, 6 and 10, while reaching an average of 86% at Site 5, in the mid reaches of the catchment. Site 7, in the headwaters of the Waimakariri South Branch, had high riparian cover and average canopy cover compared to other headwater sites, at 50% and 60% respectively.

At the time this research was undertaken, the headwaters of the Waimakariri River South Branch were mostly grassed, with some of the ephemeral reaches unfenced (Figure 3.1). There was some pugging of soil around ephemeral reaches and unplanted banks. A few sections along this reach, such as around Site 7, were buffered by exotic trees and gorse. This transitioned into alternating grassed buffers with sections of carex and the occasional flax or toitoi. Part of this area is slated for more native planting over the next few years, with some of this visibly underway as of 2020.

This reach transitioned into the manicured golf course around Site 5. While one bank of the river within the golf course was thick with vegetation, the bank bordering the golf course was closely mown to the edge, with occasional native trees and carexes. During the initial "Catchment Walk" in April

2019, the area below Site 5 had undergone bank clearance. This left bare earth exposed along a 200 m reach for most of the study period.

Like the Waimakariri River South Branch, most of the upper reaches of Ōtūkaikino Creek flowed through pastoral land with grassed margins. Within this section was a 400 m reach of QEII covenant land. This has been extensively planted with native trees, flaxes and the like over the past decade. Work on this is still ongoing.

The Ōtūkaikino Creek reach in immediate vicinity of the scout camp buildings, around Site 2, was composed of manicured lawn, which transitioned into established exotic and native trees. Downstream of here, Isaac Conservation Trust had been planting mixed native vegetation within the riparian buffer zone, though this is more established in some locations than others. This extended to the resort. Native and exotic trees, including willows, were well established on one bank, while the other has young native bushes and trees. Stormwater pipes from the resort apartments were common through here.

Downstream of the confluence of the Waimakariri South Branch and the Ōtūkaikino Creek, the main stem of the Ōtukaikino River was typically thick with native (and some exotic) vegetation as it wound through a reserve. Though the main stem was large at this point, canopy cover was high.

There were several small tributaries in the lower reaches of the Ōtūkaikino River, including Wilsons Drain, which drained a mix of urban, pastoral and reserve land. The Ōtūkaikino River confluence with the Waimakariri River was dominated by established trees.

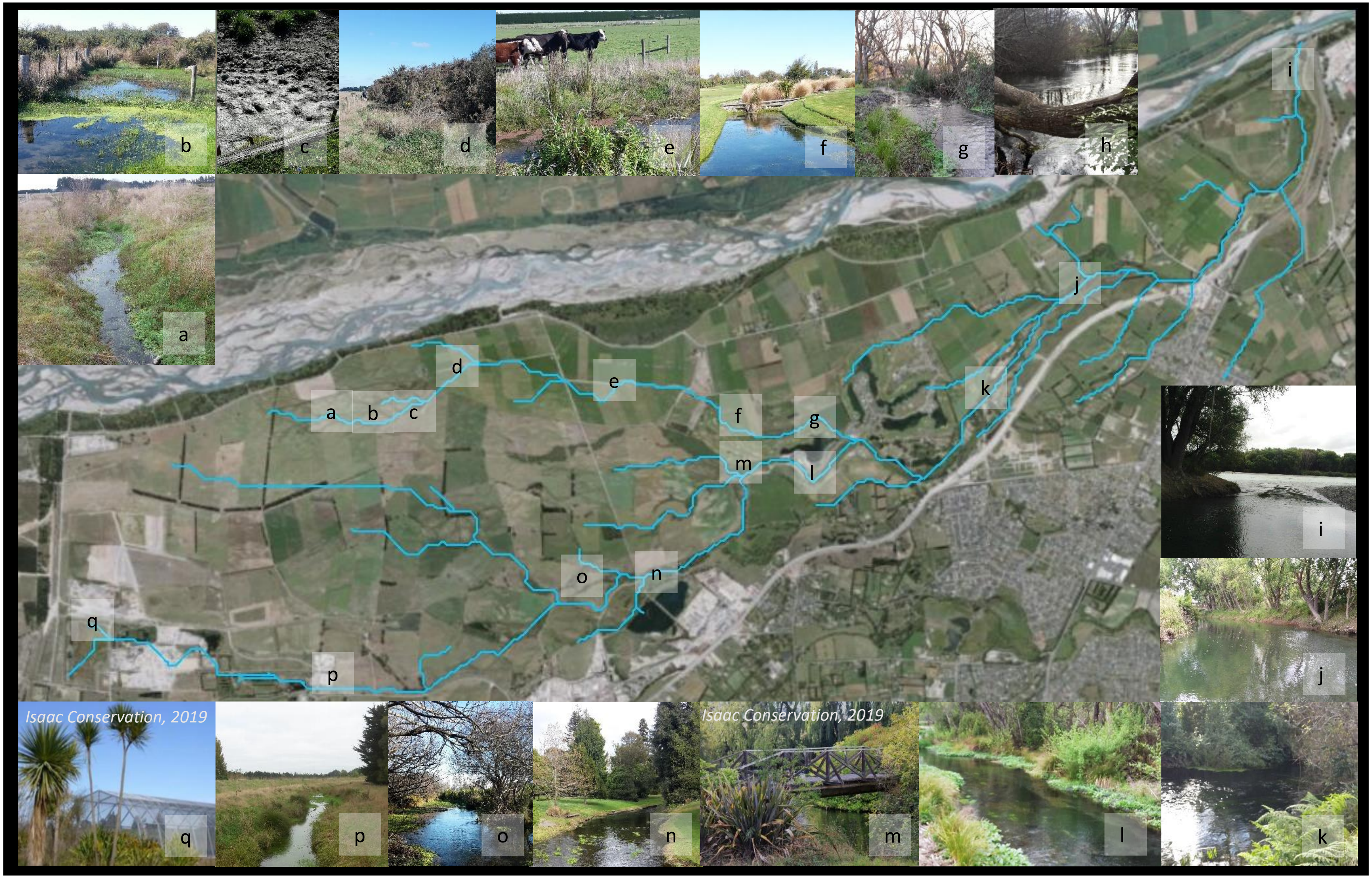

Figure 3.1: General habitat changes across the main branches of the Ōtūkaikino River in 2019.

Emergent and submergent macrophyte cover was generally low at all sites (Figures 3.2 and 3.3). However, Site 7 (Waimakariri South Branch) consistently had high levels of emergent macrophytes on the dates sampled, reaching 95% cover in May 2019. Sites 1 and 8 (Ōtūkaikino Creek) also recorded similarly high levels of emergent macrophytes on some sample dates. High levels of submerged macrophyte cover, compared to other sites monitored in the catchment, was recorded in October for the Ōtūkaikino Creek (Sites 2 and 3), as well as for Site 2 in January. At Site 1 (Ōtūkaikino Creek), as well as Sites 6 and 7 (Waimakariri South Branch), macrophyte cover was mostly composed of a mix of duckweed (*Lemna minor*) and watercress (*Nasturtium officinale*). Various species of pondweed were also present in the catchment.

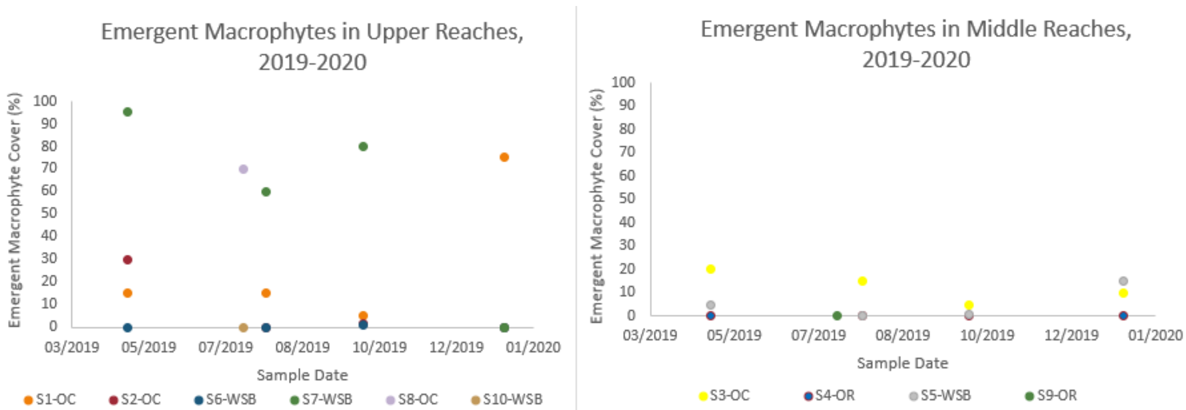

Figure 3.2: Emergent macrophyte cover at 10 sites in the Ōtūkaikino River catchment during 2019 and 2020.

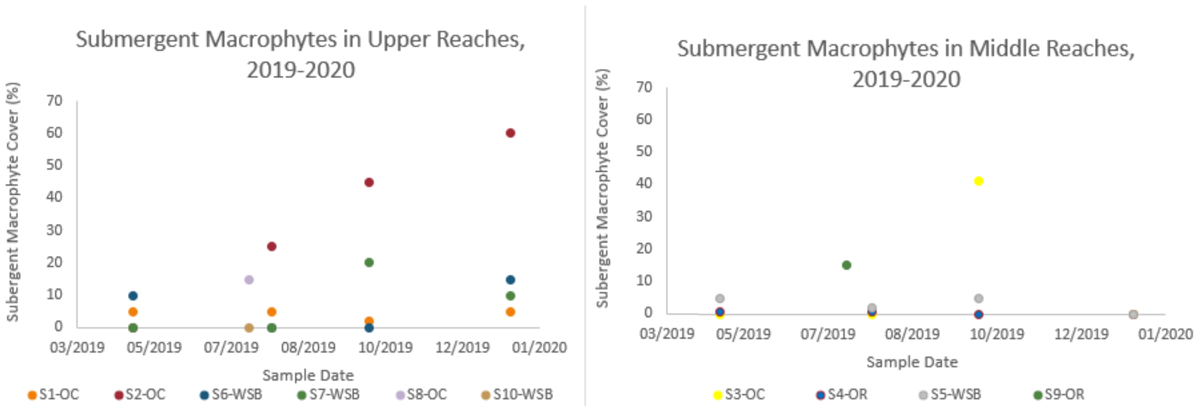

Figure 3.3: Submergent macrophyte cover at 10 sites in the Ōtūkaikino River catchment during 2019 and 2020.

Average periphyton cover differed between sites, from 0% at Site 7 to 70% at Site 10, with these sites both located in the upper Waimakariri South Branch (Figure 3.4). The highest levels were recorded at Site 1, in the upper Ōtūkaikino Creek, with 95% cover in May 2019. Most of the periphyton observed was present as thin films. Filamentous algae were uncommon, though still consistently present at Site 2 (Ōtūkaikino Creek). Mat-forming periphyton was often present at Sites 3 (Ōtūkaikino Creek) and 7 (Waimakariri South Branch), though was also observed at Site 1 in May.

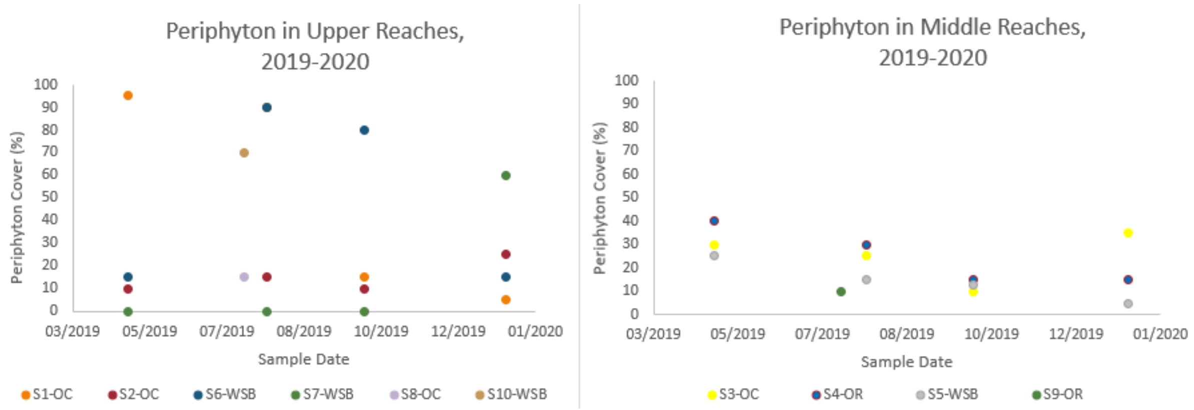

Figure 3.4: Periphyton cover at 10 sites in the Ōtūkaikino River catchment during 2019 and 2020.

3.1.2 Substrate

The average Substrate Index value was relatively similar across the catchment, ranging from 4.54 to 5.26 (Table 3.1). The lowest average substrate index values were at Site 2 (Ōtukaikino Creek), alongside Sites 6 and 7 (Waimakariri South Branch). These sites were dominated by finer substrates like silt and gravel. Sites 3 and 4 had the highest average substrate index values. They tended to have a higher portion of coarser substrates such as cobbles. Similarly, the highest fine sediment cover was generally at Sites 2 and 7 (Figure 3.5). There were no consistent seasonal trends in fine sediment cover. The middle reaches consistently had low levels of fine sediment, while the upper reaches showed more variation.

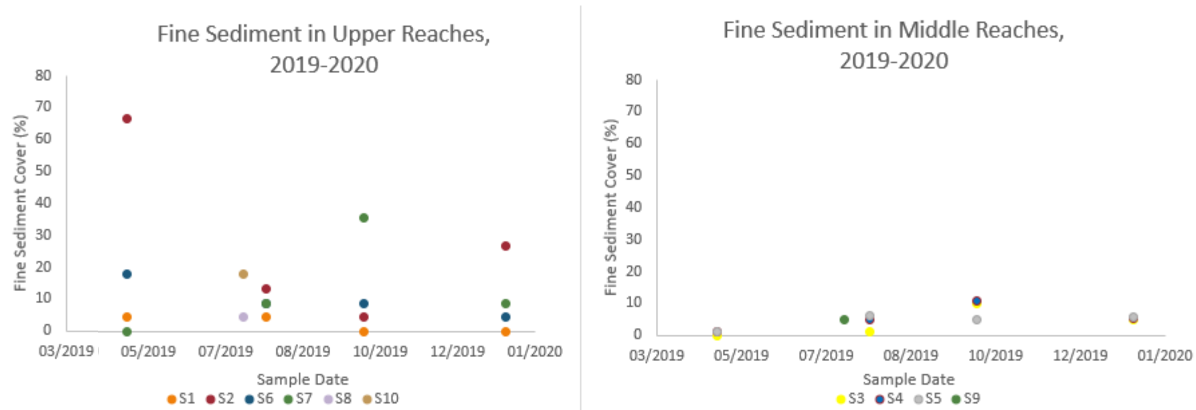

Figure 3.5: Fine sediment cover at 10 sites in the Ōtūkaikino River catchment during 2019 and 2020.

3.1.3 Wetted Width, Water Depth and Flow

Average stream wetted width varied from 1.2 m in the upper Waimakariri South Branch (Site 10) to 14.1 m in the middle of the catchment (Site 4) (Table 3.1). Similarly, both average and maximum water depth increased towards the middle of the catchment, with the deepest water depth recorded at Sites 3 and 4.

Stream flow increased from the headwater sites to the middle of the catchment (Figure 3.6). Sites 1 and 7 (the headwaters of the Ōtūkaikino Creek and Waimakariri South Branch respectively) had very little flow, decreasing to a minimum of 0.02 m^3/s and 0.01 m^3/s in January 2020. In comparison, Site 4, below the convergence of the two main branches, reached a maximum of 1.9 m^3/s in October 2019.

A seasonal pattern in stream flow was evident at most sites. Stream flow tended to increase in in the July samples, as well as the October samples at some sites, and decreased again in the January samples. Sites 3 (Ōtūkaikino Creek) and 4 (Ōtūkaikino River) did not show these seasonal trends. Site 3 increased in the July samples and stayed elevated during the remaining monitoring, while stream flow at Site 4 remained somewhat similar during the sampling events.

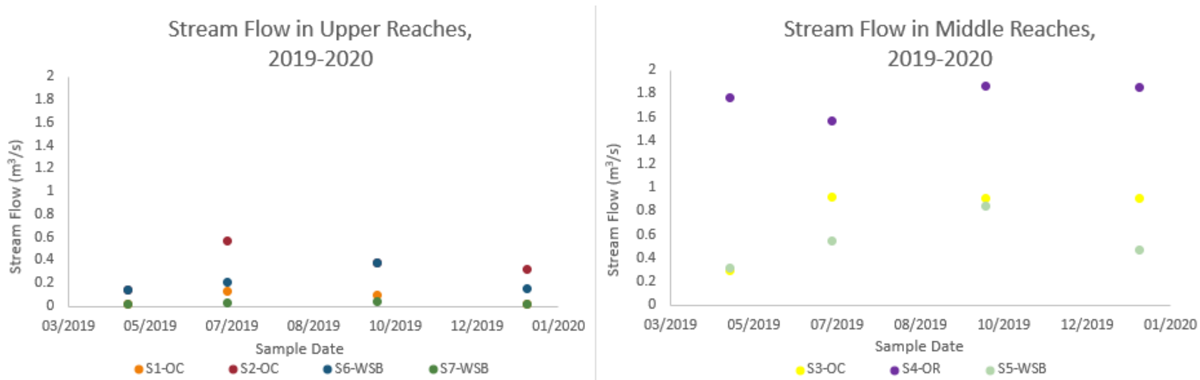

Figure 3.6: Stream flow at 7 sites in the Ōtūkaikino River catchment during 2019 and 2020.

Table 3.1: Physical characteristics of the 10 sampling sites, measured between May 2019 and January 2020. Percentage riparian cover did not change between May and January for any of the sites sampled. The remaining values for Sites 1 – 7 are averages of four samples (n=4). Sites 8-10 were only measured on one occasion in July 2019 (n=1). Water depth and flow were not measured for these three sites.

Site	Riparian cover (%)	Canopy cover (%)	Substrate Index	Wetted width (m)	Water depth (m)	
					Max	Average
S1-OC	5	0	4.98	2.21	0.16	0.11
S2-OC	30	33	4.54	5.83	0.27	0.20
S3-OC	70	42	5.13	4.13	0.76	0.51
S4-OR	100	66	5.26	14.10	0.70	0.45
S5-WSB	55	86	4.82	5.10	0.23	0.18
S6-WSB	5	0	4.75	4.24	0.26	0.22
S7-WSB	50	60	4.63	4.88	0.15	0.13
S8-OC	10	40	5.05	11.00		
S9-OR	90	25	4.65	10.50		
S10-WSB	5	0	4.80	1.20		

3.2 Surface Water Quality

3.2.1 Main Water Quality Parameters

Dissolved oxygen was generally high at all sites and dates sampled in the Ōtūkaikino River catchment (Table 3.2). Average dissolved oxygen ranged from 8.33 mg/L (Site 7) to 10.60 mg/L (Site 6), with these two sites located in the Waimakariri South Branch. Percentage saturation for dissolved oxygen showed similar site variation, with levels ranging from 55.2% (Site 7) to 101.9% (Site 6) (Figure 3.7). The lowest levels were recorded in January 2020, with 55.2% at Site 7 (Waimakariri South Branch) and 57.2% at Site 1 (Ōtūkaikino Creek). The percentage saturation of dissolved oxygen exceeded 100% on three of the five samples at Site 6. For the Ōtūkaikino River catchment, percentage saturation of dissolved oxygen across the middle reaches were similar on the dates sampled. The upper reaches showed more variation, especially in January 2020.

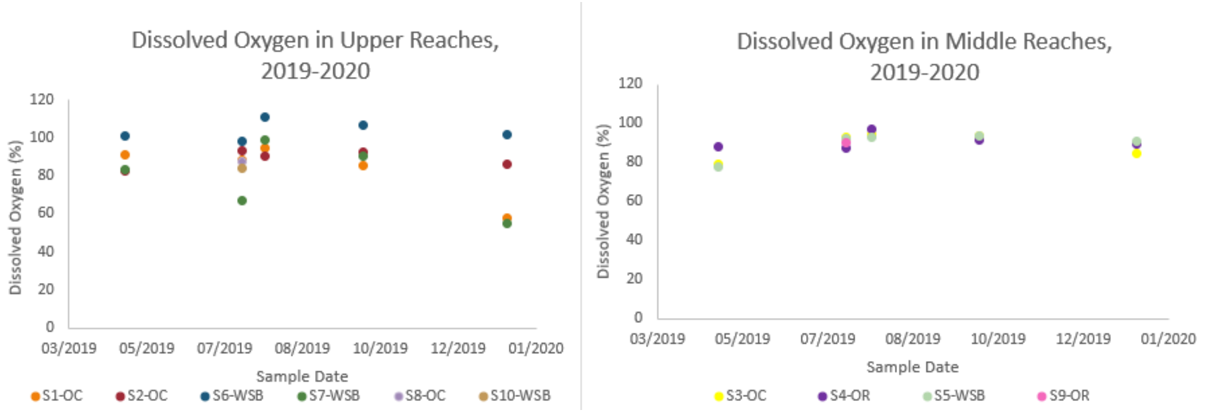

Figure 3.7: Dissolved oxygen at 10 sites in the Ōtūkaikino River catchment during 2019 and 2020.

Though variable, temperature was generally cool across all sites (Figure 3.8). The lowest temperature was recorded in the Waimakariri South Branch headwaters at Site 10 in July (7.6 ^{0}C). The key exception to this was Site 6 in January 2020 (18.4 ^{0}C). Temperature showed strong temporal variation, though this was more pronounced in the upper reaches. Temperature decreased in the July samples before increasing in the October samples to reach maximum temperatures for all sites in the January 2020 samples, aside from Site 7.

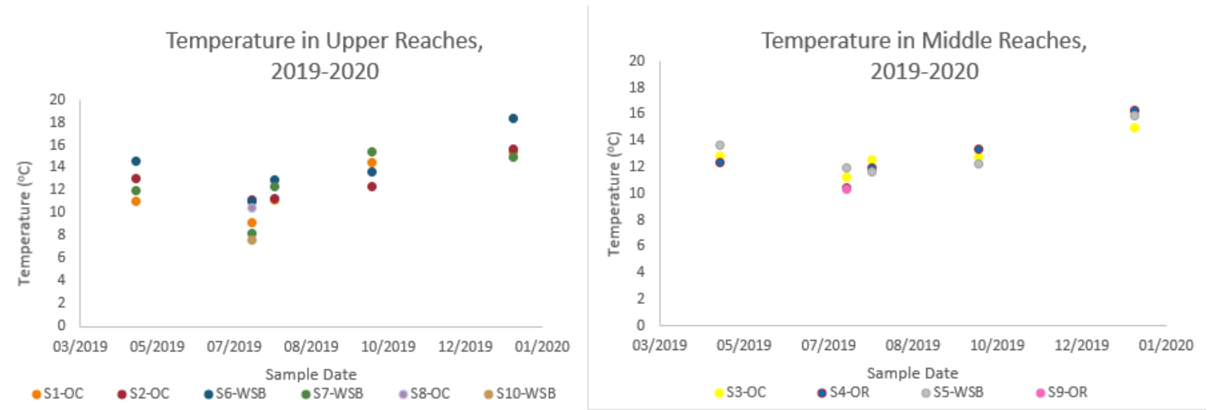

Figure 3.8: Stream temperature at 10 sites in the Ōtūkaikino River catchment during 2019 and 2020.

pH was circa-neutral at all sites across the time period surveyed (Table 3.2). On the days sampled, pH ranged from 6.64 (Site 1, Ōtūkaikino Creek) to 7.77 (Site 6, Waimakariri South Branch). There were no clear spatial or temporal trends recorded.

Similarly, there was limited variation recorded in conductivity for the dates sampled (Table 3.2). The lowest average conductivity of 64.3 µS/cm was recorded at Site 10, in the headwaters of the Waimakariri South Branch. The highest average conductivity recorded was at Site 2 (88.0 µS/cm), in the Ōtūkaikino Creek.

Table 3.2: DO (dissolved oxygen), conductivity and pH of the 10 sampling sites, as measured between May 2019 and January 2020. Sites 8-10 were only measured on one occasion in July 2019, while Sites 1-7 are averages (n=5) for DO and conductivity. pH is presented as a range for Sites 1-7 (n=5).

Site	DO	pH	Conductivity
	mg/l		µS/cm
S1-OC	8.91	6.64-7.29	82.6
S2-OC	9.40	6.84-7.31	88.0
S3-OC	9.34	6.85-6.49	84.5
S4-OR	9.55	6.98-7.42	86.6
S5-WSB	9.38	6.82-7.33	74.0
S6-WSB	10.60	7.21-7.77	73.7
S7-WSB	8.33	6.96-7.51	64.8
S8-OC	9.66	7.25	86.8
S9-OR	10.08	7.21	87.3
S10-WSB	10.02	7.25	64.3

3.2.2 Microbiological Contaminants

Faecal coliforms, an indicator of faecal contamination, varied greatly between sites and between rounds of sampling (Figure 3.9). The highest value, 8450 CFU/100 mL, was recorded at Site 1 (Ōtūkaikino Creek) in January 2020. Site 1 consistently recorded elevated levels of faecal coliforms compared to the rest of the catchment. Site 7, in the upper Waimakariri South Branch, also had high levels of faecal coliforms compared to other sites sampled in the catchment in two of the four samples. Other sites produced a variety of faecal coliform concentrations. Sites 1 and 2 (Ōtūkaikino Creek), alongside Site 6 (Waimakariri South Branch), increased steadily in faecal coliform concentrations throughout the monitored samples. Site 3, 4, and 5, in the middle reaches of the catchment, decreased in the July samples and subsequently increased to reach their maximums in the January 2020 samples.

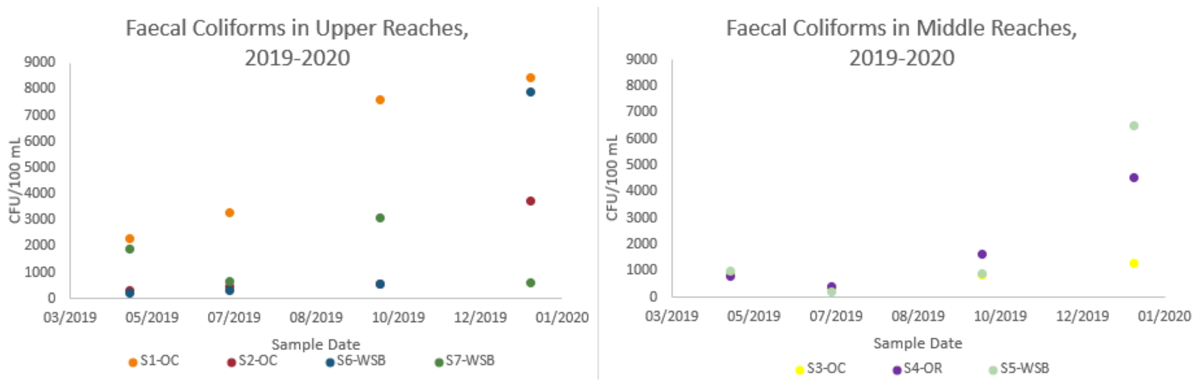

Figure 3.9: Faecal coliform concentrations at 7 sites in the Ōtūkaikino River catchment during 2019 and 2020.

E. coli tended to be at lower levels than faecal coliforms on the dates sampled (Figure 3.10). Though Site 1, in the upper Ōtūkaikino Creek, typically recorded high levels of faecal coliforms, *E. coli* levels were comparatively low. *E. coli* levels did not show any consistent seasonal pattern during this study. Sites 2 and 3 increased in the July sample, decreased in the October sample, and increased again in the January sample. In comparison, Sites 5 and 6, in the Waimakariri South Branch, had no detectable *E. coli* until October, and increased significantly in the January 2020 samples. Site 5 recorded the second highest *E. coli* concentration of this study, reaching 350 CFU/100 mL in the January 2020 sample. The maximum concentration was recorded at Site 6 in the January 2020 sample, with 1400 CFU/100 mL.

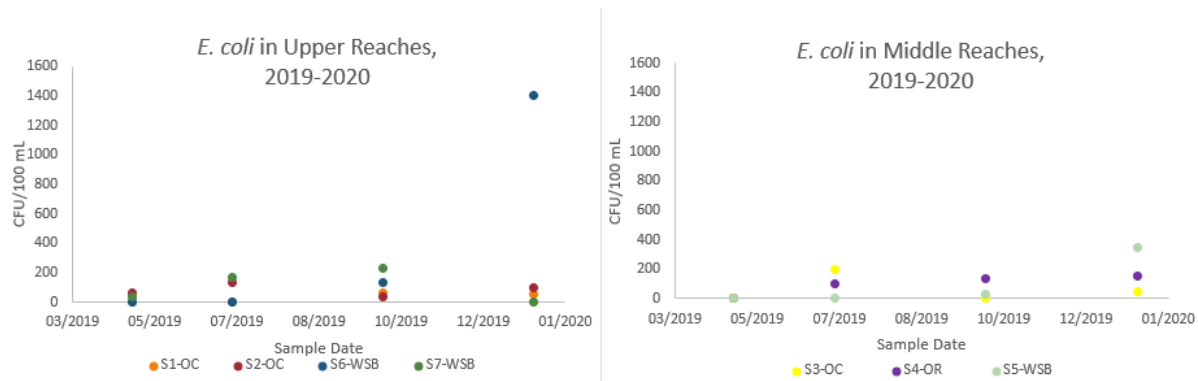

Figure 3.10: *E. coli* concentrations at 7 sites in the Ōtūkaikino River catchment during 2019 and 2020.

3.2.3 Turbidity and Total Suspended Solids

Turbidity was very low across all sites and time steps sampled (Table 3.3). Average turbidity ranged from 0.13 NTU at Sites 2 and 3 in the Ōtūkaikino Creek to 0.37 NTU at Site 6 in the Waimakariri South

Branch. Total suspended solids were similarly low across all sites (Table 3.3). Average total suspended solids varied between 0.38 mg/L (Site 6, in the Waimakariri South Branch) and 1.43 mg/L (Site 5, also in the Waimakariri South Branch) during the dates sampled. No clear spatial or temporal trends were recorded for either parameter.

3.2.4 Nutrients and Total Organic Carbon

Ammoniacal nitrogen was generally similar across all sites (Table 3.3). Average ammoniacal nitrogen ranged from 0.01 mg/L at Sites 4 (Ōtūkaikino River) and 7 (Waimakariri South Branch) to 0.05 mg/L at Sites 2 and 3 (Ōtūkaikino Creek). Average ammoniacal nitrogen was generally higher in the Ōtūkaikino Creek compared to the Waimakariri South Branch. Nitrate-nitrogen levels were typically low. Average values ranged from 0.04 mg/L at Site 7 to 0.30 mg/L at Site 2.

Dissolved reactive phosphorus (DRP) was low across most sites relative to ANZECC (2000) guidelines for lowland streams ecosystem protection (Figure 3.11). However, elevated levels of DRP were recorded at Site 1, in the upper Ōtūkaikino Creek. Average DRP at this site was 0.039 mg/L, which was well above other samples in the catchment during 2019 and 2020. DRP at Site 2, a few kilometres downstream, was consistently recorded at low levels similar to the rest of the catchment.

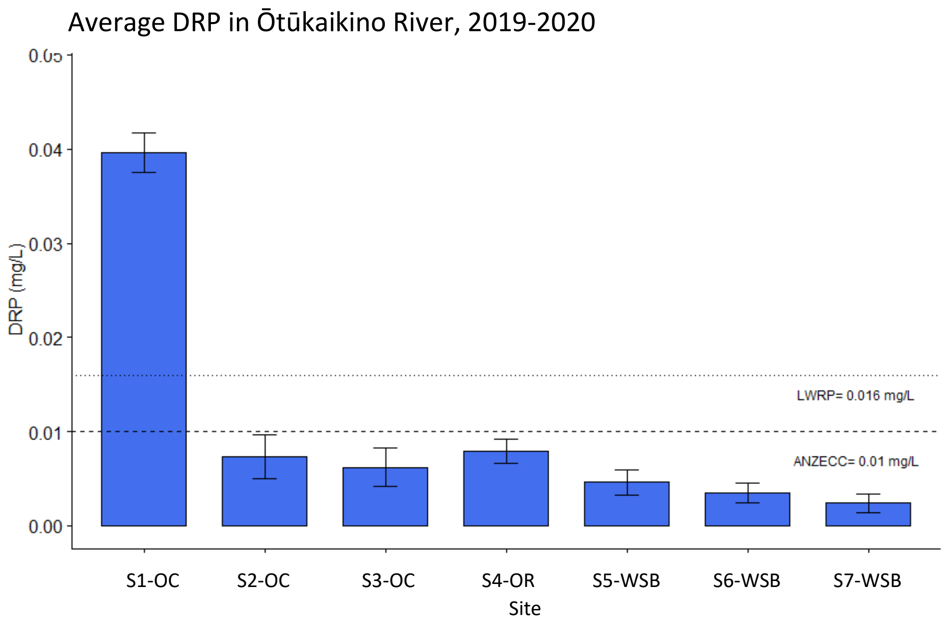

Figure 3.11: Average dissolved reactive phosphorus (n=4) of the 7 sampling sites, as measured between May 2019 and January 2020. ANZECC (2000) guidelines for lowland streams ecosystem protection and LWRP (Environment Canterbury, 2017) trigger values are included.

Total organic carbon (TOC) was similar between sites (Table 3.3). Average TOC ranged from 0.56 at Site 6 (Waimakariri South Branch) to 1.70 at Site 1 (Ōtūkaikino Creek). There were no clear spatial patterns.

Table 3.3: Average turbidity, total suspended solids, ammoniacal nitrogen, nitrate-nitrogen and total organic carbon concentration (n=4) for the 7 sampling sites, as measured between May 2019 and January 2020. ANZECC (2000) guidelines for lowland streams ecosystem protection are included for turbidity, ammoniacal nitrogen, and nitrate-nitrogen.

Site	Turbidity	Total suspended solids	NH_3-N	NO_3 -N	TOC
	NTU	mg/L	mg/L	mg/L	mg/L
S1-OC	0.16	0.58	0.034	0.153	1.70
S2-OC	0.13	1.05	0.053	0.300	0.60
S3-OC	0.12	0.85	0.046	0.283	0.64
S4-OR	0.25	1.15	0.009	0.265	1.02
S5-WSB	0.22	1.43	0.015	0.250	0.56
S6-WSB	0.37	0.38	0.018	0.153	0.93
S7-WSB	0.18	1.15	0.011	0.041	0.86
ANZECC (2000) trigger values for lowland streams	5.80		0.021	0.444	

3.3.5 Trace Element Concentrations

Major Ions

Table 3.4 presents the major ions in the catchment as measured in water samples from the July 2019 sampling round. Bromide was below detection levels at all sites. DIC as HCO_3 was the key major ion analysed in the catchment. Concentrations in July ranged from 36.2 mg/L to 51.3 mg/L. Calcium was also recorded at high levels comparative to other major ions analysed, with concentrations ranging from 10.6 mg/L to 14.3 mg/L.

Potassium, magnesium and sodium all show minor longitudinal changes; they were lower near the headwaters and increased in concentration downstream of this. Conductivity, however, did not show a clear increase downstream.

Aside from fluoride and sulphate, major ions tended to be at their lowest at Site 7, in the upper Waimakariri South Branch. This was consistent with the low average conductivity recorded at this site. Waimakariri South Branch sites also were also at lower concentrations compared to Ōtūkaikino Creek for calcium, sodium, DIC as HCO_3, and sulphate. In comparison, fluoride was consistently higher in the Waimakariri South Branch compared to the Ōtūkaikino Creek. Chloride was elevated at Site 5, in the middle reaches, compared to other sites in July, reaching 1.7 mg/L.

Table 3.4: Major ions from water samples in July 2019. Where results were below detection limits, <DL is used.

Site	Ca	K	Mg	Na	Br	Cl	DIC as HCO_3	F	SO_4
	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
S1-OC	13.3	0.64	1.34	3.37	>DL	1.44	49.4	0.040	5.26
S2-OC	14.3	0.79	1.54	3.53	>DL	1.41	51.3	0.039	5.21
S3-OC	13.8	0.84	1.60	3.76	>DL	1.37	49.7	0.046	5.10
S4-OR	14.1	0.91	1.80	4.06	>DL	1.26	49.0	0.052	5.12
S5-WSB	12.4	0.78	1.50	3.08	>DL	1.70	42.7	0.073	4.70
S6-WSB	11.6	0.71	1.31	2.58	>DL	1.21	36.4	0.073	5.27
S7-WSB	10.6	0.56	1.12	2.47	>DL	1.10	36.2	0.076	5.26

Other Trace Elements

Concentrations of various dissolved trace elements within the sample sites are shown in Table 3.5 (acid soluble fraction) and Table 3.6 (dissolved fraction). Both the acid soluble fraction and the dissolved fraction were analysed for each site.

Several trace elements were consistently below their detection limits: cadmium, cobalt, nickel, and lead. Several other trace elements were generally low (compared to ANZECC (2000) guidelines for lowland streams ecosystem protection) or below detection limits, except at a couple of sites. Acid soluble arsenic was above detection limits in the July 2019 sample at Sites 1 (Ōtūkaikino Creek) and 4 (Ōtūkaikino River), with the maximum concentration recorded being 6.7 µg/L at Site 1. Acid soluble chromium was only detectable at Site 1 in the July sample, where it was recorded at 1.1 µg/L. Acid soluble manganese was elevated at Site 4 in both May and July 2019. Most of the manganese was present in the dissolved fraction.

Acid soluble boron showed little spatial variation in the May 2019 samples (Figure 3.12). It was more variable during the July 2019 samples, with the lowest concentrations in the sites across the Waimakariri South Branch. Acid soluble boron was greater at all sites in the May samples compared to the July samples and was almost four times greater at Site 7 (Waimakariri South Branch) in May compared to July.

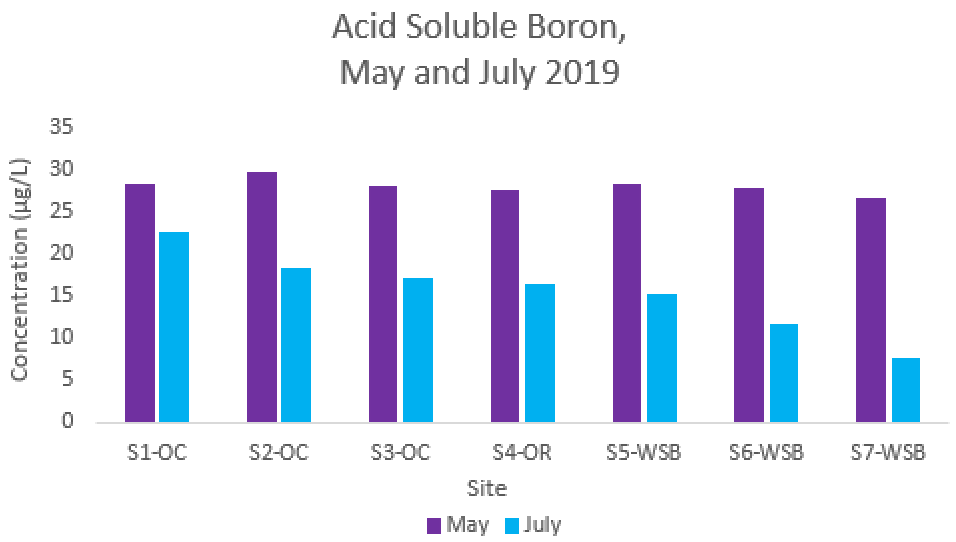

Figure 3.12: Acid soluble boron concentrations in May and July 2019 at 7 sites in the Ōtūkaikino River catchment.

Acid soluble aluminium levels were relatively consistent spatially and temporally. The exception to this was Site 6 (Waimakariri South Branch) in the July 2019 sample, which reached 35.8 µg/L. Most of the aluminium was composed of the dissolved fraction. Acid soluble molybdenum showed limited spatial variation, though was slightly elevated at Site 1 in the July 2019 sample. Acid soluble vanadium varied little between sites, though was slightly higher in the Waimakariri South Branch compared to the Ōtūkaikino Creek.

Acid soluble iron was spatially variable in the May 2019 samples (Figure 3.13). Sites 4 and 5 (mid catchment), and 6 (upper Waimakariri South Branch) were all elevated compared to the rest of the sites sampled. Site 4 recorded the maximum acid soluble iron concentration of 88.8 µg/L.

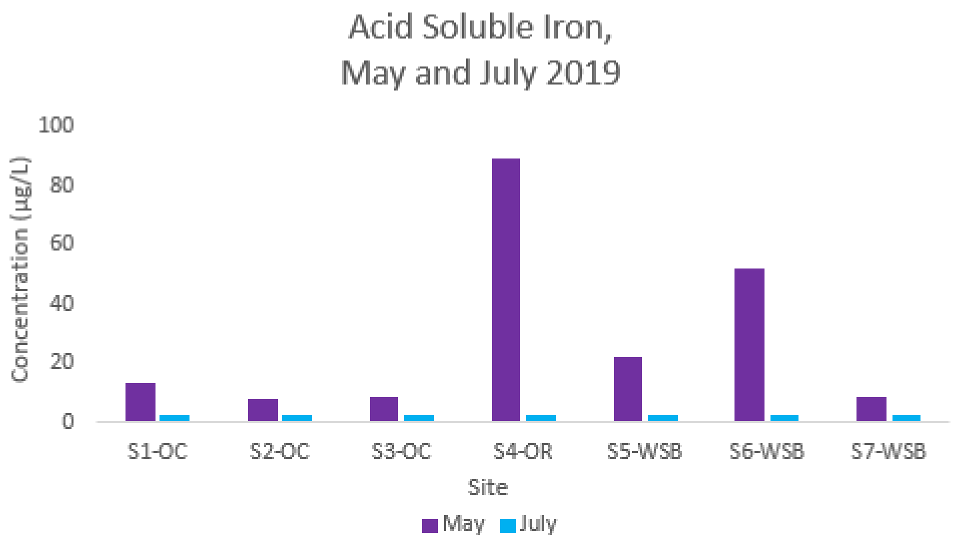

Figure 3.13: Acid soluble iron concentrations in May and July 2019 at 7 sites in the Ōtūkaikino River catchment. July concentrations were all below the detection limit.

There were a few other key differences in acid soluble trace element concentrations between the May and July 2019 sampling rounds. Acid soluble zinc was above detection limits at all sites in the May samples, though was below detection limits at all sites in the July samples (Figure 3.14). Acid soluble iron was below detectable limits at all sites in the July samples, though was detectable at all sites in the May samples.

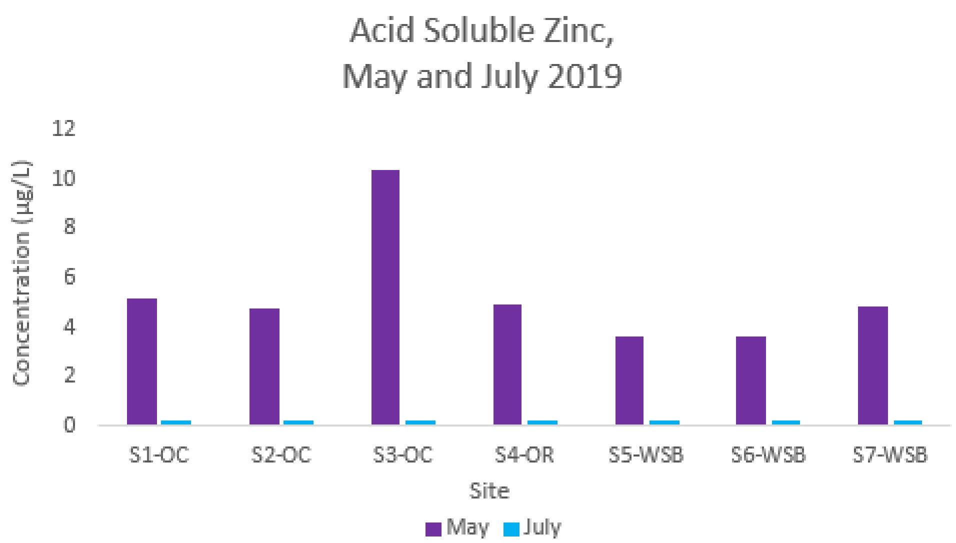

Figure 3.14: Acid soluble zinc concentrations in May and July 2019 at 7 sites in the Ōtūkaikino River catchment. July concentrations were all below the detection limit.

Table 3.5: Acid soluble fraction of trace element concentrations, as measured in May and July 2019 at 7 sites in the Ōtūkaikino River catchment. Where results were below detection limits, <DL is used. ANZECC 99%, 95% and 90% trigger values for ecosystem protection are included (ANZECC, 2000).

Site	Al	As (V)	B	Cd	Co	Cr (VI)	Cu	Fe	Mn	Mo	Ni	P	Pb	S	V	Zn
	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L
$S1_{May}$	10.2	<DL	28.1	<DL	<DL	<DL	<DL	13.1	0.9	1.2	<DL	69.2	<DL	130	8.7	5.1
$S1_{July}$	8.2	6.7	22.4	<DL	<DL	1.10	<DL	<DL	0.6	4.8	<DL	40.0	<DL	1930	10.0	<DL
$S2_{May}$	8.4	<DL	29.7	<DL	<DL	<DL	<DL	7.4	0.7	1.0	<DL	13.5	<DL	151	5.1	4.7
$S2_{July}$	11.9	<DL	18.1	<DL	<DL	<DL	<DL	<DL	1.5	1.9	<DL	<DL	<DL	2000	10.0	<DL
$S3_{May}$	6.5	<DL	27.9	<DL	<DL	<DL	<DL	8.4	0.7	1.3	<DL	8.15	<DL	172	<DL	10.4
$S3_{July}$	8.7	<DL	16.9	<DL	<DL	<DL	<DL	<DL	1.3	1.0	<DL	<DL	<DL	2000	<DL	<DL
$S4_{May}$	9.6	<DL	27.5	<DL	<DL	<DL	<DL	88.8	20.5	1.5	<DL	13.5	<DL	195	6.1	4.9
$S4_{July}$	10.8	6.3	16.3	<DL	<DL	<DL	<DL	<DL	18.0	1.1	<DL	<DL	<DL	1980	<DL	<DL
$S5_{May}$	13	<DL	28.2	<DL	<DL	<DL	1.2	21.5	1.1	1.2	<DL	15.7	<DL	232	<DL	3.6
$S5_{July}$	16.5	<DL	15.0	<DL	<DL	<DL	<DL	<DL	0.7	1.2	<DL	<DL	<DL	2140	10.0	<DL
$S6_{May}$	35.8	<DL	27.6	<DL	<DL	<DL	2.0	51.8	1.3	1.6	<DL	9.8	<DL	252	<DL	3.6
$S6_{July}$	13.3	<DL	11.6	<DL	<DL	<DL	<DL	<DL	0.1	<DL	<DL	<DL	<DL	2000	<DL	<DL
$S7_{May}$	5.8	<DL	26.5	<DL	<DL	<DL	<DL	8.0	0.4	1.1	<DL	10.7	<DL	274	<DL	4.8
$S7_{July}$	6.2	<DL	7.6	<DL	<DL	<DL	<DL	<DL	<DL	0.9	<DL	<DL	<DL	1900	10.0	<DL
ANZECC 99%	27.0	0.8	90.0	0.06		0.01	1.0		1200.0		8.0		1.0			2.4
ANZECC 95%	55.0	13.0	370.0	0.20		1.00	1.4		1900.0		11.0		3.4			8.0
ANZECC 90%	80.0	42.0	680.0	0.40		6.00	1.8		2500.0		13.0		5.6			15.0

Table 3.6: Dissolved fraction of trace element concentrations, as measured in May and July 2019 at 7 sites in the Ōtūkaikino River catchment. Where results were below detection limits, <DL is used. These results should not be compared to ANZECC (2000) guideline values for lowlnd river ecosystem protection.

Site	Al	As (V)	B	Cd	Co	Cr (VI)	Cu	Fe	Mn	Mo	Ni	P	Pb	S	V	Zn
	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L
S1$_{May}$	6.3	<DL	28.0	<DL	**<DL**	<DL	<DL	8.8	0.5	1.1	<DL	63.4	<DL	139[1]	4.2	1.5
S1$_{July}$	6.1	<DL	20.1	<DL	<DL	<DL	<DL	<DL	0.3	2.5	<DL	40	<DL	1900	<DL	<DL
S2$_{May}$	7.0	<DL	29.3	<DL	<DL	<DL	<DL	3.6	0.7	1.0	<DL	13.2	<DL	151[1]	<DL	0.9
S2$_{July}$	7.7	<DL	17.9	<DL	<DL	<DL	<DL	<DL	1.2	1.5	<DL	<DL	<DL	2000	<DL	<DL
S3$_{May}$	5.3	<DL	27.9	<DL	<DL	<DL	<DL	3.7	0.5	1.2	<DL	8.2[1]	<DL	172[1]	<DL	0.6
S3$_{July}$	7.0	<DL	16.5	<DL	<DL	<DL	<DL	<DL	1	<DL	<DL	<DL	<DL	2020	<DL	<DL
S4$_{May}$	6.5	<DL	27.5[1]	<DL	<DL	<DL	<DL	58.7	19.2	1.1	<DL	13.5[1]	<DL	195[1]	<DL	0.7
S4$_{July}$	8.7	<DL	16.0	<DL	<DL	<DL	<DL	<DL	16.2	1.1	<DL	<DL	<DL	1950	<DL	<DL
S5$_{May}$	7.7	<DL	27.6	<DL	<DL	<DL	0.6	8.8	0.7	0.8	<DL	14.7	<DL	232[1]	<DL	0.5
S5$_{July}$	11.4	<DL	14.5	<DL	<DL	<DL	<DL	<DL	0.7	1.0	<DL	<DL	<DL	2130	<DL	<DL
S6$_{May}$	15.0	<DL	27.0	<DL	<DL	<DL	1.1	17.7	0.8	1.4	<DL	7.9	<DL	252[1]	<DL	0.5
S6$_{July}$	8.8	<DL	10.6	<DL	<DL	<DL	<DL	<DL	<DL	<DL	<DL	<DL	<DL	1990	<DL	<DL
S7$_{May}$	5.0	<DL	26.5	<DL	<DL	<DL	<DL	6.7	0.3	1.1[1]	<DL	7.9	<DL	274[1]	<DL	1.5
S7$_{July}$	5.8	<DL	7.0	<DL	<DL	<DL	<DL	<DL	<DL	0.8	<DL	<DL	<DL	1900[1]	10	<DL

[1] Where the soluble fraction concentration was higher than the acid soluble fraction, it was assumed that these were in fact the same. This was likely due to contamination during filtration.

3.3 Sediment Quality

3.3.1 Trace Elements in Sediment

Table 3.7 provides a summary of the sediment quality as measured in the May 2019 samples. Arsenic concentrations were low at all sites except for Site 7 (26.9 mg/L). Site 7 (upper Waimakariri South Branch) also showed elevated levels of several other trace elements: boron, chromium, molybdenum nickel, sulphur, vanadium and zinc. These trace elements were at comparatively low concentrations across the rest of the sites sampled.

Copper concentrations were elevated compared to the rest of the catchment in both Site 6 marginal sediment and at Site 7, at 345.8 mg/kg and 380.6 mg/kg respectively (Figure 3.15). Lead concentrations were also elevated at some sites compared to ANZECC (2000) interim sediment quality guidelines in Site 6 marginal sediment (134.1 mg/kg) and at Site 7 (211.5 mg/kg) in the upper Waimakariri South Branch, as well as at Site 2 in the upper Ōtūkaikino Creek (61.6 mg/kg) (Figure 3.16).

Of potential interest was the difference between the marginal and instream sediment concentrations at Site 6, particularly for copper and lead. Copper levels were almost 50 times higher in marginal sediment compared to instream sediment, reaching 380.6 mg/kg. Similarly, lead concentrations were approximately 20 times higher in marginal sediment compared to instream sediment at Site 6.

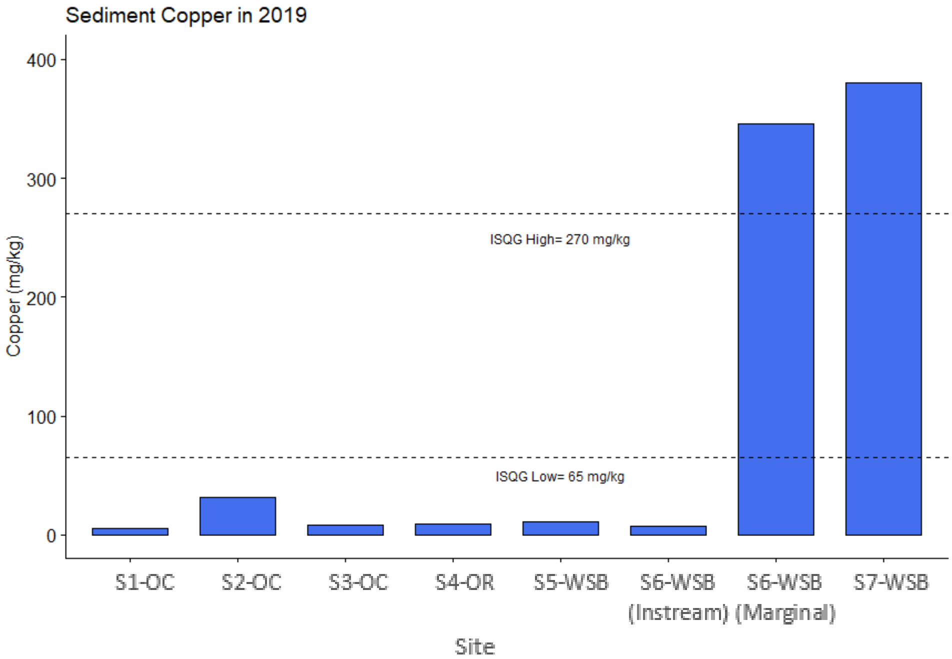

Figure 3.15: Sediment concentrations of copper, as measured in May 2019 at 7 sites in the Ōtūkaikino River catchment. ANZECC (2000) interim sediment quality guidelines (ISQG) are included.

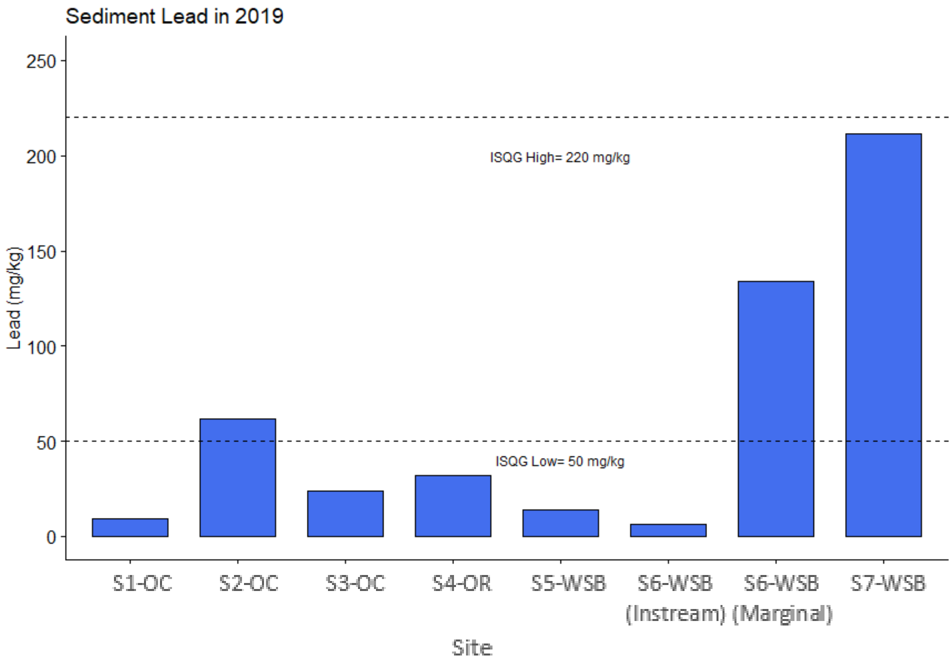

Figure 3.16: Sediment concentrations of lead, as measured in May 2019 at 7 sites in the Ōtūkaikino River catchment. ANZECC (2000) interim sediment quality guidelines (ISQG) are included.

Table 3.7: Sediment metal and nutrient concentrations as measured in May 2019 at 7 sites in the Ōtūkaikino River catchment. For Site 6, the sediment concentration of both marginal and instream sediment is included. ANZECC (2000) interim sediment quality guidelines (ISQG) are included.

Site	Al	As	B	Cd	Co	Cr	Cu	Fe	Mn	Mo	Ni	P	Pb	S	V	Zn
	wt%	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	wt%	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
S1-OC	0.70	1.9	5.2	1.9	0.6	11.0	5.8	0.91	197	0.3	7.5	521	9.2	113	151	70.0
S2-OC	0.78	2.2	4.8	1.9	0.5	12.2	31.2	1.00	167	0.3	7.8	562	61.6	159	170	48.3
S3-OC	0.94	2.7	5.2	1.8	0.8	14.1	8.6	1.17	207	0.4	9.9	422	23.7	85	208	58.9
S4-OR	0.80	2.9	5.0	1.6	0.7	11.7	9.3	1.00	186	0.2	7.5	480	31.7	43	200	40.8
S5-WSB	0.93	2.3	4.7	1.7	0.6	13.1	11.3	1.11	176	0.2	10.7	378	13.9	36	199	66.9
S6-WSB (Marginal)	1.38	4.6	8.2	1.7	1.2	21.2	345.8	1.63	267	0.9	13.3	959	134.1	258	306	67.7
S6 -WSB (Instream)	0.65	1.7	4.1	3.1	0.4	10.2	7.1	0.77	133	0.8	6.2	428	6.4	196	136	34.6
S7-WSB	1.91	26.9	44.0	4.5	1.1	59.0	380.6	2.56	289	5.2	34.2	933	211.5	2719	853	379.9
ANZECC Low		20.0		1.5		80.0	65.0				21.0		50.0			200.0
ANZECC High		70.0		10.0		370.0	270.0				52.0		220.0			410.0

3.3.2 Grain Size Analysis

Particle size distribution of the sediment samples is shown in Figure 3.17. Site 3, in the Ōtūkaikino Creek, was dominated by gravel (>2 mm). In comparison, Site 6 margins and Site 7, in the upper Waimakariri South Branch, had a higher proportion of silt/clay (<0.063 mm).

Of potential interest was also the difference in grain size composition at Site 6 between the sample taken from the stream margins and the sample taken from instream. The marginal sediment, with a strong smell and different texture, had a high proportion of silt/clay (<0.063 mm). In comparison, the instream sediment was primarily composed of sand (0.25-0.5 mm).

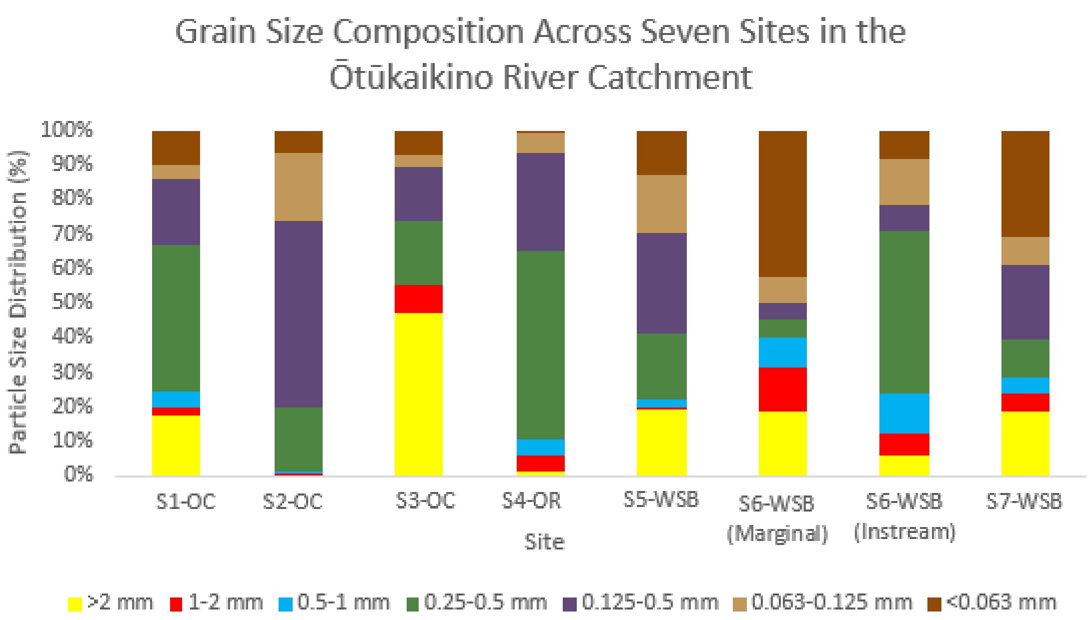

Figure 3.17: Particle size distribution (%) in sediment samples as measured in May 2019 at 7 sites in the Ōtūkaikino River catchment.

3.4 Macroinvertebrates

A total of 27765 macroinvertebrates, across 44 taxonomic groups, were collected in July 2019 from the 10 sites sampled across the Ōtūkaikino River catchment. The various biotic indices calculated are shown in Table 3.8 and the raw data in Appendix 2.

Table 3.8: Macroinvertebrate indices calculated from one sample date in July 2019 at 10 sites in the Ōtūkaikino River catchment.

Site	Total Abundance	Taxonomic Richness	EPT Taxa	% EPT	SQMCI	MCI
S1-OC	2162	22	10	45.5	4.2	91.8
S2-OC	5208	28	12	42.9	5.7	99.3
S3-OC	4892	29	14	48.3	5.9	100.7
S4-OR	1125	23	10	43.5	5.8	105.2
S5-WSB	2636	27	11	40.7	7.7	103.7
S6-WSB	1761	25	10	40.0	3.3	98.4
S7-WSB	1703	24	9	37.5	3.6	90.0
S8-OC	4580	27	11	40.7	6.7	103.7
S9-OR	2966	24	13	54.2	6.8	104.2
S10-WSB	732	20	8	40.0	3.6	91.0

3.4.1 Abundance and Taxonomic Richness

Total macroinvertebrate abundance varied between sites, from 732 individuals at Site 10 in the headwaters of the Waimakariri South Branch to 5208 individuals at Site 2 in the Ōtūkaikino Creek (Figure 3.18). Sites 2, 3 and 8 in the Ōtūkaikino Creek all contained more than 4500 individuals within the kick-net sample.

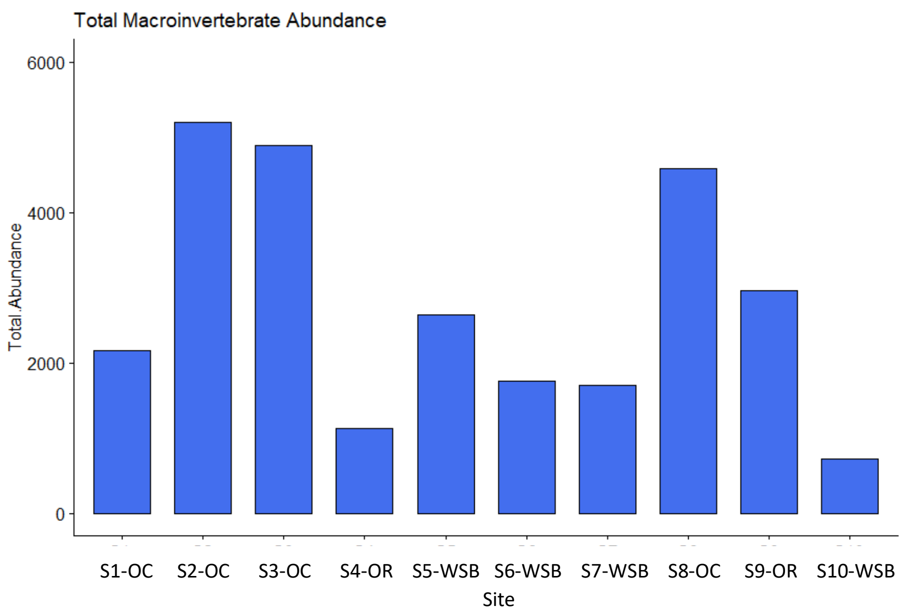

Figure 3.18*: Total macroinvertebrate abundance at 10 sites in the Ōtūkaikino River catchment, sampled using a kick-net in July 2019.*

Taxonomic richness was somewhat similar across the sites (Figure 3.19). It ranged from 20 to 29 taxa per site. Sites 10 and S1 in the catchment headwaters had the fewest taxa recorded in 2019, with 20 and 22 respectively. Site 3 in the Ōtūkaikino Creek recorded the highest taxonomic diversity (29 taxa), followed by Site 2 (28 taxa).

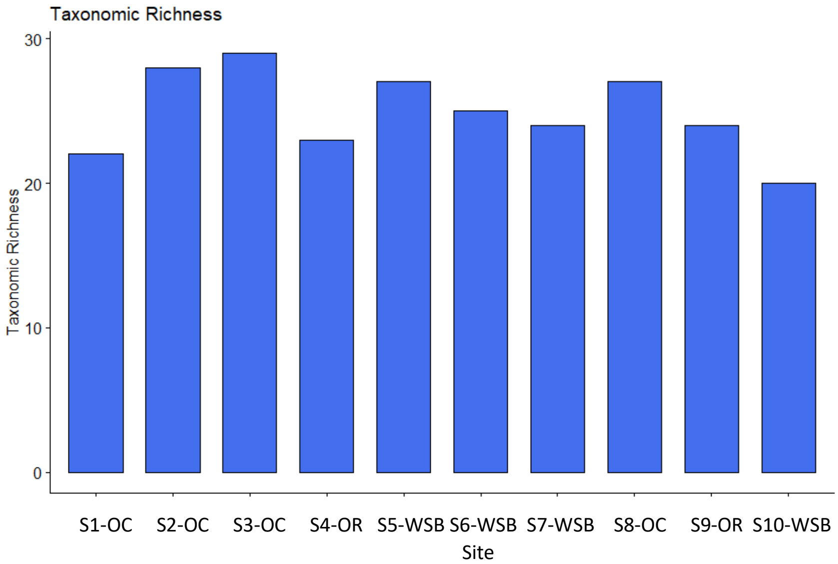

Figure 3.19: Taxonomic richness at 10 sites in the Ōtūkaikino River catchment, sampled in July 2019.

EPT richness was also somewhat consistent between sites (Figure 3.20). EPT refers to the three main insect orders typically used as indicators of good stream health: *Ephemeroptera* (mayflies), *Plectoptera* (stoneflies) and *Tricoptera* (caddisflies). EPT richness was lowest at Site 10 in the Waimakariri South Branch headwaters, with 8 EPT taxa, and highest at Site 3 in the Ōtūkaikino Creek (14 EPT taxa). Percentage EPT was also consistent across the sites visited, ranging from 37.5% EPT (Site 3) to 54% EPT (Site 9) (Figure 3.21).

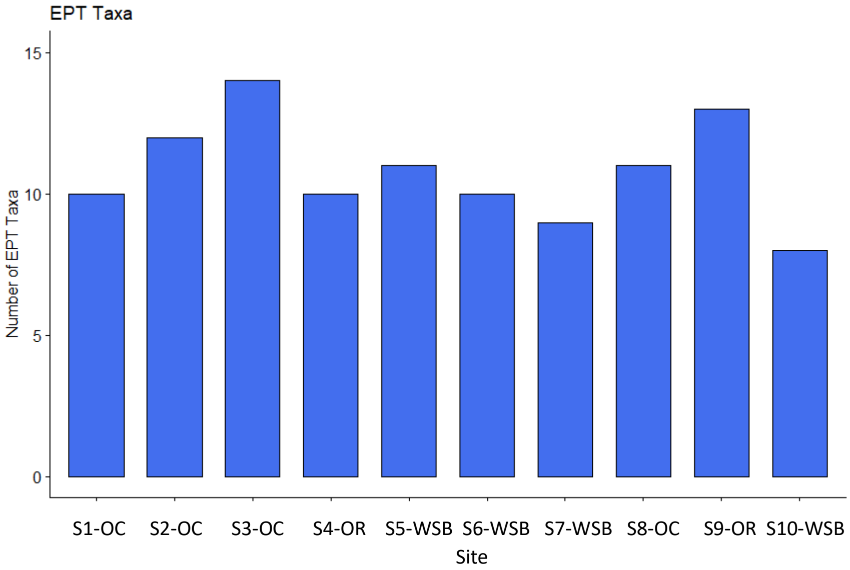

Figure 3.20: Number of EPT (*Ephemeroptera, Plecoptera,* and *Tricoptera*) taxa at 10 sites in the Ōtūkaikino River catchment, sampled in July 2019.

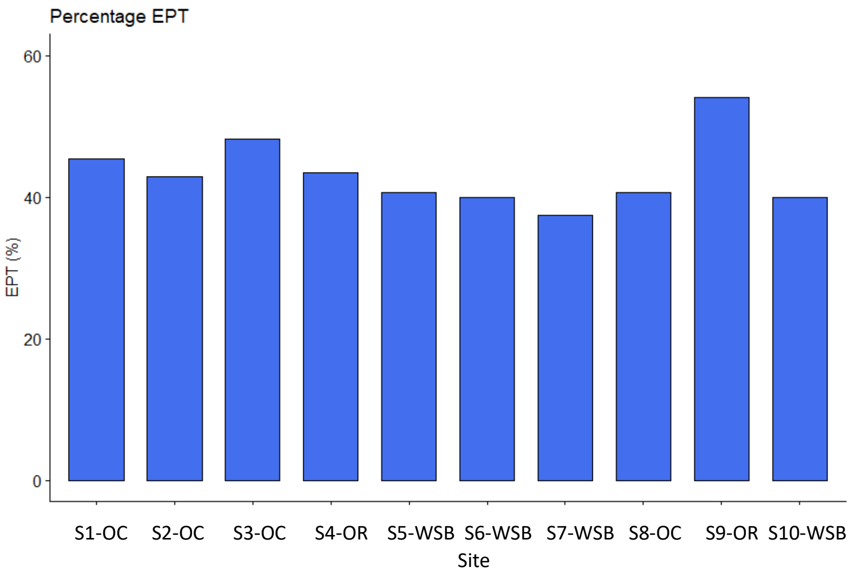

Figure 3.21: Percentage EPT (*Ephemeroptera, Plecoptera,* and *Tricoptera*) taxa at 10 sites in the Ōtūkaikino River catchment, sampled in July 2019.

3.4.2 Macroinvertebrate Indices

MCI scores varied between the sites monitored in July 2019 (Figure 3.22). MCI scores ranged from 90 at Site 7 in the upper Waimakariri South Branch to 105 at Site 4 in the mid catchment. These were assessed using the categories of Stark and Maxted (2007). The scores placed half of the sites as having fair scores (in relation to macroinvertebrate community composition) and the remainder as having good scores. MCI scores were consistently lower in the headwater sites and increased down the catchment, with the highest MCI scores generally recorded around the middle of the catchment.

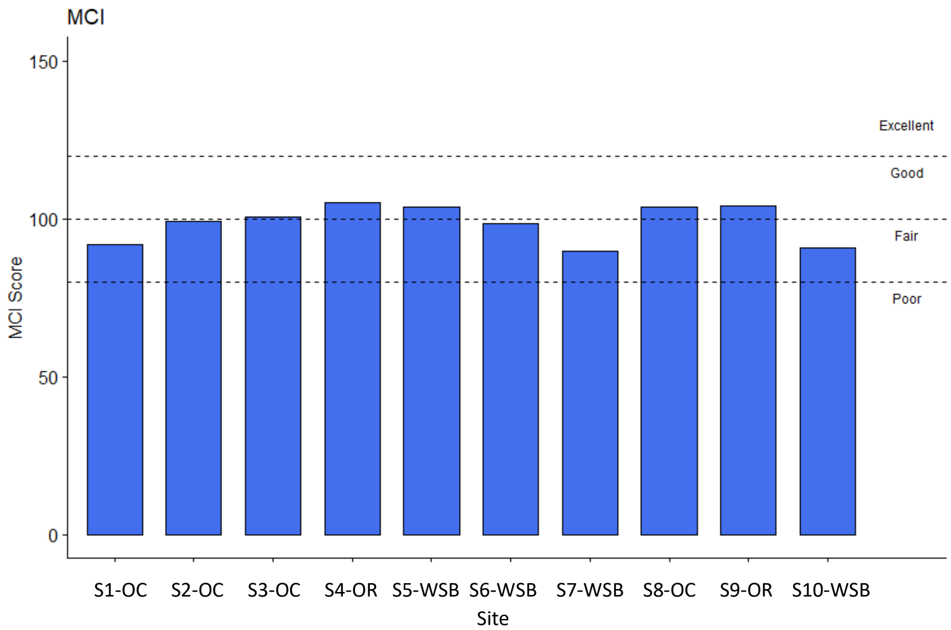

Figure 3.22: MCI scores at 10 sites in the Ōtūkaikino River catchment, sampled in July 2019. Categories are taken from Stark and Maxted (2007).

The SQMCI scores in 2019 showed more variation between sites than the MCI scores (Figure 3.23). SQMCI scores ranged from 3.3 (Site 6 in the upper Waimakariri South Branch) to 7.7 (Site 5 in the mid catchment). Sites 6, 7 and 10, located in the upper reaches of the Waimakariri South Branch, had poor SQMCI scores. Using SQMCI scores compared to MCI scores, Site 1 was still rated as fair. Site 2 now had a good score, as well as Sites 3 and 4. Sites 5, 8 and 9, with good MCI scores, had excellent scores for SQMCI.

SQMCI scores, like MCI scores, tended to be a bit higher in the mid-reaches compared to the headwaters. However, Site 8 in the Ōtūkaikino Creek had an excellent SQMCI score, despite being upstream of Sites 2, 3 and 4.

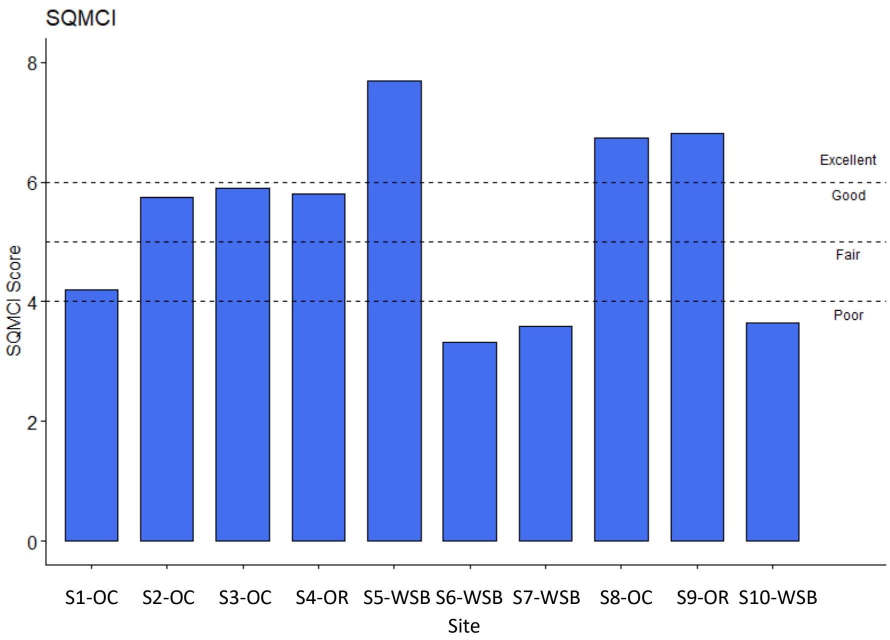

Figure 3.23: SQMCI scores at 10 sites in the Ōtūkaikino River catchment, sampled in July 2019. Categories are taken from Stark and Maxted (2007).

3.4.3 Macroinvertebrate Community Composition

The most diverse group of macroinvertebrates in 2019 was *Tricoptera* (caddisflies), with 12 different taxa recorded across the 10 sites (Table 3.9). This was closely followed by *Diptera* (true flies), with 11 different taxa. Several of the taxa found in 2019 are shown in Figure 3.24.

Table 3.9: Taxa presence (in orange) at 10 sites in the Ōtūkaikino River catchment sampled in July 2019.

Higher Taxonomic Group	Taxon	Presence/Absence at Sites									
		Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10
Acarina	Acarina	●	●	●		●	●	●	●	●	
Annelia	Oligochaeta	●	●	●	●	●	●	●	●	●	●
Coleoptera	Elmidae	●	●	●		●	●	●	●		●
Collembola	Collembola			●		●			●		
Crustacea	Amphipoda			●		●		●			
Crustacea	Cladocera				●	●					
Crustacea	Copepoda			●	●			●	●		●
Crustacea	Ostracoda	●	●	●	●	●	●	●	●	●	●
Crustacea	Paracalliope		●	●	●	●			●	●	
Diptera	Austrosimulium	●	●				●	●	●		●
Diptera	Corynoneura	●						●			●
Diptera	Empididae			●							
Diptera	Lobodiamesa		●	●	●	●	●		●		
Diptera	Mischoderus		●	●		●		●	●		
Diptera	Orthocladiinae	●	●	●	●	●	●	●	●	●	●
Diptera	Paradixa										●
Diptera	Tanyderidae					●					
Diptera	Tanypodinae	●	●						●	●	
Diptera	Tanytarsini						●				
Diptera	Tipulidae		●						●		
Ephemeroptera	Coloburiscus		●	●	●	●			●	●	
Ephemeroptera	Deleatidium	●	●	●	●	●		●	●	●	●
Hemiptera	Sigara						●			●	
Mollusca	Gyraulus	●	●		●		●			●	●
Mollusca	Physella	●	●	●	●	●	●	●	●	●	●
Mollusca	Potamopyrgus	●	●	●	●	●	●	●	●	●	●
Mollusca	Sphaeriidae	●	●	●	●	●	●		●	●	●
Odonata	Xanthocnemis					●	●	●			
Plecoptera	Zelandobius		●	●				●			
Trichoptera	Helicopsyche			●	●	●	●		●	●	
Trichoptera	Hudsonema	●	●	●	●	●	●	●	●	●	●
Trichoptera	Hydrobiosis	●	●	●	●	●	●	●	●	●	●
Trichoptera	Hydropsyche-Aoteapsyche	●	●	●	●	●		●	●	●	
Trichoptera	Neurochorema	●	●	●						●	●
Trichoptera	Oecetis		●	●			●			●	
Trichoptera	Olinga	●	●	●	●	●	●	●	●	●	●
Trichoptera	Oxyethira	●	●	●	●	●	●	●	●	●	●
Trichoptera	Polyplectropus				●		●	●			
Trichoptera	Psilochorema	●	●	●	●	●		●	●	●	●
Trichoptera	Pycnocentria	●	●	●	●	●	●	●	●	●	●
Trichoptera	Pycnocentrodes	●	●	●	●	●	●	●	●	●	
Trichoptera	Triplectides						●	●			

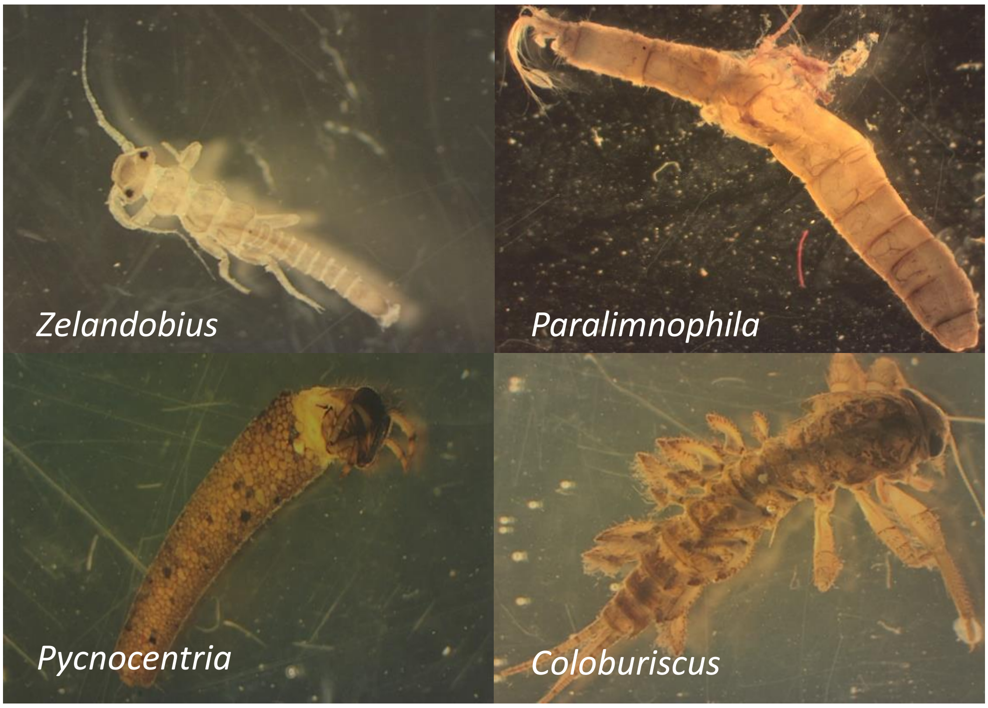

Figure 3.24: Photos of several of the taxa found during macroinvertebrate sampling in July 2019.

The cased caddisfly *Pycnocentria* was a dominating taxon at many sites, though *Potamopurgus* snails and *Deleatidium* mayflies were also typically in high numbers. *Tricoptera* were found in all sites sampled, typically dominating the macroinvertebrate community. In 2019, their percentage composition varied between 15% and 61% of the total macroinvertebrate community abundance (Figure 3.25). The stony cased *Pycnocentria* was found in high numbers at all sites. *Mollusca*, especially *Physa* and *Potamopyrgus*, were also relatively common across the 10 sites, with percentage abundance varying between 1% and 50%.

Zelandobius stoneflies were found in the Ōtūkaikino River catchment, though at very low abundances (Figure 3.26). Across the 10 sites sampled, there was one stonefly at Site 2 (Ōtūkaikino Creek), one stonefly at Site 3 (mid catchment) and two stoneflies at Site 7 (Waimakariri South Branch). Stoneflies had previously been recorded at Sites 3 and 7 (as well as Sites 1 and 6) but not at Site 2.

In addition, two *Paralimnophila* individuals, a primitive cranefly, were found at Sites 2 and 3. This taxon was last found in the catchment in 2008, with one individual at Site 5.

The spiral cased *Helicopsyche*, a genus of *Tricoptera*, was found at 6 of the sites sampled in 2019. With an MCI score of 10, this taxon is very sensitive to water quality. This taxon composed 25% of the macroinverebrates sampled at Site 5 in the mid catchment. *Olinga*, with an MCI score of 9, was found at all 10 sites. *Coloburiscus*, which also has an MCI score of 9, was found at 6 of the 10 sites.

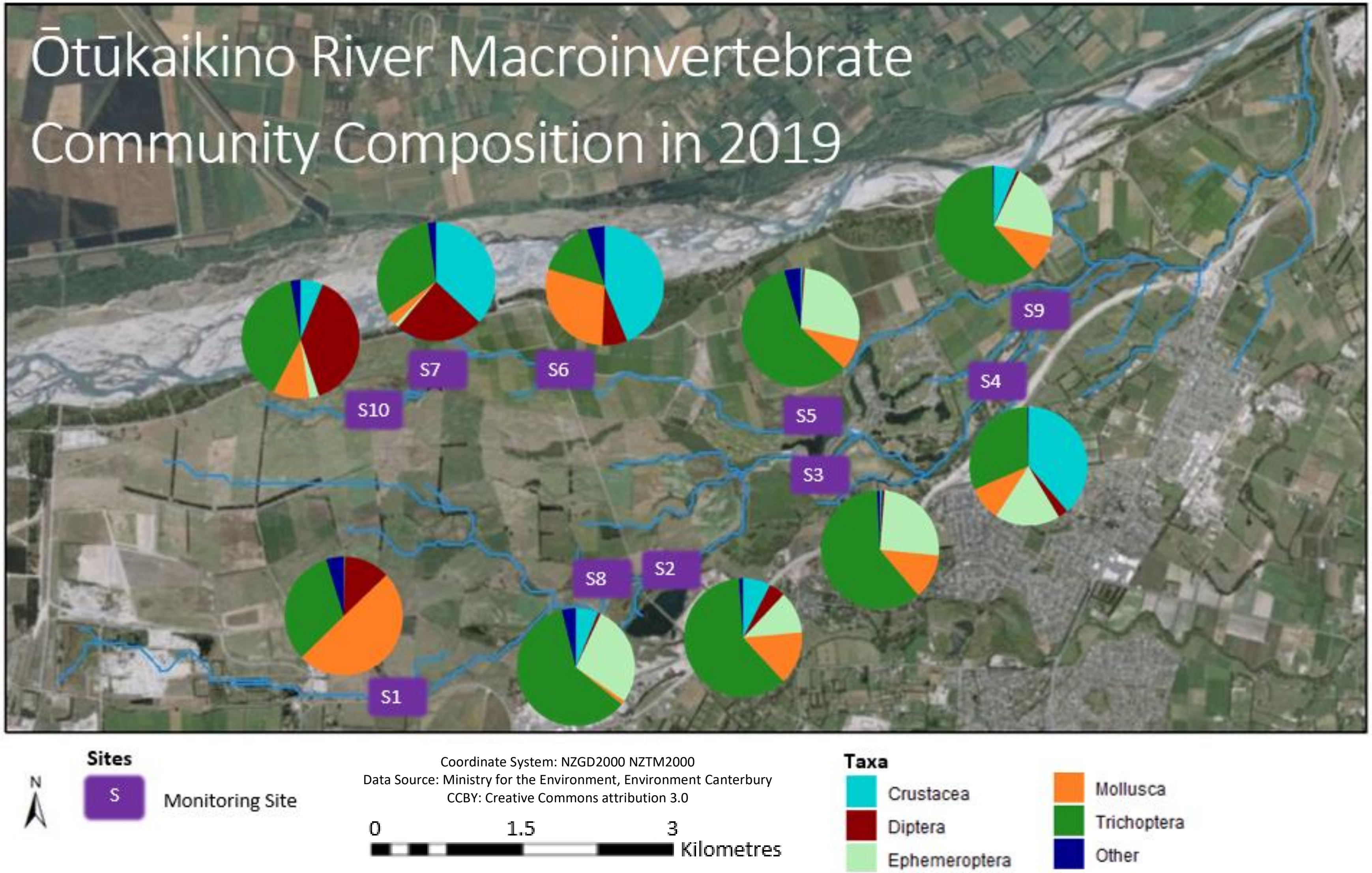

Figure 3.25: Macroinvertebrate community composition (%) found at 10 sites in the Ōtūkaikino River catchment sampled in July 2019.

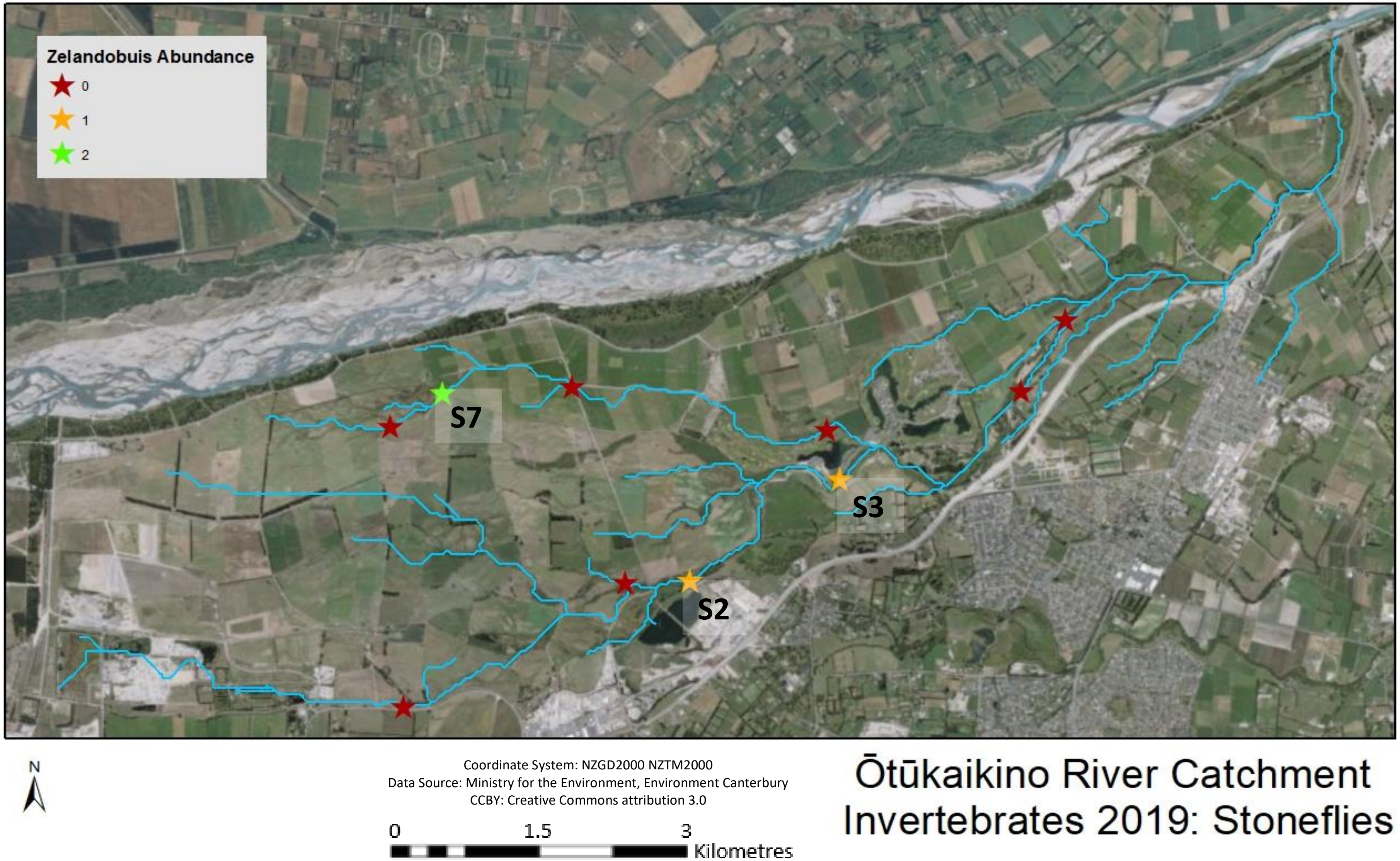

Figure 3.26: Location and number of *Zelandobius* stoneflies found in July 2019.

4 Discussion

4.1 Spatial and Temporal Changes in Ōtūkaikino Surface Water and Sediment Quality

A key objective of this research was to quantify any spatial and temporal changes in the state of the Ōtūkaikino River. During the monitoring programme undertaken in 2019 and 2020, numerous areas of interest were identified in the surface water and sediment quality of the Ōtūkaikino River.

4.1.1 Macrophytes and Periphyton

Macrophyte cover was consistently low in the mid catchment at Sites 4, 5 and 9. This was likely linked to the high canopy cover at these sites, which helps to shade out the macrophytes. The coarser substrates also make it more challenging for macrophytes to take root.

The Land and Water Regional Plan (LWRP) (Environment Canterbury, 2017) guidelines of a maximum of 50% total cover for spring-fed plains waterways were exceeded six times. This was generally at sites with limited canopy cover and so limited ability to shade out macrophytes. Macrophyte cover at Site 7 in the upper Waimakariri South Branch exceeded this threshold three times, with high levels of emergent macrophytes compared to other sites. While the site did have a buffer of willows on one bank, the other bank had no established riparian margin. The reach just upstream of the sample site also lacked a riparian buffer. Site 6 in the Waimakariri South Branch would have recorded high levels of emergent macrophyte cover, but they were mechanically removed in April 2019, prior to sampling. The riparian zone around Site 6 was mostly grassed, with no canopy cover. Macrophyte cover was also high at Site 1 (upper Ōtūkaikino Creek) in January 2020. This site also had minimal riparian vegetation, as well as increased levels of DRP relative to other sites in the catchment. Both of these factors could have contributed to the increased macrophyte growth in January 2020.

Periphyton cover was highest at Sites 1, 6 and 10 in the upper catchment. The regular flow and lack of shading, and consequently increased temperatures, likely contributed to this.

4.1.2 Main Water Quality Parameters

pH, water temperature and conductivity were all relatively consistent during the monitoring programme. pH showed little variation across the catchment. The sites all met the Land and Water Regional Plan's guidelines of 6.5 – 8.5 pH (Environment Canterbury, 2017).

Water temperatures were generally similar across the sites sampled. The coolest site was Site 10 in July 2019, while the warmest was Site 6 in January 2020. While shading did vary across the catchment, the Ōtūkaikino River is relatively small and fed from Waimakariri River seepage. This helped keep water temperatures cool, particularly at headwater sites like Site 10. All sites sampled were below the Land and Water Regional Plan's maximum guidelines of 20 ^{0}C (Environment Canterbury, 2017). However, temperature can also vary daily and seasonally, so exceedances of this guideline value are still possible (Riđanović et al., 2010). Temperature variation was less in the middle reaches compared to the upper reaches, likely because of canopy cover and shading, which increased downstream. Stream temperatures were warmer in January than in other sampling rounds. As this was mid-summer, with warmer air temperature compared to other months sampled, this was not a surprise.

Conductivity was generally lower at the sites sampled in the Waimakariri South Branch compared to the Ōtūkaikino Creek or main stem of the Ōtukaikino River. This was in line with the major ion concentrations recorded in July 2019, which were also typically lowest in the Waimakariri South Branch. These conductivity levels were similar to those previously recorded in the catchment (Marshall & Noakes, 2019). Wilsons Drain, a tributary in the lower reaches of the Ōtūkaikino River which was not sampled in this monitoring programme, does typically record higher conductivity, though is still low compared to rivers like the Avon (Marshall & Noakes, 2019).

Most sites sampled did not meet the ANZECC (2000) guidelines of 98% - 105% dissolved oxygen saturation for daytime sampling in slightly disturbed lowland rivers. The Waimakariri River Regional Plan (WRRP) guidelines of greater than 80% saturation were also not met on five occasions (Environment Canterbury, 2011). Only Sites 2, 4 and 6 met this guideline consistently, with the remaining seasonal sites dropping below at least once.

On four of the five occasions sampled, dissolved oxygen at Site 6 in the Waimakariri South Branch exceeded 100% saturation. The site typically had high levels of periphyton and was sampled a little after midday each time. The higher levels of photosynthesis at that time likely contributed to the increased levels of dissolved oxygen. This does indicate that dissolved oxygen levels are likely to show increased diurnal variation at this site (Cummings et al., 2016). However, as each site was revisited at a similar time of day, this wasn't tested during the monitoring programme.

In January 2020, Sites 1 and 7 had low levels of dissolved oxygen. At Site 1 in the upper Ōtūkaikino Creek, where dissolved oxygen was recorded at 5.77 mg/L, there was high macrophyte cover and low flow, with the stream disappearing completely about 30 m downstream of the sample site. At Site 7 in the upper Waimakariri South Branch, with dissolved oxygen at 5.57 mg/L, water velocity was also slower than in previous sampling. These somewhat stagnant conditions, along with warm temperatures and increased faecal contamination, are known to contribute to low concentrations of dissolved oxygen (Riđanović et al., 2010).

The high macrophyte cover at Site 1 also indicates that dissolved oxygen was likely to decrease substantially at night (Cummings et al., 2016), putting the biological community under increased stress. This dissolved oxygen variation could have implications for fish and invertebrate life, especially for species like trout that are sensitive to low oxygen levels (Abowei, 2010).

4.1.3 Faecal Contamination

The *E. coli* levels recorded were below recreational standards in New Zealand, except for Site 6 in January 2020 (Ministry for the Environment & Ministry of Health, 2003). Despite the high faecal coliform levels at some sites, the measured *E. coli* levels indicated low pathogenic disease risk.

Faecal coliforms were consistently high at Site 1 in the upper Ōtūkaikino Creek, reaching a maximum of 8450 CFU/100mL in January 2020. However, *E. coli* levels tended to be low at Site 1, not exceeding 66.7 CFU/100mL. Site 1 was located on a sheep farm, though it did not appear that sheep had been grazing near the waterway during the times monitored. While the site lacked a riparian buffer, the fence line on the true left bank extended for more than 30 m. This likely limited the potential contributions of the sheep farming to faecal coliform levels at the site.

Isaac Conservation Trust operates several aviaries located a couple of kilometres upstream of Site 1. These aviaries are used for captive breeding of threatened New Zealand birds. They are cantilevered over the waterway by its headwaters, in an area called Peacock Springs. These birds therefore have the potential to contribute high levels of faecal contamination to the water (Wither et al., 2005). This is the most likely source of the elevated faecal coliform levels at Site 1.

Site 7 in the upper Waimakariri South Branch also showed elevated levels of faecal coliforms in May and October 2019, though to a lesser extent than Site 1. Site 7 bordered a paddock which was stocked with cattle in May and October. While one bank had a thick riparian buffer, the other bank, where the cattle were present, was only covered in grass. Just upstream of the site, there was limited riparian planting across much of the headwaters, as well as gaps in fencing. A paddock upstream that bordered the river was also occasionally stocked with sheep, including in October 2019, which could be a contributing factor. Stock was therefore a likely source of the recorded faecal contamination.

E. coli and faecal coliform levels were particularly high at most sites in January 2020 compared to previous sampling in 2019. At Site 6 in the Waimakariri South Branch, this was likely linked to the movement of cattle through the neighbouring paddocks the day before sampling, with cattle being shifted to the paddock adjoining the waterway. In addition, a pivot irrigator was in operation just beside the sampling site on the day of monitoring and was spraying across the area where cattle had been walking, contributing surface runoff into the waterway. With limited riparian vegetation aside from grass, this all provided a probable source for the elevated *E. coli* and faecal coliform levels in January 2020 at this site.

As Sites 1, 6 and 7 are located within pastoral land, high levels of faecal contamination were not a concern for recreation. Though the elevated levels of faecal contamination at Sites 2 and 4 in January 2020 were potentially of concern, as these two sites are used for recreation, *E. coli* counts were still below recreational guidelines (Ministry for the Environment & Ministry of Health, 2003).

These results were in line with those recorded in 2018: generally low levels of faecal contamination with some exceedances (Marshall & Noakes, 2019). Sites 2 and 4 were generally below recreational guidelines in 2018, though both reached about 4000 CFU/100 mL for *E. coli* on one occasion. Wilsons Drain, in the lower catchment, tended to be a bit higher and was recorded at >24,000 CFU/100 mL for *E. coli* in 2018 (Marshall & Noakes, 2019). These exceedances were likely related to rain events. A popular swimming hole, at the mouth of the Ōtūkaikino River, has a poor grade for contact recreation as elevated levels of *E. coli* are regularly recorded over the summer (LAWA, 2020). These levels of faecal contamination, as well as being public health concerns in some locations, can impact on how people view and interact with the waterway.

4.1.4 Turbidity and total suspended solids

All sites during the monitoring period had low levels of suspended sediment. Turbidity and total suspended solids were consistently low. ANZECC (2000) and the LWRP (Environment Canterbury, 2017) lack appropriate guidelines for total suspended solids in New Zealand waters. Following the methodology of Margetts and Marshall (2018), a guideline value of 25 mg/L for total suspended solids was chosen for this project (Hayward et al., 2009; Stevenson et al., 2009). All sites measured were well below this. In addition, turbidity at all sites were well below the ANZECC (2000) lowland river guidelines of 5.6 NTU. Fine sediment was likewise relatively low over the sample period. The key exception to this was Site 2 in May, at 75%.

These results were despite the limited riparian vegetation at some of the headwater sites, where erosion (and hence decreased clarity) might have been expected. However, the grassed margins likely helped to reduce this. Sampling was also not undertaken when there had been recent rain, which likely reduced inputs of sediment from surface runoff. These results were consistent with Christchurch City Council monitoring over the last few years. However, turbidity at Wilsons Drain did exceed ANZECC (2000) guidelines for lowland rivers on a couple of occasions in 2018 (Marshall & Noakes, 2019). These were still a lot lower than turbidity levels recorded in the Avon River or Heathcote River in the same year (Marshall & Noakes, 2019). Low levels of suspended sediment are important for photosynthetic activity, macroinvertebrate behaviour and available habitat within the stream gravels (Nogaro et al., 2009).

4.1.5 Nutrients and Total Organic Carbon

While low across the rest of the sites monitored, dissolved reactive phosphorus (DRP) was consistently elevated at Site 1 in the upper Ōtūkaikino Creek. All four samples from this site exceeded the ANZECC (2000) trigger value for slightly disturbed lowland rivers, with the average DRP being four times greater than this trigger value. Site 1 also regularly exceeded the LWRP trigger value of 0.016 mg/L for 'spring-fed – plains – urban' waterways (Environment Canterbury, 2017). This pattern was not shown in the phosphorus in the sediment at Site 1. By Site 2, a couple of kilometres downstream, DRP had returned to low levels. Sites 2, 3 and 4 did slightly exceed the ANZECC (2000) guideline value for slightly disturbed lowland rivers in July 2019, though not to the same levels as Site 1.

One potential source of this was the aviaries run by Isaac Conservation Trust. Along with elevated levels of faecal contamination, birds are a common source of phosphorus inputs in some systems (Wetzel et al., 2009). Isaac Conservation Trust also operates two salmon farms upstream of Site 1. Fish production results in some nutrient loading due to excretion of ammonium, with nitrogen and phosphorus inputs from fish faeces, sediments and excess feed (James et al., 2018). However, nitrate-nitrogen and ammoniacal nitrogen levels in 2019 and 2020 do not appear to be similarly high to DRP levels at Site 1, though ammoniacal nitrogen was slightly elevated in the Ōtūkaikino Creek compared to the Waimakariri South Branch. In addition, previous investigations of the impacts of salmon farming have shown limited nutrient outputs compared to other sources (Taranger et al., 2014).

There was also the potential for some inputs of DRP from overland flow. Fertiliser application and dung patches are known sources of phosphorus to streams (Monaghan et al., 2008). However, during the time period sampled, the paddock bordering Site 1 did not appear to have been stocked with sheep, making this unlikely to be a key source of the elevated DRP. In addition, turbidity and suspended solid levels were low at Site 1 and comparable to the rest of the catchment, making it unlikely that this DRP was primarily entering bound to sediment off the surrounding land.

Christchurch City Council monitoring in 2018 indicated that DRP might be of concern in other parts of the catchment. Site 2 and Site 4 exceeded the guidelines levels on at least one occasion in 2018 (Marshall & Noakes, 2019). Wilsons Drain, which was not monitored during this project, was of particular concern as was recorded at 0.47 mg/L during a rain event in 2018 (Marshall & Noakes, 2019). This was nearly 30 times the guideline value for DRP. As this stream flows through part of Belfast, this was likely associated with stormwater and wastewater overflow. However, Site 4 has still had a significant decrease in DRP of 14% since 2008 (Marshall & Noakes, 2019). Elevated levels of DRP are of concern as they contribute to nuisance algal blooms and their associated effects on biota (Hobbie et al., 2017).

Nitrate-nitrogen was not noted as a parameter of concern in 2019 and 2020. ANZECC (2000) does not include guidelines for total organic carbon. Both ammoniacal nitrogen and nitrate-nitrogen were well below ANZECC (2000) trigger values for slightly disturbed lowland streams ecosystem protection at all sites. For ammoniacal nitrogen, average concentrations exceeded the ANZECC (2000) trigger values for lowland streams ecosystem protection in the three sites along the Ōtūkaikino Creek (Sites 1 – 3). However, all sites sampled were consistently below the ANZECC (2000) 95% level of protection for freshwater biota. For the assessment of nitrate-nitrogen, this study used Hickey (2013)'s updated ANZECC (2000) guidelines of 1.0 mg/L (99% protection) and 2.4 mg/L (95% protection). During the time period monitored, there were no exceedances of either trigger value.

Nitrogen was recorded as a parameter of interest in the catchment during 2018 by the Christchurch City Council. Nitrate-nitrite-nitrogen levels, not analysed during the 2019 - 2020 monitoring programme, exceeded ANZECC (2000) guidelines at Sites 2 and 4 on at least one occasion, with Wilsons Drain exceeding in all analysed samples (Marshall & Noakes, 2019). Since 2014, nitrate-nitrite-nitrogen levels at Site 2 have increased by 25%, while dissolved inorganic nitrate levels have increased by 23%. This temporal change can largely be attributed to high peaks in 2017 and 2018 levels associated with rain events. This shows that there are potential nutrient inputs to the upper Ōtūkaikino Creek, though the specific source of these warrants further investigation.

The four samples taken during the 2019 – 2020 monitoring programme did not reflect these concentrations. This may be due to that fact that the Christchurch City Council monitoring captured several rain events, whereas the 2019 – 2020 monitoring programme was timed to avoid rain events to provide a more consistent baseline for water quality.

4.1.6 Trace Elements

Acid soluble trace elements found exceeding guidelines were arsenic, chromium, copper and zinc. Acid soluble arsenic was typically below detection limits, though was slightly elevated in July. Sites 1 and 4 exceeded the ANZECC (2000) 99% trigger value of 0.8 µg/L. The highest acid soluble arsenic concentration was recorded as 6.7 µg/L at Site 1 in July. All sites were under the ANZECC 95% trigger value for lowland steams ecosystem protection. Given the relative locations, the arsenic had potentially been introduced through the use of pesticides or insecticides containing arsenic (Chung et al., 2014). Acid soluble chromium was also below detection limits at most sites, though at 1.1 µg/L in the July 2019 sample, Site 1 exceeded the ANZECC (2000) 95% trigger value for lowland streams ecosystem protection of 1.0 µg/L. The source of this elevated chromium was unclear in this study.

Acid soluble copper was generally low across the catchment. The ANZECC 99% trigger value for lowland steams ecosystem protection was exceeded twice in the May 2019 samples, both along the Waimakariri South Branch: 1.2 µg/L at Site 5 and 2 µg/L at Site 6. The acid soluble copper concentrations recorded at Site 6 in May also exceeded the ANZECC (2000) 90% trigger value for lowland steams ecosystem protection, though remained below the 80% trigger value of 2.5 µg/L. Site 5 was located within a golf course. Maintenance of the turf grass generally involves frequent addition of pesticides and fertilisers, some of which may contain copper (Winter & Dillon, 2005). In addition, Site 5 was located downstream of pastoral land use, as was Site 6, which can also utilise pesticides.

Acid soluble zinc exceeded the ANZECC 99% trigger value for lowland steams ecosystem protection at all sites during May. The highest acid soluble zinc concentration (10.368 µg/L) was at Site 3 in May 2019, which also exceeded the ANZECC 95% trigger value for lowland steams ecosystem protection. The higher levels of acid soluble zinc at Site 3 were characteristic of stormwater inputs. These likely entered through runoff from the nearby resort buildings.

Acid soluble boron concentrations were higher at all sites in May 2019 compared to July 2019. Boron can be sourced from anthropogenic inputs like detergents and soap powders (Pennisi et al., 2006). As the upper catchment is predominantly pasture, this was likely showing a groundwater signature.

Consequently, lower levels in winter indicated dilution of groundwater inputs by another source. This was supported by increased flows at all sites except for Site 4 in July compared to May.

The acid soluble fraction gives the total concentration of the trace element, while the dissolved fraction indicates which trace elements are more mobile and more toxic (Morillo et al., 2004). For trace elements above detection limits, acid soluble boron showed the highest degree of mobility, with the disolved fraction consistently making up 90% to 100% of the total concentration. Zinc tended to have the lowest mobility, with the dissolved fraction making up 6% to 31% of the total concentration. The low mobility of acid soluble zinc was a good sign, especially for Site 3, as the elevated total concentrations were predominantly not in the more toxic state.

4.1.7 Sediment

Sediment concentrations of phosphorus are not included in the ANZECC (2000) guidelines. However, phosphorus concentrations were similar to the 2017 concentrations at each site (Figure 4.1). In both 2017 and 2019, phosphorus concentrations in sediment were highest at Site 7 in the upper Waimakariri South Branch, likely linked to the pastoral land use. This site had limited riparian margins in most places, as well as gaps in fencing, making it easier for phosphorus to enter from the surrounding land bound to sediment.

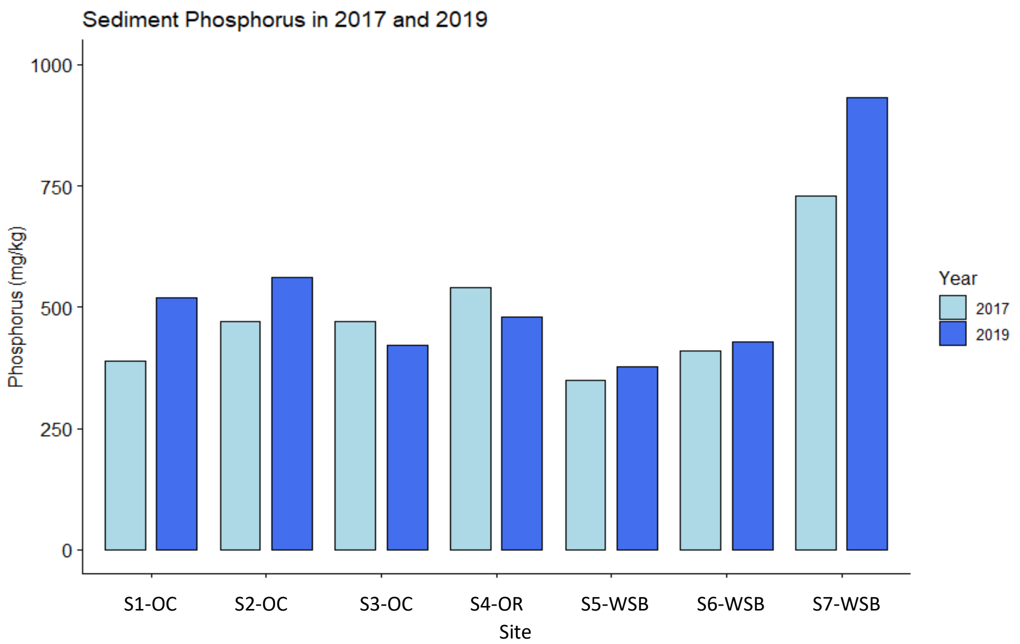

Figure 4.1: Phosphorus concentrations in sediment between 2017 (Boffa Miskell, 2017) and 2019. Site 6 uses the instream, not marginal, phosphorus concentration to provide more temporally comparable methodology.

The grain size composition of the sediment samples varied between sites. Site 6 instream and Site 3 sediment were dominated by gravel/sand, while Site 6 marginal sediment and Site 7 had a higher proportion of silt/clay. Grain size composition was of interest as metal contaminants more readily bind to small particles, with the increased surface area allowing for better attachment (Maslennikova et al., 2012). This results in metal contamination in sediment typically being recorded at higher levels at locations where the sediment has high levels of silt and clay (Boffa Miskell Limited, 2017).

This might, in part, help to explain the large differences in trace element concentrations observed at Site 6 between marginal and instream sediment. Site 6 marginal sediment and Site 7 contained elevated levels of almost all trace elements analysed, especially copper and zinc, compared to Site 6 instream sediment, as well as to the rest of the catchment. Given Sites 6 and 7 were located within land utilised for sheep and dairy support, with no obvious sources of these heightened metal concentrations (apart from a shed just upstream of Site 6), these levels seemed unusual. Acid soluble trace element levels were not elevated at either site in the May or July samples of 2019. Sediment trace element concentrations from Boffa Miskell Limited (2017) showed a slight increase in lead and copper concentrations at Site 6 in 2017, with copper levels exceeding the ANZECC (2000) ISQG-low guidelines. However, these were still well below the levels recorded in 2019. Grain size composition in 2017 at Sites 6 and 7 were shifted towards larger grain sizes compared to 2019. Alongside the influence of differing grain size compositions, there is the possibility of sporadic inputs of metal contaminants in the upper Waimakariri South Branch, though this was not reflected in the dissolved trace element levels of the May or July sampling runs. This may warrant further investigation.

Aside from the elevated results at Site 6 margins and Site 7, total metal levels in sediment were generally low at all sites. Cadmium exceeded ANZECC (2000) ISQG-low values at all sites, varying between 1.6 and 4.5 mg/kg. These also exceeded the background sediment concentrations of 0.19 mg/kg cadmium measured in the catchment in 2006 (Environment Canterbury, 2006). Slow accumulation of cadmium from historic phosphorus fertiliser use was likely a key source of this (Loganathan et al., 2003). Part of this might also be natural variation, as cadmium accumulation can vary over different soil types (Auckland Council, 2015). Lead was above ANZECC (2000) ISQG-low levels at Site 2. This may be a legacy issue from historic use of leaded paint and fuel (Pearson et al., 2010). Arsenic concentrations were low at all sites except for Site 7. It was recorded at 26.9 mg/L, which exceeded the ANZECC (2000) ISQG-low limit. This was still below the ISQG-high value for arsenic concentrations in sediment. Arsenic has historically been used in pesticides, so this may also be a legacy effect from past land use (Saldaña-Robles et al., 2018).

4.2 Pollution Sources and Habitat Limitation

The results of the 2019 – 2020 monitoring programme identify several key pollution sources and habitat limitations in the Ōtūkaikino River catchment. These can be used to prioritise areas for remediation and improved management.

4.2.1 Habitat Limitation

One overall area of concern for the Ōtūkaikino River was the distinct longitudinal changes in riparian habitat across the catchment. These general trends were explored in Chapter 3. The spatial and temporal variation in riparian composition is an important consideration for identifying vulnerable parts of the catchment to contaminant input.

These spatial trends in riparian habitat provide important indications of parts of the catchment that might be more prone to contaminant inputs from erosion and surface runoff. Riparian buffers primarily help shield the aquatic environment from nearby land use, performing various functions like filtering surface runoff, stabilising banks and regulating instream temperature (in particular via shading), while also providing habitat and food for biota (Collins et al., 2013; Greenwood et al., 2012).

This is not to say that riparian buffers solve all water quality concerns. Poorly designed and managed buffer zones can still contribute to contaminant inputs, especially where there are direct inputs that bypass the buffer like drains and stormwater pipes (Collins et al., 2013; Storey et al., 2017). In addition, until these plants become fully established, improvements in water quality and macroinvertebrate community health may be slow to occur (Collins et al., 2013). There can be other factors, such as available populations for recruitment, which can also limit the effectiveness of riparian buffers. However, in the case of catchments like the Ōtūkaikino River, any planted buffer is generally better than none.

The fact that much of the headwaters, including around many springheads, was relatively unplanted is therefore an area of concern for management in the catchment. Headwater reaches are a common target internationally for riparian improvements. This is due to increased buffer effectiveness and the requirement to safeguard the larger tributaries downstream (Lovell & Sullivan, 2006; Smiley Jr et al., 2011).

Some of the potential effects of this vulnerability were visible in the results of the 2019 – 2020 monitoring programme. The highest levels of fine sediment (75% at Site 1 in May 2019 and 40% at

Site 7 in October 2019) were from headwater sites. Macroinvertebrate communities in the headwaters were typically shifted towards more pollution tolerant taxa, as shown in MCI and SQMCI scores. Two sites in the upper reaches of the Waimakariri South Branch showed very elevated levels of trace elements in sediment. Stream temperature was generally elevated at headwater sites compared to the middle reaches. Macrophyte growth regularly reached more than 75% at several headwater sites. Dissolved oxygen levels dropped to about 55% at two headwater sites in January 2020.

These results further support the importance of well managed riparian buffers in the Ōtūkaikino River catchment. Large shade trees, planted on north banks, can help reduce stream temperature. This in turn helps increase dissolved oxygen levels, as well as reduce unwanted macrophyte growth. The addition of staggered sediment filtering plants like carexes complement these well. They intercept surface runoff from surrounding land use, helping reduce fine sediment and contaminant inputs.

4.2.2 Stormwater and Urbanisation

A second key pollutant source was the urbanisation mentioned to be occurring in the catchment. Over the last few decades, there has been some shift in land use, though this has been more apparent in recent years. Between 1996 and 2018, urban area in the Ōtūkaikino River catchment grew by 1.91 km^2, a 3.1% growth (Figure 4.2) (Landcare Research, 2020). Much of this was converted from grassland (Landcare Research, 2020). A key location of this growth was the construction of Clearwater Resort, in the middle reaches of the catchment. This construction also led to a slight increase of water bodies in the catchment through the creation of several lakes.

With the steady population growth in the Ōtūkaikino River catchment, the impacts of urban-related pressures are only likely to increase. As shown in Figure 4.3, resident population grew by 1560 individuals between 1996 and 2013 (Statistics New Zealand 2013). While most areas of the catchment have experienced a population increase during this time, this population growth has been particularly concentrated in the lower reaches of the catchment (Figure 4.4).

Several tributaries that flow into the lower reaches of the Ōtūkaikino River cross urban Belfast, including Wilson's Drain. As a result, it's also important to keep in mind the population growth and urban sprawl occurring in Belfast, which is predicted to reach 15,000 residents by 2041 (Christchurch City Council, 2010). This is almost double its 2013 population (Christchurch City Council, 2010).

Some of the impacts of urbanisation are already apparent in the catchment, both in the 2019 – 2020 monitoring programme and in previous monitoring of the catchment. Sites 3 was located by minor urban inputs, with several stormwater pipes located near the sampling reach that take water off the nearby resort building. Elevated zinc, compared to other sites and to ANZECC (2000) guidelines for lowland steams ecosystem protection, was recorded at Site 3 in May 2019. At Site 2, which receives runoff from the nearby scout camp, elevated levels of copper and lead were recorded in the sediment. These elevated metals could potentially have severe impacts on macroinvertebrate communities (Beasley & Kneale, 2002).

With these effects detectable in the upper and middle catchment, which have minor urban influences, the effects in the lower catchment are likely to be greater. Wilsons Drain, in the lower catchment, is already heavily impacted by the surrounding urban land use, with high levels of nutrients and *E. coli*, along with occasional spikes in dissolved zinc and copper (Marshall & Noakes, 2019). Its upper reaches are within an urbanised part of Belfast, with at least one consent to discharge stormwater into its headwaters (Environment Canterbury, 2008).

There is also evidence that rain events are likely to have large impacts on urban areas in the catchment, including around Site 2 in 2017 (Margetts & Marshall, 2018). For the sites monitored in 2019 and 2020, Site 3, with several stormwater inputs from the nearby resort, is predicted to show elevated levels of urban-related contaminants during rain events. However, this was not investigated in this study due to the absence of suitable rainfall. This also highlights the importance of monitoring water quality in the lower catchment, where population growth is concentrated, though this was outside the scope of this study.

There have been ongoing changes in the catchment to help reduce the impacts of urbanisation in the Ōtūkaikino River catchment. Major municipal sewage discharges were removed from the catchment in 2006, resulting in significant decreases in turbidity and dissolved phosphorus (Wilks & Meredith, 2009). Some consents for stormwater discharge to the catchment also have measures in place to reduce their contaminant inputs, including requiring vegetated banks to reduce erosion and developing retention areas for high rainfall events (Environment Canterbury, 2008).

Water quality impacts will therefore be important to keep in mind during continued urban development in this area. The Long-Term Plan for the Belfast township already acknowledges the need for best practice approaches for stormwater management (Christchurch City Council, 2010). Current objectives for Belfast include the creation of treatment and retention basins for stormwater, as well as the establishment of buffer zones around waterways and increased support for low-impact urban design (Christchurch City Council, 2010).

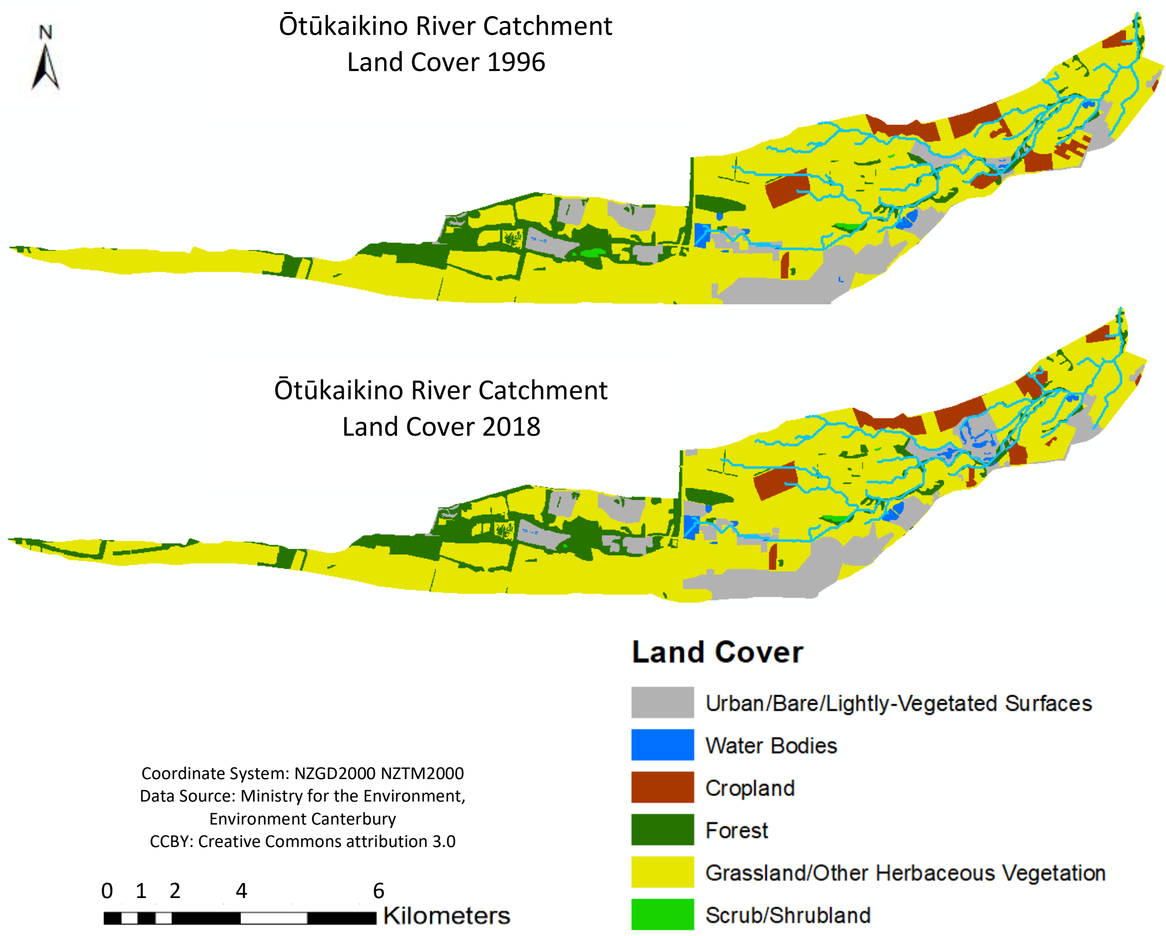

Figure 4.2: Land cover change in the Ōtūkaikino River catchment between 1996 and 2018. Categories follow those used by Landcare Research. Further explanation of these is included in Appendix 3.

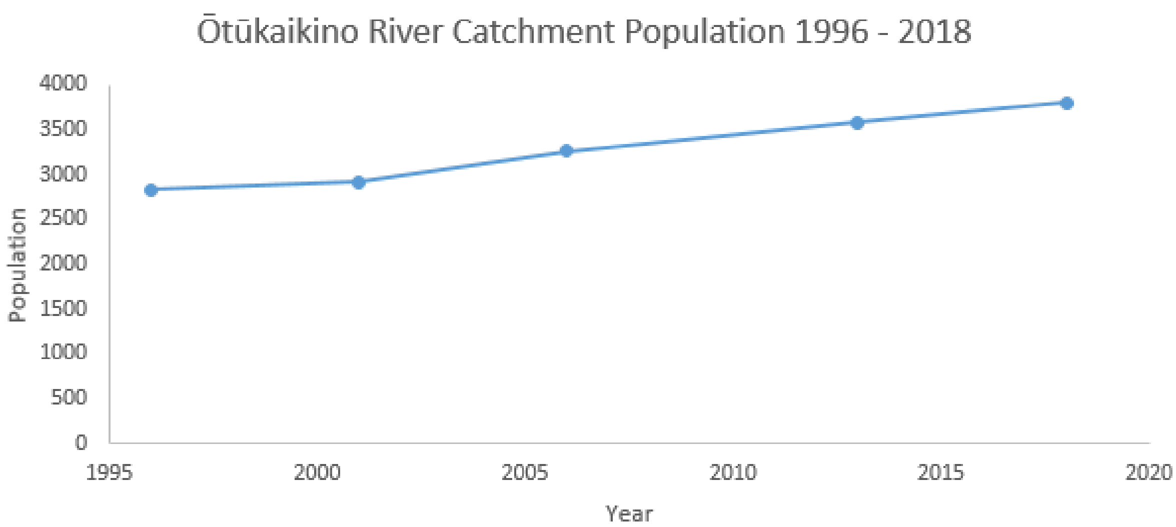

Figure 4.3: Population change in the Ōtūkaikino River catchment between 1996 and 2018.

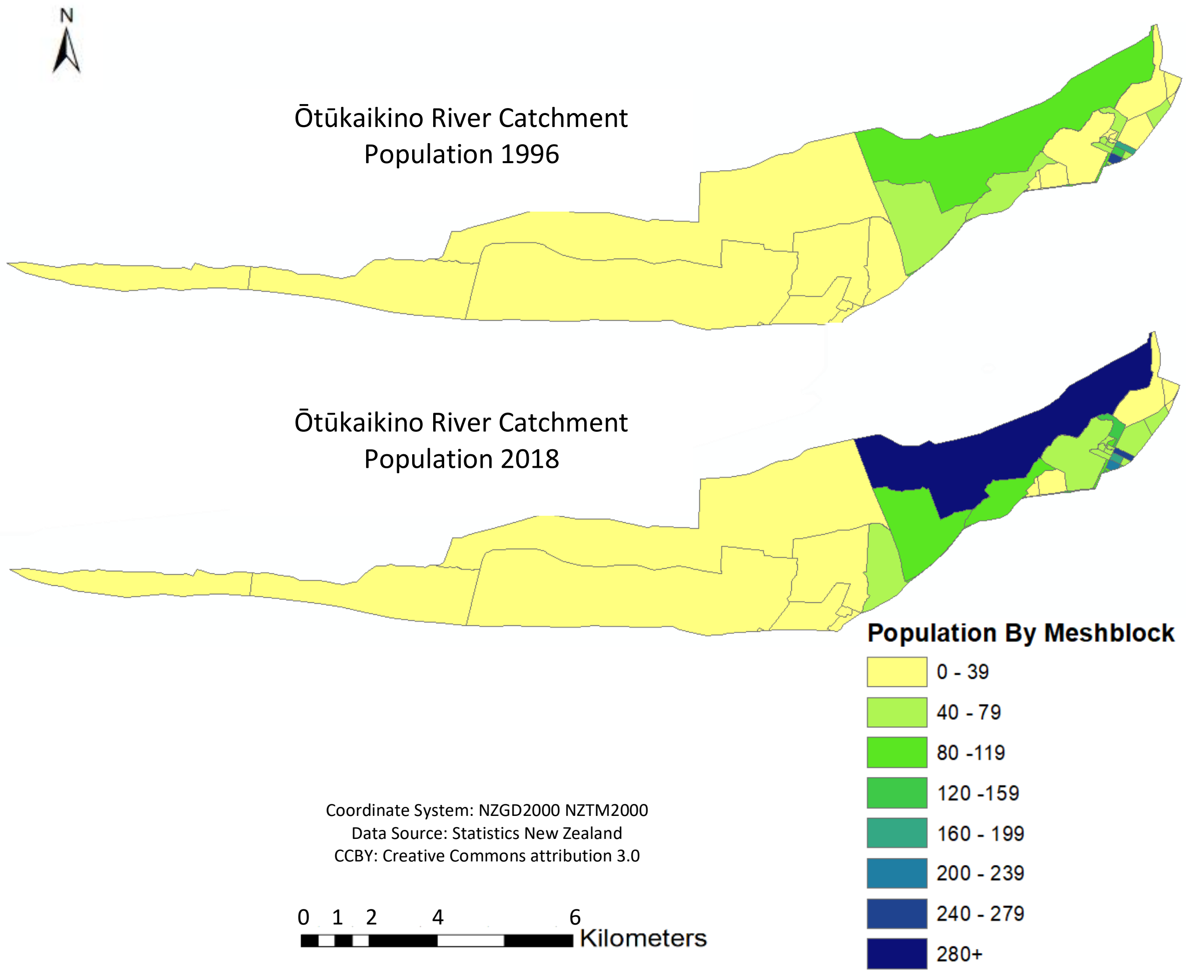

Figure 4.4: Resident population in the Ōtūkaikino River catchment between 1996 and 2018 at a meshblock scale.

4.2.3 Aviaries

As previously mentioned, the aviaries operated by Isaac Conservation Trust appear to be a potential source of pollution in the catchment. The cantilevered design is a likely source of faecal contamination to the Ōtūkaikino Creek, particularly at Site 1. In addition, these aviaries are a potential source of the elevated DRP. However, the levels of both contaminants appear to drop substantially before Site 2. This suggests that the faecal contamination from the aviaries is very unlikely to be negatively impacting recreational use in reaches further downstream.

4.2.4 Sediment Contamination in the Waimakariri South Branch

During this study, there were no consistent longitudinal trends of accumulation or dilution in trace element concentrations between sites. Instead, elevated trace elements of concern seemed to be localised to specific sites. Of interest was the high trace elements in sediment recorded at Sites 6 and 7, in the upper Waimakariri River.

While there was some indication of a difference in grain size composition contributing to the levels recorded, given the differences noted between marginal and instream sediment at Site 6, this was probably not the only contributing factor to the elevated metal concentrations. There was also evidence of temporal differences in trace element levels in sediment at Sites 6 and 7. In 2017, copper concentrations in sediment exceeded the ANZECC (2000) ISQG-low value (Boffa Miskell Limited, 2017). Lead concentrations in sediment were also slightly elevated, though were below the ANZECC (2000) ISQG-low value. Copper and lead levels in sediment were comparatively low at Site 7 that year (Boffa Miskell Limited, 2017). In comparison, the copper and lead concentrations in sediment were much higher at Sites 6 and 7 in 2019 compared to 2017. This indicates that the contaminant source might have been present before 2017 but that the effects have either accumulated since then or that the contributions of this pollutant source have increased over time. This warrants further investigation.

4.3 Spatial and Temporal Changes in Macroinvertebrate Communities

Previous work expressed concern over the potential decline in the aquatic macroinvertebrate health of the Ōtūkaikino River catchment (Boffa Miskell Limited, 2017). These concerns were primarily due to the potential absence of a taxa, *Zelandobius*, from the nine sites monitored in 2017. The 2017 report also highlighted the potential for declines in other aquatic macroinvertebrate taxa in the catchment. A key objective of this study was to identify any spatial and temporal trends in macroinvertebrate community health to examine whether this potential loss of a taxa may be an indicator of wider degradation.

4.3.1 Spatial Variation in Macroinvertebrate Communities

There was clear evidence for spatial variation in macroinvertebrate community composition within the Ōtūkaikino River catchment during 2019.

For the 10 sites sampled in 2019, taxonomic richness was between 20 and 29. Both Waimakariri South Branch and Ōtūkaikino Creek showed longitudinal increases, with less taxonomic diversity in the headwater sites. However, Sites 4 and 9, located downstream of the confluence of the two main tributaries, showed a slight decrease in richness. Taxonomic richness in both pastoral and urban catchments can vary greatly. The 2019 results were comparable to the upper range from previous studies conducted in similar land uses (Collier, 1995; Niyogi et al., 2007; Townsend et al., 1997).

The number of EPT taxa varied slightly between sites, reaching 14 at Site 3, though didn't show any clear longitudinal pattern or relationship with MCI or SQMCI scores. Some researchers argue against the use of EPT taxa as a pollutant indicator as the score does not reflect potential shifts from more sensitive *Ephemeroptera* to more tolerant *Tricoptera*, thereby skewing the metric (Clements & Kiffney, 1994). For the Ōtūkaikino River catchment sites sampled in 2019, the differences in the number of EPT taxa were more influenced by the presence or absence taxa found in low numbers rather than larger changes in macroinvertebrate community composition.

The most common taxon recorded in the catchment was typically *Pycnocentria* caddisflies, which were found at all sites. Also common were *Potamopyrgus* snails, *Deleatidium* mayflies and *Ostracoda* crustaceans*. Diptera* (true flies) were more common in the upper reaches, especially in the Waimakariri South Branch, compared to the mid-reaches. The low MCI scores of these *Diptera* taxa reflect their tolerance to pollution (Winterbourn et al., 2006). Their high numbers are therefore likely a reflection of the poorer water quality within this area, particularly the high levels of metal contamination in the sediment at Sites 6 and 7, as well as the higher proportion of the finest fraction in the sediment.

Four *Zelandobius* individuals were found in 2019: two in the Waimakariri South Branch, and two in the Ōtūkaikino Creek. At such low abundances, stoneflies are functionally not able to play a role as collectors/gatherers in the catchment. However, their continued presence in 2019 does indicate that it is possible for them to survive in the current conditions at some locations. They were also able to tolerate the high levels of metal contamination recorded in sediment at Site 7, in the Waimakariri South Branch, suggesting that metal contamination might not be a key limiting feature of *Zelandobius* presence in the catchment.

Less than 20 *Deleatidium* mayflies were recorded at each of the Waimakariri South Branch headwater sites (Sites 7 and 10) and none downstream at Site 6. Previous studies have concluded that *Deleatidium* are very sensitive to chronic metal contamination (Hickey & Vickers, 1992). Sites 6 and 7 recorded high levels of metal contamination in sediment, which would provide long-term stress for the taxa and hence a potential reason for their low numbers or absence at these sites.

The most diverse order was *Tricoptera*, with 13 taxa recorded in 2019, followed by *Diptera* with 11 taxa. This was primarily recorded as *Pycnocentria*, though *Pycnocentrodes* and *Hudsonema* were also common at some sites. This is similar to various other New Zealand studies, where *Diptera* and *Tricoptera* are often the most diverse orders (Suren & McMurtrie, 2005).

Of potential interest were the high number of the spiral-cased caddisfly, *Helicopsyche*, at Site 5. These are among the most sensitive of New Zealand's aquatic macroinvertebrate taxa, with an MCI score of 10. Site 5 had high habitat and water quality, as well as a substrate dominated by larger pebbles and cobbles. These conditions have been previously found to support high *Helicopsyche* abundance (Collier & Winterbourn, 2000).

MCI and SQMCI varied across the catchment in 2019. MCI scores were consistently lower in the headwater sites and increased down the catchment, with the highest MCI scores generally recorded around the middle of the catchment. This longitudinal pattern was consistent with MCI scores across previous years. The only exception to this was Site 8, located upstream of Site 2 in the Ōtūkaikino Creek. This had a good MCI score and an excellent SQMCI score. When the four rounds of invertebrate sampling in the Ōtūkaikino (2008, 2012, 2017 and 2019) were aggregated, there was a significant difference between the repeated sites in the catchment ($F_{6,18}$=6.58, P<0.001) (Figure 4.5).

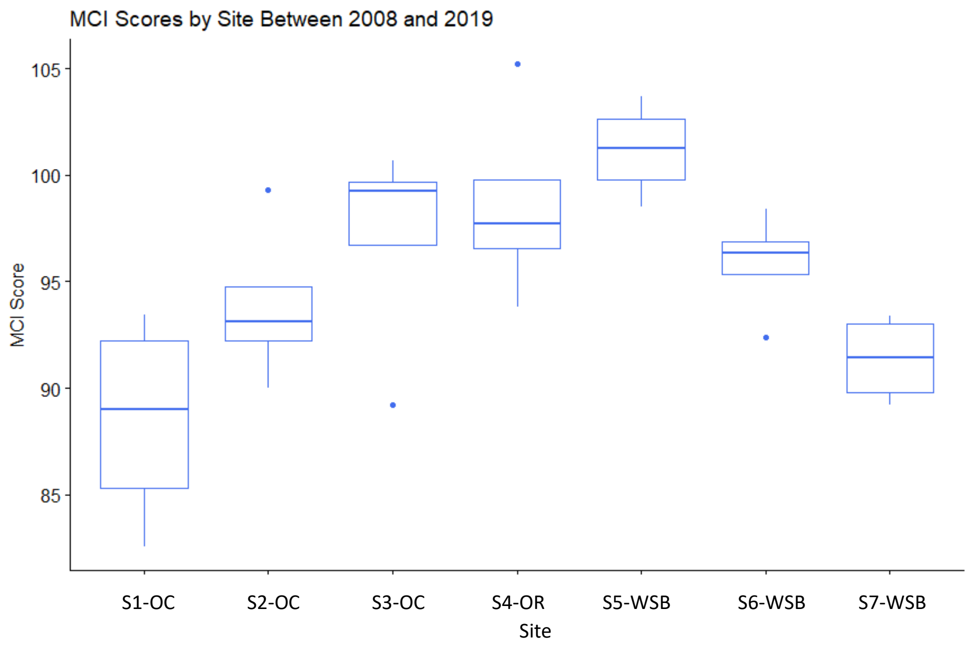

Figure 4.5: MCI scores across 7 sites, with four repeat samples (2008, 2012, 2017, and 2019) aggregated ($F_{6,18}$=6.58, P<0.001).

Often, a decline in biotic indices downstream might be expected due to factors like higher levels of fine sediment and intensifying land use (Niyogi et al., 2007). This was not the case in the Ōtūkaikino

River catchment. The middle reaches, as well as the area around Site 8, had thick riparian buffers compared to the upper reaches, which likely helped to moderate some of the effects of land use. These results are in line with some previous studies that conclude that, even with the contribution of contaminants from upstream land use, forested riparian areas can still lead to improved macroinvertebrate community health indices (Niyogi et al., 2007). The middle reaches also lacked some of the contaminant inputs noted in the upper reaches, including high levels of metal contamination at some sites, along with low levels of dissolved oxygen and high proportions of the finest sediment fraction at others.

Though both look at community composition, there were some differences in the scores given by MCI and SQMCI metrics (Figure 4.6). All sites scored as fair or good according to their MCI scores. The SQMCI scores showed more variation. Sites 6, 7 and 10, located in the upper reaches of the Waimakariri South Branch, had poor SQMCI scores compared to fair MCI scores, indicative of "probable severe pollution" (Stark & Maxted, 2007). Site 2 moved from a fair MCI score to a good SQMCI score. Sites 5, 8 and 9, with good MCI scores, had excellent scores for SQMCI. Sites 1, 3 and 4 were unchanged. Given the way that the two indices are designed, this difference is not surprising. MCI scores are only a measure of the presence or absence of taxa, whereas SQMCI use ranked abundances to provide a more detailed analysis of the community composition (Stark & Maxted, 2007). This means that the occurrence of rare taxa will have a greater impact on MCI scores compared to SQMCI scores. The poor SQMCI scores in the Waimakariri South Branch upper reaches reflect the greater proportion of pollution tolerant *Diptera* and *Crustacea* taxa that dominated these sites. Site 5, which had the maximum SQMCI score recorded of the 10 sites, was dominated by the pollution sensitive *Helicopsyche*. Sites 5, 8 and 9 still had some pollution tolerant taxa present, such as *Oligochaeta* worms and *Ostracoda* crustaceans, but these were at low abundances.

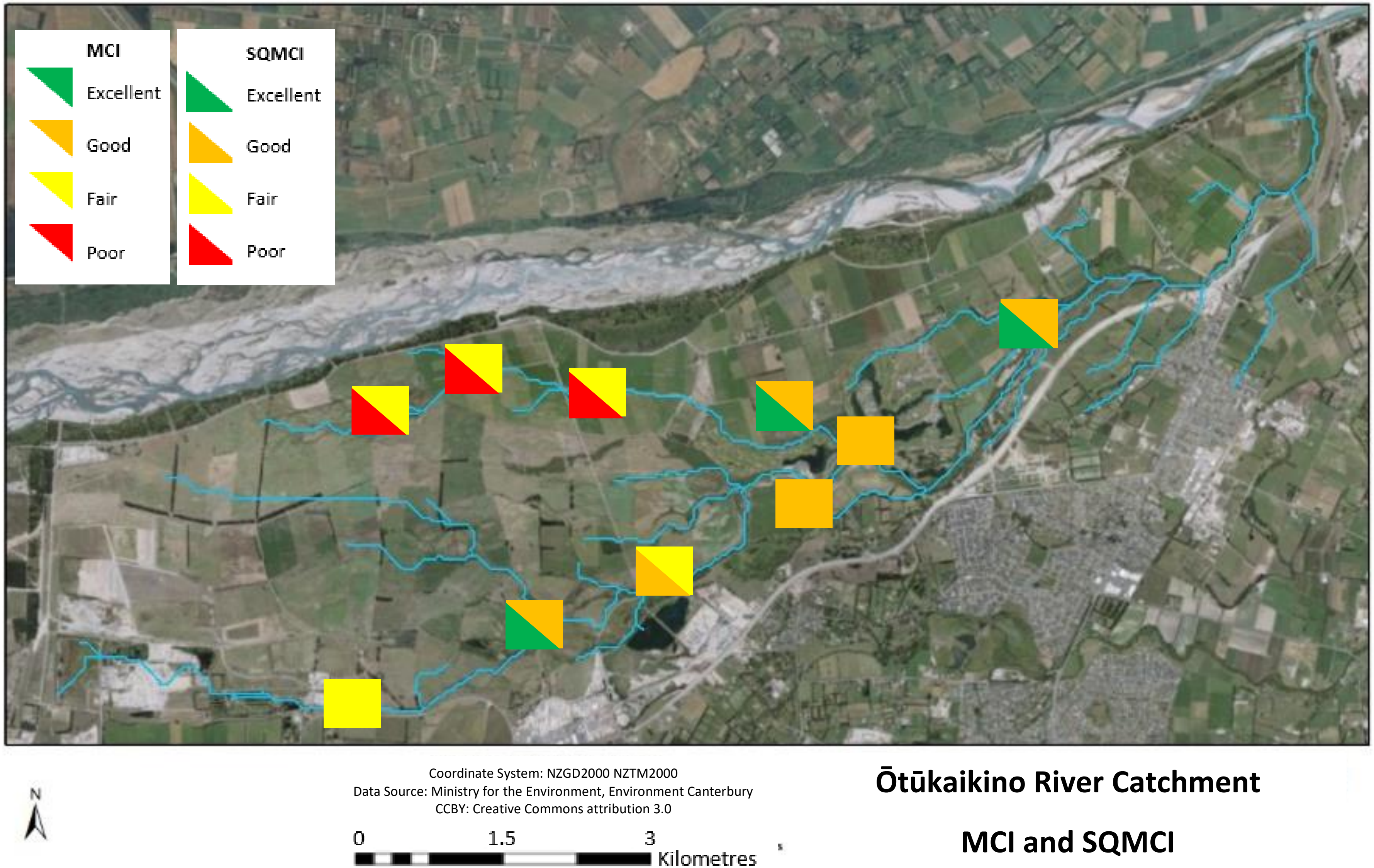

Figure 4.6: MCI and SQMCI scores at 10 sites in the Ōtūkaikino River catchment in July 2019.

4.3.2 Temporal Variation in Macroinvertebrate Communities

Overall taxonomic abundance appears to have declined since 2008: 67 taxa in 2008, 58 taxa in 2012, 50 taxa in 2017 and 42 taxa in 2019 (Boffa Miskell Limited, 2017). However, there are several potentially confounding variables here. It is important to note that the earlier studies analysed to a different taxonomic resolution (with the 2008 and 2012 studies analysing to species level for taxa such as *Hydrobiosis*). The 2008 and 2012 studies also included a general *Hydrobiosis* spp category, alongside more specific taxa, for any that they could not identify down to species level, such as for early instars.

The area sampled in 2008 and 2012 was three times greater than in 2017 and 2019. Biotic indices for this study compensated for that by using averages of the 2008 and 2012 studies, which gave an equivalent area. However, following species-area curves, the greater area originally sampled in 2008 and 2012 means that rarer taxa were more likely to be detected (Samuel et al., 2000). Sampling in 2019 was conducted at a different time of year (July compared to March), which some studies have shown to effect recorded abundance (Stark & Phillips, 2009). This likely impacted overall taxonomic abundance, as well as EPT taxonomic richness and MCI, so apparent trends might not be the case.

There was a clear visual change at several sites in macroinvertebrate community composition between the 2017 (Boffa Miskell Limited, 2017) and 2019 studies (Figure 4.7). In the 2017 study, Site 6 (Waimakariri South Branch) was dominated by *Deleatidium* and *Pycnocentria*. However, in the 2019 study, *Ostracoda* now dominated (44%). At Site 7 (Waimakariri South Branch) in the 2017 study, 22% of the community was composed of *Pycnocentria*. In comparison, in the 2019 study, *Ostracoda* dominated Site 7 (36%), followed by *Orthocladiinae* (17%) and *Pycnocentria* (17%). *Ostracoda* and *Orthocladiinae* are pollution tolerant taxa, so their increased dominance in 2019 at Sites 6 and 7 contributed to the poor SQMCI score compared to the good and excellent QMCI scores, respectively, in the 2017 study. Site 5 (middle reaches along the Waimakariri South Branch) in the 2017 study was dominated by *Potamopyrgus*, followed by *Pycnocentria*. In the 2019 study, this had shifted to be dominated by *Deleatidium*, *Helicopsyche* and *Pycnocentria*. The dominance of these pollution sensitive taxa contributed to the excellent SQMCI score in the 2019 study compared to the good QMCI score in the 2017 study. Though different biotic indices, SQMCI and QMCI tend to respond similarly to macroinvertebrate community change, hence their comparison here (Stark & Maxted, 2007).

There were 10 taxa reported in the 2017 study that were not found in the 2019 study. Most of these taxa, including the *Chironomus* midge, were previously noted at low numbers, so their absence from the samples in 2019 might have been by chance, rather than indicative of wider declines. More intensive sampling would be required to determine this.

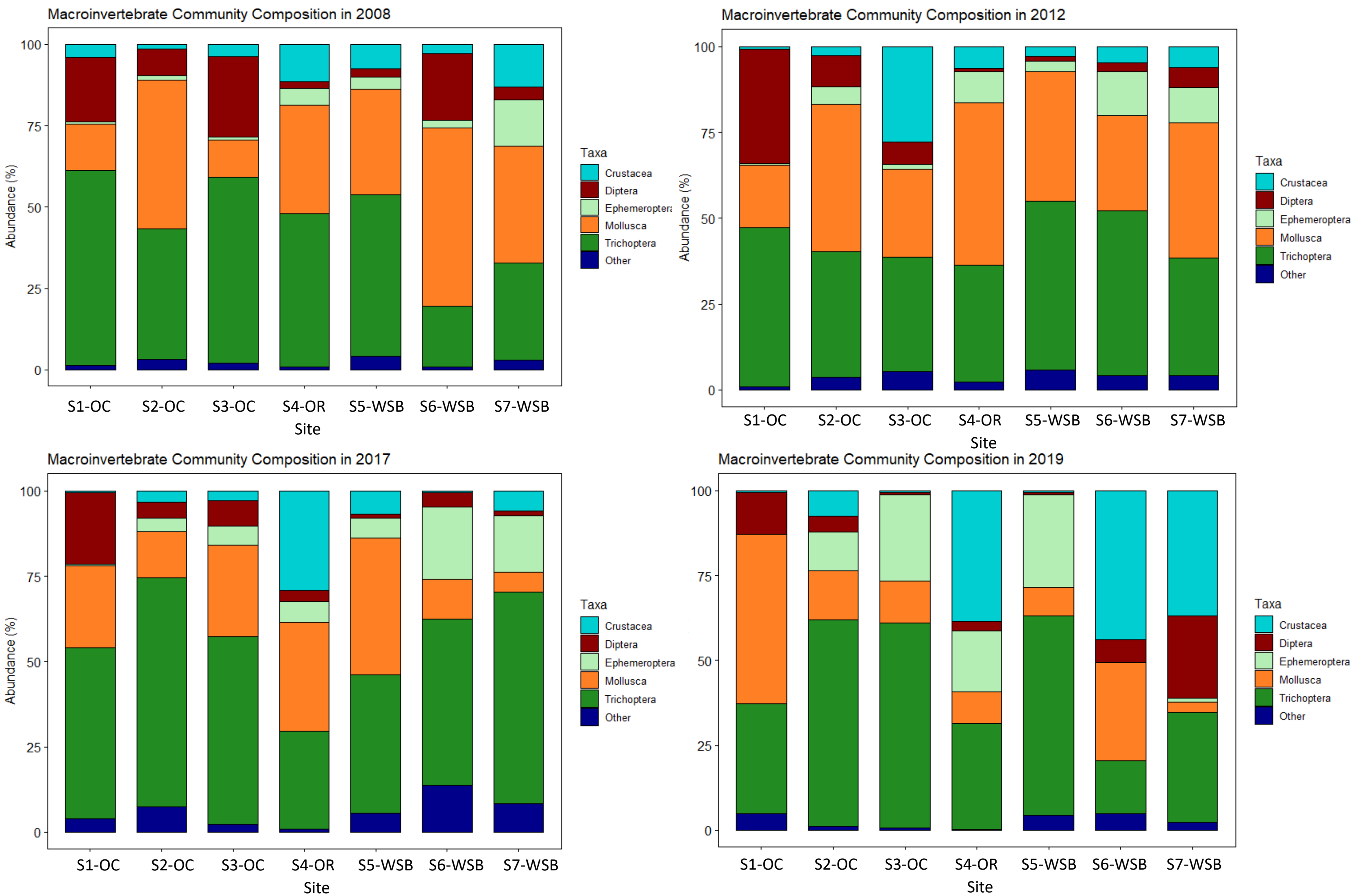

Figure 4.7: Macroinvertebrate community composition in 2008 (EOS, 2008), 2012 (EOS, 2012), 2017 (Boffa Miskell, 2017) and 2019 of 7 sites in the Ōtūkaikino River catchment. "Other" included *Acarina, Coleoptera, Collembola, Hemiptera, Hirudinea, Odonta, Platyhelminthes* and *Plecoptera.*

Site 7, where one *Zelandobius* stonefly was recorded in 2012 (EOS Ecology, 2012), was located less than 700 m south of the Waimakariri River. A number of *Plecoptera* taxa, including *Zelandobius*, are known to be present in the Waimakariri River (Gray & Harding, 2010). This presents the possibility that this individual in 2012 was just a lost colonist from the Waimakariri River and was not an indication of a stable *Zelandobius* population at that time. However, in 2019, *Zelandobius* individuals were recorded at both Site 2 and Site 3, which are located some distance away and on a different tributary of the river, the Ōtūkaikino Creek. This distribution indicates that *Zelandobius* are more widely present in the catchment than previous sampling had recorded.

The low abundances recorded means that there is also no statistical evidence of any temporal trends in *Zelandobius* abundance in the catchment, especially with the lack of robust data prior to 2008. However, given the general declines in EPT taxa across Christchurch, as shown by surveys in the 1980s and 1990s, a reduction in *Zelandobius* abundance over time would not come as a surprise (Suren & McMurtrie, 2005).

Their absence in other parts of the catchment which might be suitable, such as around Site 5, could be due to a variety of site preferences. *Zelandobius* are thought to be restricted to areas with lower temperatures and high dissolved oxygen (Quinn et al., 1994). However, they were still present at Site 7, in the upper Waimakariri South Branch, which recorded low level of dissolved oxygen in the January 2020 sample. A wide variety of other factors may be important for their presence, such as fine sediment and available oviposition habitat (Collier & Winterbourn, 2000; Storey et al., 2017). It is also possible that sampling effort happened not to detect them even if they were present, or that colonists have yet to reach that part of the catchment. This highlights the importance of linking riparian habitat to encourage colonisation.

NMDS ordination was conducted twice to compare 2017 and 2019 macroinvertebrate community compositions as well as all four timesteps (2008, 2012, 2017, and 2019). Both NMDS ordinations gave good representations of the actual community dissimilarities, with low two-dimensional stress values.

Though they appear visually separate, NDMS ordination of the 2017 and 2019 community compositions indicated that there were no statistically significant changes between these years (pseudo-F = 0.866, p = 0.492) (Figure 4.8). This was supported by the ANOSIM results (ANOSIM R = -0.01166, p = 0.454). NDMS ordination of the four sampled years, however, showed a small but statistically significant changes in the macroinvertebrate community at the sampled sites during this time (pseudo-F = 1.857, p = 0.011) (Figure 4.9). ANOSIM results supported this (ANOSIM R = 0.1582, P = 0.007).

SIMPER was then run on the 2008 – 2019 dataset. It indicated that any community differences were due to differing abundances of certain species instead of variation in presence/absence of taxa (Appendix 2). The most influential taxa tended to be the cased caddisfly *Pycnocetria* and the native snail *Potamopyrgus*, accounting for 21.1% and 14.4% of the variation in macroinvertebrate community composition between 2008 and 2019. These two taxa were similarly important for variation in macroinvertebrate community composition between 2017 and 2019. *Pycnocentrodes*, *Deleatidium* and *Ostracoda* abundance changes were also important between some years. These results support those in Figure 4.7.

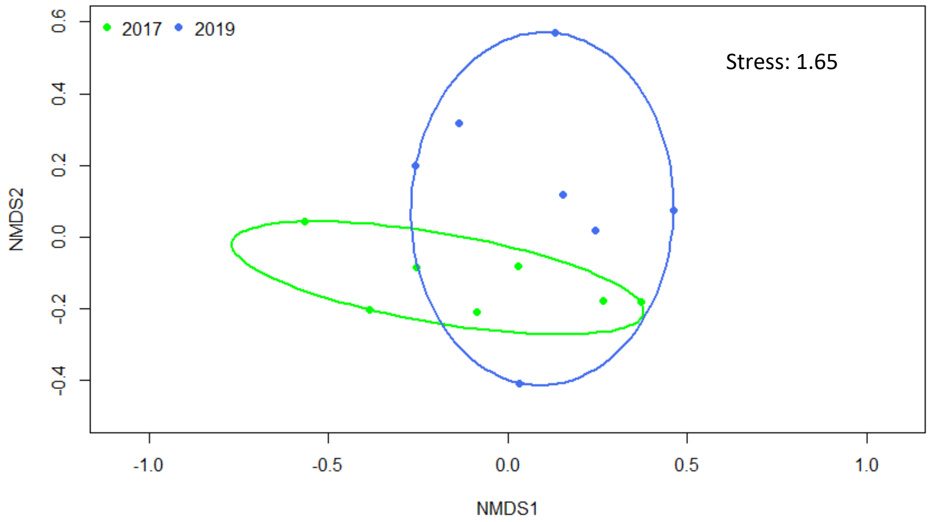

Figure 4.8: Non-metric multidimensional scaling (NDMS) of taxonomic community relationships between 2017 and 2019.

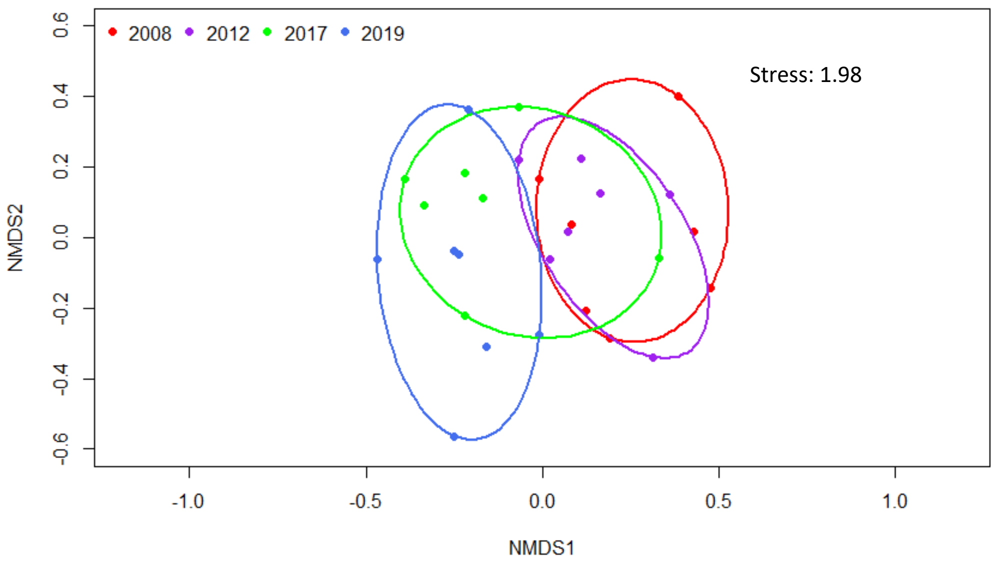

Figure 4.9: Non-metric multidimensional scaling (NDMS) of taxonomic community relationships between 2008, 2012, 2017 and 2019.

There were a few key differences in MCI scores between the 2019 study and previous sampling. In the 2017 study, all sites had a fair MCI score. In the 2019 study, Sites 3, 4, and 5, in the middle reaches were now at a good MCI score. Site 4, for example, changed from an MCI score of 93.8 in the 2017 study to an MCI score of 105.2 in the 2019 study. This change was due to the presence of two additional *Tricoptera* taxa (*Olinga* and *Polyplectropus*) and the decrease in *Diptera* taxa (2 taxa compared to 5 in the 2017 study).

When the sites were aggregated, there was evidence of significant change in MCI scores in the catchment through time ($F_{3,18}$=3.429, P=0.039) (Figure 4.10). Visually, there was a decrease in MCI scores in the catchment between the 2008 and 2017 studies. Overall, MCI scores in the 2019 study (from the 7 repeated sites) appeared higher than in the previous years of monitoring. However, it is still important to keep in mind the confounding variables of taxonomic resolution, area sampled and time of year when attempting to draw any temporal comparisons.

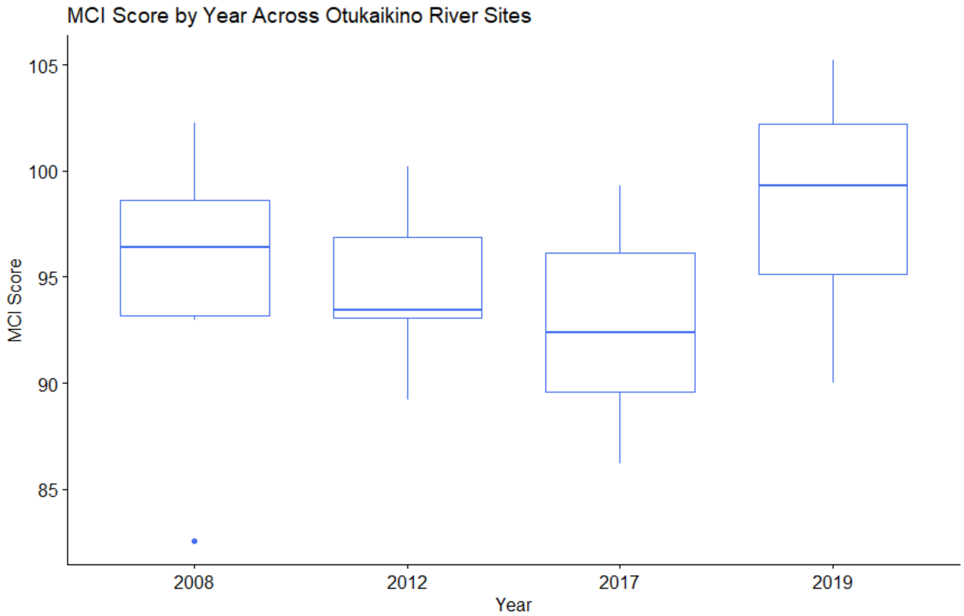

Figure 4.10: MCI scores between 2008 and 2019, with the 7 repeated sites aggregated ($F_{3,18}$=3.429, P=0.039).

4.4 Potential Options for Remediation

The results from the 2019 – 2020 monitoring programme present several implications for management in the Ōtūkaikino River catchment. Many of the areas of concern identified in this study seemed to be localised to specific sites, with limited evidence for longitudinal accumulation or dilution of contaminants. This means that potential options for remediation need to address specific point source inputs across the whole catchment as opposed to applying general management approaches to the entire catchment.

In the upper reaches of the Waimakariri South Branch, the identified areas of concern are the existing riparian buffers, gaps in fencing, and high metal contamination of sediment.

Much of the headwaters upstream of Site 7 had no or minimal riparian vegetation. Some sections, such as around Site 7, had some riparian vegetation on one or both banks, but these were often composed of exotic species such as willows. This is a key area to target for improved riparian vegetation. Shade trees and staggered sediment filtering plants will be particularly important. The use of indigenous species would provide increased canopy cover year-round, compared to many exotic tree species. This in turn would help to shade out unwanted macrophytes and improve dissolved oxygen levels, which were parameters of concern at this site during the 2019 to 2020 study. In addition, indigenous tree species would provide more appropriate allochthonous inputs through leaf litter compared to exotic species. This will be important for the improvement of macroinvertebrate communities over these reaches.

At Sites 6 and 7, high levels of metal contamination were found in the sediment. The use of staggered sediment filtering plants would be beneficial for treating surface runoff from surrounding land use. This might help reduce inputs of both fine sediment and metal contamination, both of which can impact macroinvertebrate communities. However, without clear indication of the primary source of this contamination, aside from its potential link to the sediment grain size, this may have limited effectiveness.

There were also some gaps in fencing, as well as fencing that could have improved setbacks, in the upper reaches. The ephemeral headwaters should be assessed to determine what sections, if any, would benefit from fencing. This would support improvements in the riparian buffer, including the reduction of fine sediment cover in the upper reaches. The upper reaches of the Waimakariri South Branch are close to potential sources of colonisers from the Waimakariri River. Improved fencing and riparian buffers in this area would therefore have the greatest effects compared to other parts of the catchment.

Reaches around Site 6, downstream of Site 7 and along the Waimakariri South Branch, show similar management implications. Stream temperature is a key concern here, with the warmest temperature recorded at this site during the monitoring programme. Instream macrophytes were not recorded at high levels during the days of sampling. However, extensive drain clearance was noted during the "Catchment Walks" in April 2019, with macrophyte cover high prior to this. Consequently, improved riparian planting through this reach will be important to reduce the need for regular mechanical excavation, which can have severe impacts on fish communities, as was observed in April 2019. This part of the catchment is currently mostly grassed, with some carexes and occasional native species like toitoi. As previously noted, planting is currently underway for this area. Around this reach, having shade plants on the north bank and staggered sediment filtering plants would have even more positive effects. Having a few layers of staggered sediment filtering plants will help intercept some of the runoff from surrounding land use. This could include occasional high levels of faecal contamination, such as in the January 2020 sample at Site 6. The reaches around Site 6 would also benefit from improved fencing setbacks to support this riparian buffer. This appears to already be in progress. Like Site 7, Site 6 had a poor SQMCI score in the 2019 survey. Improved riparian buffers could therefore support the macroinvertebrate community through decreasing water temperatures, alongside serving as important habitat and food sources (Greenwood et al., 2012).

The upper reaches of the Ōtūkaikino Creek show some similar areas of concern to the upper reaches of the Waimakariri South Branch. Low dissolved oxygen was a concern at Site 1 in January 2020. At the same time, the stream had low flow and high emergent macrophyte cover. Stream flow appeared to naturally fluctuate seasonally at the site, with a downstream reach somewhat ephemeral. Increased canopy cover through improved riparian vegetation would help shade out these unwanted macrophytes and cool stream temperatures, consequently helping to improve dissolved oxygen levels. This could help support the presence of more pollution sensitive macroinvertebrate taxa like *Deleatidium*, which were present at Site 1 in low numbers in 2019 compared to other sites.

Another area of concern for Site 1 is the high faecal contamination from the aviaries. This will require ongoing monitoring and will also be important to keep in mind if any changes are made in bird numbers in the programme. Elevated DRP levels were also highlighted as a concern at Site 1. If this is coming off the surrounding land, improved riparian vegetation would help reduce this. If this is more due to the upstream aviaries, it will also be important to carefully consider any potential increases in bird numbers. While the high levels of faecal contamination are probably not having a large impact on the macroinvertebrate community at the site, it is still an important parameter to note when considering the overall state of the Ōtūkaikino River catchment.

The reach upstream of Site 2 lacked riparian vegetation other than a mown grass bank, though it transitioned further downstream to a mix of native and exotic vegetation. Submergent macrophyte cover increased over the sample period, reaching 60% in the January 2020 sample. Site 2 also had high fine sediment cover in the May 2019 sample. Extending this riparian buffer, particularly via shade trees on the north bank, would help to reduce impacts from surrounding land use. This would provide increased shade while still allowing for recreational access and ease of maintenance. However, the fine sediment, as well as other elevated parameters, might be linked to land use upstream, so some collaboration with multiple landowners and further investigations into likely sources might be required here.

Across the middle and lower reaches of the catchment, a key area of concern is the ongoing effects of urbanisation. Around Site 3, potential stormwater inputs were present in 2019. For example, acid soluble zinc levels were elevated compared to other sites sampled in the catchment and above ANZECC (2000) guidelines for lowland streams ecosystem protection. If these effects are already being seen around Site 3, with a small number of buildings, then similar effects are likely to be visible in the lower catchment from stormwater inputs, where urban density is higher. While the lower catchment was outside the scope of this thesis, whole catchment management is an important consideration. With the increasing population in the surrounding area, it will be important to build in safeguards to protect water quality and biotic health in the lower Ōtūkaikino River. Some of these have already been discussed regarding the growing urban population in Belfast (Christchurch City Council, 2010). Turbidity, nutrient, *E. coli* and dissolved zinc and copper have all been noted in other studies to be contaminants of concern in a lower urban drain within the catchment (Marshall & Noakes, 2019). Ongoing monitoring and good management practices in the lower catchment should therefore be encouraged.

5 Conclusions

5.1 Summary of Main Findings

For this study, a 12-month monitoring programme was created for the Ōtūkaikino River catchment. The primary goal was to determine potential sources of pollution and habitat limitation in the Ōtūkaikino River catchment related to the potential decline of a sensitive macroinvertebrate taxa. Within this, spatial and temporal changes in a range of physical, chemical and biological parameters were investigated.

Riparian and canopy cover varied across the catchment. They were typically highest in the middle reaches, though there were some other areas of thick vegetation, such as around Site 8 in the Ōtūkaikino Creek. Periphyton cover was generally highest in the upper reaches, reaching more than 80% on several occasions. It was still present at most other sites in the catchment.

There was some spatial variation in substrate composition between sites. Sites 2, 6 and 7 in the upper catchment were dominated by finer substrates like silt and gravel. Sites 3 and 4 in the mid catchment tended to have a higher portion of coarser substrates such as cobbles.

Stream flow increased downstream. The highest flows were consistently at Site 4, in the middle reaches. The reach just downstream of the Ōtūkaikino headwater site was ephemeral. Flow disappeared through here during the last round of sampling.

Percentage saturation of dissolved oxygen exceeded 100% on three of the five samples at Site 6 in the upper Waimakariri South Branch. Dissolved oxygen was also very low at some sites, reaching about 55% saturation at two headwater sites in the January 2020 survey. Temporal variation in dissolved oxygen was generally greater in the upper reaches than the middle reaches. Stream temperatures were generally cool at all sites. The January 2020 survey did show an increase in temperature at all sites, though none exceeded relevant guidelines.

pH was circa neutral at all sites. Conductivity was also somewhat consistent spatially and temporally, though was typically slightly lower in the Waimakariri South Brach compared to the Ōtūkaikino Creek. The conductivity was low relative to other rivers in Christchurch. Turbidity and total suspended solids were low at all sites and showed no clear spatial or temporal trends.

Faecal coliforms were consistently elevated at Site 1 in the upper Ōtūkaikino Creek compared to the rest of the catchment, as well as Site 7 in the upper Waimakariri South Branch to a lesser extent. Faecal coliforms were elevated in the January 2020 survey compared to previous rounds, particularly at Site 6 in the Waimakariri South Branch. However, *E. coli* levels were generally low at all sites. The key exception to this was Site 6 in the January 2020 sample, which recorded elevated faecal coliforms and *E. coli* compared to previous sampling.

Nitrate-nitrogen concentrations were low across the catchment compared to ANZECC (2000) guidelines for lowland streams ecosystem protection. TOC (total organic carbon) was consistently recorded at similarly low levels. Ammoniacal nitrogen was elevated at Sites 1 – 3 compared to other sites monitored in the catchment. DRP (dissolved reactive phosphorus) was also consistently elevated at Site 1 compared to other sites. All DRP samples at Site 1 exceeded the relevant ANZECC (2000) guidelines for lowland streams ecosystem protection.

Trace elements were generally low in the Ōtūkaikino River catchment. The acid soluble trace elements found exceeding ANZECC (2000) guidelines for lowland streams ecosystem protection were arsenic, chromium, copper and zinc. In sediment, metal concentrations were generally low, except for two headwater sites (Sites 6 and 7) of the Waimakariri South Branch. These two sites recorded high levels of most parameters analysed. Lead exceeded ANZECC (2000)-low guidelines at both, while copper exceeded ANZECC (2000)-high guidelines. These levels of metal contamination were generally only present in the marginal sediment of Site 6, not the instream sediment.

A large degree of spatial and temporal variation was clear from the 10 sites sampled for macroinvertebrates. Macroinvertebrate abundance fluctuated greatly between sites, with the highest at Site 2. However, the taxonomic richness, and its EPT component, remained relatively consistent across sites.

MCI scores were highest near the confluence of the Waimakariri South Branch and the main stem of the Ōtūkaikino River, in general increasing downstream towards this point in the middle reaches. This longitudinal pattern was consistent with MCI scores across previous years. Sites were all rated as fair or good. SQMCI scores showed a similar pattern, though with greater spatial variation in scores. Site scores ranged from poor (upper Waimakariri South Branch) to excellent (middle reaches). The key exception to this pattern was the high biotic indices scores from a covenant site in the upper part of Ōtūkaikino Creek.

In total, 42 taxa were found in the 2019 survey. The most diverse group was *Tricoptera*, followed by *Diptera.* The cased caddisfly *Pycnocentria* was a dominating taxon at many sites, though

Potamopurgus snails and *Deleatidium* mayflies were also typically in high numbers. The spiral cased *Helicopsyche*, with a low tolerance to pollution, was found at 6 of the sites sampled in 2019.

Zelandobius stoneflies were present in the catchment in the 2019 sampling round: one at Site 2, one at Site 3 (both in the Ōtūkaikino Creek), and two at Site 7 in the Waimakariri South Branch. Stoneflies had not previously been recorded at Site 2. In comparison to the 2019 survey, no *Zelandobius* were found in 2017. This indicates that the catchment is still able to support populations of stoneflies. It is also likely that stoneflies are present in other parts of the catchment, particularly with the nearby Waimakariri River serving as a source of colonists.

Temporal variability was also present in macroinvertebrate communities between the 2008 and 2019 studies. MCI scores visually appeared to be higher in the 2019 study compared to the 2017 study. However, a variety of confounding variables make drawing accurate temporal comparisons difficult with MCI scores. While NDMS ordination did not indicate that the sampled 2017 and 2019 macroinvertebrate communities differed significantly, there were small, but significant differences across the 2008, 2012, 2017 and 2019 studies. This was primarily related to changing abundances of taxa like *Pycnocentria*, as opposed to the occurrence of rare taxa.

Through this study, several potential sources of pollution have been identified. The aviaries in the upper Ōtūkaikino Creek produce high levels of faecal contamination, and likely DRP. Some urbanisation effects were visible, including increased dissolved zinc around Site 3 and increased lead concentrations in sediment at Site 2 (both in the Ōtūkaikino Creek). Previous work had indicated that these effects are more pronounced in the lower catchment (Marshall & Noakes, 2019). The increasing urban population is likely to further increase these effects if not well managed. The source of the elevated metals in the upper Waimakariri South Branch was not clear from this study. However, a potential link was seen with the higher fine fraction of sediment compared to other sites sampled, which can increase metal absorption. While not all of these are areas of concern for stoneflies, it is important to have a wider view of stream health in the catchment beyond the interactions with a single taxon.

Overall, most potential issues of concern in the catchment seem to be localised impacts, rather than overall spatial trends of contaminant accumulation or dilution. One consequence of this is that localised habitat is important in the Ōtūkaikino River catchment. Habitat limitation is a key concern in the catchment, with the Ōtūkaikino River catchment showing large spatial variation in the quality and spread of its riparian buffers. The headwater reaches had little to no riparian buffer, interspersed with occasional gorse and willow. Riparian buffer quality increased downstream to areas where native plantings had been prioritised. The middle catchment generally had thick established riparian vegetation. This spatial variation is important to keep in mind when interpreting both water quality and macroinvertebrate community metrics due to the various roles that well-managed riparian buffers play, specifically through shade trees and sediment filtering plants. This site-specific variation in water and sediment quality, as opposed to general catchment-wide trends, means that riparian margins need to be designed and managed for particular purposes, such as shading out unwanted macrophytes or reducing sediment inputs to water.

5.2 Potential Options for Remediation

Out of this study, there are several key implications for management in the Ōtūkaikino River catchment.

Size and quality of riparian buffers varied greatly across the catchment. In particular, the headwater reaches should be targeted for improvements with shade plants and staggered sediment filtering plants. This will help reduce fine sediment inputs, reduce stream temperatures (in turn shading out unwanted macrophytes) and provide food and habitat for stream biota. However, until these plants become fully established, improvements in water quality and macroinvertebrate community health may be slow to occur (Collins et al., 2013). There can be other factors, such as available populations for recruitment, which can also limit the effectiveness of riparian buffers or cause a delay in the changes desired for the waterway. This is important to bear in mind when it comes to setting timeframes for desired outcomes within the catchment.

Faecal contamination was high compared to ANZECC (2000) guidelines for lowland streams ecosystem protection on occasion. This was particularly apparent at Sites 1 and 6, in the upper catchment. For Site 1, it is recommended that aviary managers consider how they can reduce faecal contamination contributions. For Site 6, improved riparian buffers should also help filter some of the faecal contamination entering through surface runoff.

The quality of stream fencing also varied across the headwater reaches. Some ephemeral reaches were open to stock. These should be assessed as to whether their flow is high enough that they warrant stock exclusion. Fencing setbacks were also narrow in other reaches. Improved setbacks will support the improvements in riparian vegetation.

Increasing urbanisation in the lower catchment is an area of concern. With an increasing urban population, it will be important to effectively manage stormwater inputs and other urban-related pressures. This might include approaches like silt traps.

5.3 Study Limitations

A key limitation of this study was the spatial and temporal intensity of sampling. This was primarily constrained by funding and time. Water quality was sampled four times, while sediment and macroinvertebrates were sampled once. This meant that diurnal variation in water quality was not observed, nor was the response to rain events. During the survey timeframe, no significant rain events occurred that would have been suitable for sampling. However, the sampling frequency that was used for this study gave a broad picture of stream health in the catchment while remaining feasible within the time available.

Due to the short-term nature of the project, the timing of invertebrate sampling was not optimal. Previous research has indicated that seasonality is unlikely to significantly affect macroinvertebrate community indices due to poorly synchronised life histories of aquatic invertebrates in New Zealand, except for a changing likelihood of flood disturbance (Stark & Phillips, 2009). Despite this, the 2019 macroinvertebrate results may have been more comparable with previous years sampling had it occurred at the same time of year.

The invertebrate sampling methods also excluded checking for live adults, such as through malaise trapping, which may have noted more rare taxa. This would have required an intensive monitoring programme over the summer months for best results, which did not align with the dates set for this study. A different skill set would also have been required regarding identifying terrestrial adults as opposed to the aquatic larvae. The single sample of benthic invertebrates from each site was instead used to provide a representation of stream biota present during the monitoring period.

5.4 Recommendations for Further Research

This study analysed water, habitat and sediment quality over the course of a year. Rain events were avoided as possible to reduce their effect on sample results. Previous monitoring by Christchurch City Council has shown large variation in water quality in the catchment during storm events (Margetts & Marshall, 2018). This is consistent with international literature that suggests that high flow events tend to be a key pathway for suspended sediment into waterways, along with the associated trace metals and nutrients (Bach et al., 2010; Horowitz, 2009). During the recorded weather events in 2017, Site 2 experienced large increases (compared to monthly water sampling) in total suspended solids, turbidity, dissolved reactive phosphorus and *E. coli* (Margetts & Marshall, 2018). In particular, *E. coli* levels at Site 2 reached 17,000 CFU/100mL during the first rain event recorded in 2017 (Margetts & Marshall, 2018). The effect of storm events, therefore, would be an important part of understanding water quality in the catchment.

From this study, the source of the high metal concentrations in sediment at Sites 6 and 7 in the Waimakariri South Branch is unclear. While this may be a consequence of the higher fine fraction at those sites, there could also be another contamination source. This might include sheep dips and seepage from dump sites. This will require further investigation.

Site characteristic data from the previous CREAS survey in the catchment (EOS Ecology, 2008a) was not available during this project. This dataset could provide a valuable comparison to current habitat conditions in the Ōtūkaikino River catchment to identify areas of improvement and decline and better understand factors that might be contributing to other changes in the state of the catchment. This might involve comparing changes at specific sites, such as those monitored for macroinvertebrates, or a full repeat survey.

Sites sampled focused on the upper and middle catchment, as these were theorised to have some of the rarer taxa and better habitat. The lower catchment has been identified as having areas of concern in previous studies (Marshall & Noakes, 2019). For a more comprehensive understanding of the catchment, these should be further investigated.

There are likely other locations within the catchment not targeted for sampling that could support populations of rarer taxa. For example, the presence of stoneflies in the two main tributaries in the 2019 sampling suggests they might be presence in other locations. Further investigations into stream biota in other parts of the catchment would allow management to identify and target these potential strongholds for further support.

While this study was prompted by the potential disappearance of *Zelandobius* stoneflies from the Ōtūkaikino River catchment, it is important that the presence (or lack thereof) of this taxon is not the central priority for targeting future management and research. Further investigation into changes in the abundance of a range of rare taxa might provide a more complete picture of temporal changes in the macroinvertebrate communities within the Ōtūkaikino River catchment.

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Appendix

Appendix 1: Site Field Sheets

Site		
Date/time		
Weather		
Water clarity/colour		
pH		
DO		
Temp		
Cond		
Land use		
Riparian vege %		
Riparian comp		
Canopy cover		
Wetted width		
Fine sediment		
Water depth		
Substrate % comp	Silt	
	Gravel	
	Pebbles	
	Sm cobbles	
	Lg cobbles	
Macrophyte %	Emergent	
	Submergent	
Periphyton %		
Flow		

Appendix 2: Raw Data

Table 6.1: Sampling conditions and in situ water measurements in 2019 and 2020.

Site	Date	Time	Weather	Water clarity	Water Colour	pH	DO mg/L	DO %	Temp	Cond
Site 1	24/05/19	10.54	Overcast, not raining	Clear	Colourless	6.72	10.0	90.8	11.0	83.4
Site 2	24/05/19	11.51	Overcast, not raining	Clear	Colourless	6.84	8.78	82.8	13.0	89.2
Site 3	24/05/19	17.23	Overcast, not raining	Clear	Colourless	6.97	8.42	79.1	12.9	84.2
Site 4	24/05/19	13.02	Overcast, not raining	Clear	Colourless	7.12	9.45	87.9	12.4	86.8
Site 5	24/05/19	16.49	Overcast, not raining	Clear	Colourless	7.01	8.14	78.0	13.7	75.1
Site 6	24/05/19	14.24	Overcast, not raining	Clear	Colourless	7.56	10.31	100.8	14.6	70.7
Site 7	24/05/19	15.32	Overcast, not raining	Clear	Colourless	7.29	9.02	83.5	12	65.5
Site 1	15/07/19	9:06	Clear, not raining	Clear	Colourless	7.29	9.98	88.1	9.2	86.6
Site 2	15/07/19	12:11	Clear, not raining	Clear	Colourless	7.31	10.09	93.1	11.2	87.1
Site 3	15/07/19	14:20	Overcast, not raining	Clear	Colourless	7.49	10.06	92.9	11.2	84.4
Site 4	12/07/19	12:35	Overcast, not raining	Clear	Colourless	7.42	9.75	87.7	10.4	87.5
Site 5	15/07/19	13:36	Clear, not raining	Clear	Colourless	7.33	9.81	92.2	11.9	73.1
Site 6	12/07/19	11:20	Clear, not raining	Clear	Colourless	7.77	10.79	98.0	11.1	68.5
Site 7	12/07/19	9:22	Clear, not raining	Clear	Colourless	7.51	7.86	67.0	8.2	65.3
Site 8	15/07/19	9:36	Clear, not raining	Clear	Colourless	7.25	9.66	87.6	10.5	86.8
Site 9	12/07/19	13:20	Overcast, not raining	Clear	Colourless	7.21	10.08	90.4	10.3	87.3
Site 10	12/07/19	10:24	Clear, not raining	Clear	Colourless	7.25	10.02	84.2	7.6	64.3
Site 1	30/07/19	9:10	Overcast, not raining	Clear	Colourless	6.64	10.23	94.7	11.2	80.6
Site 2	30/07/19	9:58	Overcast, not raining	Clear	Colourless	6.90	9.77	90.4	11.3	88.1
Site 3	30/07/19	11:33	Clear, not raining	Clear	Colourless	7.18	9.87	94.1	12.6	85.7
Site 4	30/07/19	12:58	Clear, not raining	Clear	Colourless	7.26	10.35	97.5	11.9	88.7
Site 5	30/07/19	10:58	Overcast, not raining	Clear	Colourless	6.96	10.00	93.2	11.6	77.6
Site 6	30/07/19	13:45	Clear, not raining	Clear	Colourless	7.32	11.45	110.6	12.9	71.4
Site 7	30/07/19	15.02	Clear, not raining	Clear	Colourless	7.15	10.34	98.6	12.3	64.5
Site 1	10/10/19	8:54	Overcast, not raining	Clear	Colourless	6.83	8.58	85.5	14.5	80.7

Site 2	10/10/19	9:47	Clear, not raining	Clear	Colourless	6.92	9.85	92.2	12.4	89.8
Site 3	10/10/19	11:24	Clear, not raining	Clear	Colourless	6.85	9.81	94.0	12.8	86.1
Site 4	10/10/19	12:56	Overcast, not raining	Clear	Colourless	6.98	9.43	91.5	13.4	86.4
Site 5	10/10/19	10:53	Clear, not raining	Clear	Colourless	6.82	9.9	93.8	12.3	72.6
Site 6	10/10/19	14:10	Overcast, not raining	Clear	Colourless	7.21	10.9	106.6	13.6	68.6
Site 7	10/10/19	15:08	Overcast, not raining	Clear	Colourless	7.16	8.88	90.3	15.4	65.1
Site 1	9/01/20	9:02	Clear, not raining	Clear	Colourless	7.14	5.77	57.7	15.4	81.5
Site 2	9/01/20	9:40	Clear, not raining	Clear	Colourless	6.95	8.53	86.0	15.7	85.6
Site 3	9/01/20	10:40	Clear, not raining	Clear	Colourless	7.02	8.54	84.6	15.0	81.9
Site 4	9/01/20	12:05	Clear, not raining	Clear	Colourless	7.15	8.77	89.4	16.3	83.5
Site 5	9/01/20	11;20	Clear, not raining	Clear	Colourless	7.06	9.03	91.2	15.9	71.8
Site 6	9/01/20	14:20	Clear, not raining	Clear	Colourless	7.29	9.55	101.9	18.4	89.2
Site 7	9/01/20	15:20	Clear, not raining	Clear	Colourless	6.96	5.57	55.2	14.9	63.8

Table 6.2: Land use and vegetation in 2019 and 2020.

Site	Date	Land use	Riparian % Cover	Riparian Composition	Canopy cover %	Macrophyte % Cover		Periphyton cover
						Submergent	Emergent	
Site 1	24/05/19	Sheep farm	5	Mostly grass, some carexes	0	5	15	95
Site 2	24/05/19	Scout camp - manicured lawn	30	Mature trees, open understory, grass	50	0	30	10
Site 3	24/05/19	Golf course	70	Willows on TRB, native planting TLB	50	0	20	30
Site 4	24/05/19	Dog park and reserve	100	Mostly mature trees, some ferns, thick understorey	70	0	0	40
Site 5	24/05/19	Manicured golf course (TRB), farm (TLB)	55	Few flax (TRB), mature trees with thick understorey (TLB)	90	5	5	25
Site 6	24/05/19	Beef farm	5	Fenced off 1-5m narrow zone. Carex, grass, rare flax and toitoi	0	10	0	15
Site 7	24/05/19	Sheep and cattle	50	Exotic deciduous trees (TLB), all grass/gorse (TRB)	40	0	95	0

Site 1	15/07/19	Sheep farm	5	Mostly grass, some carexes	0	5	15	90
Site 2	15/07/19	Scout camp - manicured lawn	30	Mature trees, open understory, grass	45	25	0	15
Site 3	15/07/19	Golf course/park	70	Willows on TRB, native planting TLB	40	0	15	25
Site 4	12/07/19	Dog park and reserve	100	Mostly mature trees, some ferns, thick understorey	60	0	0	30
Site 5	15/07/19	Manicured golf course (TRB), farm (TLB)	55	Few flax (TRB), mature trees with thick understorey (TLB)	90	0	0	15
Site 6	12/07/19	Beef farm	5	Fenced off 1-5m narrow zone. Carex, grass, rare flax and toitoi	0	0	0	90
Site 7	12/07/19	Sheep and cattle	50	Exotic deciduous trees (TLB), all grass/gorse (TRB)	40	0	60	0
Site 8	15/07/19	Sheep farm, within 400m covenant	10	Planted natives (carex, flax, cabbage trees, pittosporums etc) and willows	40	15	70	15
Site 9	12/07/19	Reserve/public area	90	Exotic with planted natives	25	15	0	10
Site 10	12/07/19	Sheep and cattle	5	Gorse and grasses with some willows	0	0	0	70
Site 1	30/07/19	Sheep farm	5	Mostly grass, some carexes	0	5	15	90
Site 2	30/07/19	Scout camp - manicured lawn	30	Mature trees, open understory, grass	45	25	0	15
Site 3	30/07/19	Golf course/park	70	Willows on TRB, native planting TLB	40	0	15	25
Site 4	30/07/19	Dog park and reserve	100	Mostly mature trees, some ferns, thick understorey	60	0	0	30
Site 5	30/07/19	Manicured golf course (TRB), farm (TLB)	55	Few flax (TRB), mature trees with thick understorey (TLB)	90	0	0	15
Site 6	30/07/19	Beef farm	5	Fenced off 1-5m narrow zone. Carex, grass, rare flax and toitoi	0	0	0	90
Site 7	30/07/19	Sheep and cattle	50	Exotic deciduous trees (TLB), all grass/gorse (TRB)	40	0	60	0
Site 1	10/10/19	Sheep farm	5	Mostly grass, some carexes	0	2	5	15
Site 2	10/10/19	Scout camp - manicured lawn	30	Mature trees, open understory, grass	10	45	2	10

Site 3	10/10/19	Golf course/park	70	Willows on TRB, native planting TLB	30	40	5	10
Site 4	10/10/19	Dog park and reserve	100	Mostly mature trees, some ferns, thick understorey	70	0	0	15
Site 5	10/10/19	Manicured golf course (TRB), farm (TLB)	55	Few flax (TRB), mature trees with thick understorey (TLB)	90	5	1	10
Site 6	10/10/19	Beef farm	5	Fenced off 1-5m narrow zone. Carex, grass, rare flax and toitoi	0	0	1	80
Site 7	10/10/19	Sheep and cattle	50	Exotic deciduous trees (TLB), all grass/gorse (TRB)	90	20	80	0
Site 1	9/01/20	Sheep farm	5	Mostly grass, some carexes	0	5	75	5
Site 2	9/01/20	Scout camp - manicured lawn	30	Mature trees, open understory, grass	15	60	0	25
Site 3	9/01/20	Golf course/park	70	Willows on TRB, native planting TLB	50	0	10	35
Site 4	9/01/20	Dog park and reserve	100	Mostly mature trees, some ferns, thick understorey	70	0	0	15
Site 5	9/01/20	Manicured golf course (TRB), farm (TLB)	55	Few flax (TRB), mature trees with thick understorey (TLB)	70	0	15	5
Site 6	9/01/20	Beef farm	5	Fenced off 1-5m narrow zone. Carex, grass, rare flax and toitoi	0	15	0	15
Site 7	9/01/20	Sheep and cattle	50	Exotic deciduous trees (TLB), all grass/gorse (TRB)	90	10	0	60

Table 6.3: Substrate and water quantity in 2019 and 2020.

Site	Date	Substrate composition					Substrate Index	Wetted width (m)	Water Depth		Flow (m^3/s)
		Silt	Gravel	Pebbles	Sm cobbles	Lg cobbles			Max	Av	
Site 1	24/05/19	5	35	40	20	1	4.81	1.3	0.16	0.11	0.019
Site 2	24/05/19	75	5	10	10	0	3.55	4.9	0.21	0.18	0.15
Site 3	24/05/19	0	0	50	50	0	5.50	4.0	0.82	0.62	0.30
Site 4	24/05/19	1	13	50	35	1	5.21	14.0	0.65	0.45	1.81
Site 5	24/05/19	1	5	78	15	1	5.09	3.8	0.27	0.21	0.32

Site 6	24/05/19	20	10	60	10	0	4.60	3.5	0.24	0.285	0.14
Site 7	24/05/19	0	0	50	50	0	5.50	5.0	0.15	0.15	0.015
Site 8	15/07/19	5	20	40	30	5	5.05	11.0			
Site 9	12/07/19	5	30	60	5	0	4.65	10.5			
Site 10	12/07/19	20	10	40	30	0	4.80	1.2			
Site 1	30/07/19	5	35	40	20	1	4.81	3.0	0.17	0.14	0.13
Site 2	30/07/19	15	10	40	35	0	4.95	6.2	0.29	0.20	0.57
Site 3	30/07/19	1	0	49	50	0	5.48	4.5	0.72	0.52	0.92
Site 4	30/07/19	5	14	50	35	1	5.37	14.0	0.68	0.44	1.57
Site 5	30/07/19	5	5	75	15	0	5.00	6.2	0.18	0.13	0.55
Site 6	30/07/19	10	15	65	10	0	4.75	4.4	0.22	0.14	0.21
Site 7	30/07/19	10	15	65	10	0	4.75	5.2	0.11	0.11	0.023
Site 1	10/10/19	0	20	35	40	5	5.25	3.0	0.15	0.094	0.093
Site 2	10/10/19	5	20	30	40	5	5.15	6.2	0.3	0.19	0.38
Site 3	10/10/19	10	20	60	10	0	4.70	4.0	0.82	0.48	0.91
Site 4	10/10/19	10	10	30	35	15	5.20	14.0	0.64	0.45	1.86
Site 5	10/10/19	5	20	65	10	0	4.80	5.8	0.22	0.18	0.847
Site 6	10/10/19	10	40	45	5	0	4.45	4.6	0.26	0.27	0.38
Site 7	10/10/19	40	40	5	5	0	3.35	4.2	0.08	0.08	0.035
Site 1	9/01/20	0	25	45	30	0	5.05	1.55	0.16	0.10	0.016
Site 2	9/01/20	30	15	30	20	5	4.50	6.05	0.30	0.22	0.32
Site 3	9/01/20	5	40	50	5	5	4.85	4.0	0.68	0.42	0.91
Site 4	9/01/20	5	15	30	35	15	5.25	14.4	0.73	0.48	1.85
Site 5	9/01/20	5	60	25	10	0	4.40	4.6	0.26	0.19	0.47
Site 6	9/01/20	5	15	35	40	5	5.20	4.5	0.32	0.17	0.15
Site 7	9/01/20	10	20	40	30	0	4.90	4.3	0.19	0.19	0.013

Table 6.4: Lab analysis of water quality parameters measured in 2019 and 2020, except for trace elements and sediment.

Site	Date	TSS (mg/L)	Faecal Coliforms (CFU/100mL)	*E. coli* (CFU/100mL)	NH_4-N (mg/L)	NO_3-N (mg/L)	DRP (mg/L)	Turbidity (NTU)	Total Carbon (mg/L)	Total Inorganic Carbon (mg/L)	Total Organic Carbon (mg/L)
Site 1	24/05/19	0.1	2267	0	0.014	0.26	0.043	0.10	5.49	4.85	0.65
Site 2	24/05/19	0.4	300	67	0.0053	0.14	0.004	0.07	5.78	5.28	0.50
Site 3	24/05/19	0.7	867	0	0.0094	0.15	0.004	0.06	5.25	4.91	0.33
Site 4	24/05/19	0.9	800	0	0.014	0.18	0.008	0.18	5.60	5.12	0.48
Site 5	24/05/19	1.3	967	0	0.013	0.20	0.007	0.14	4.65	4.40	0.25
Site 6	24/05/19	0.1	200	0	0.013	0.14	0.003	0.58	3.83	3.73	0.10
Site 7	24/05/19	1.1	1900	33	0.0053	0.0030	0.002	0.06	4.17	3.57	0.60
Site 1	30/07/19	0.2	3300	0	0.021	0.24	0.041	0.22	5.09	4.28	0.81
Site 2	30/07/19	0.6	467	133	0.0019	0.41	0.013	0.25	4.94	5.01	0.00
Site 3	30/07/19	0.8	300	200	0.0038	0.37	0.011	0.18	4.98	4.89	0.09
Site 4	30/07/19	0.3	400	100	0.0076	0.38	0.01	0.21	5.31	5.04	0.27
Site 5	30/07/19	2.0	200	0	0.0046	0.40	0.0052	0.25	4.11	4.13	0.00
Site 6	30/07/19	0.4	300	0	0.0031	0.28	0.0058	0.34	3.77	3.78	0.00
Site 7	30/07/19	0.1	633	167	0.0034	0.053	0.0046	0.16	3.69	3.51	0.19
Site 1	10/10/19	1.2	7600	67	0.093	0.085	0.037	0.24	7.13	5.93	1.20
Site 2	10/10/19	1.7	567	33	0.20	0.45	0.0054	0.12	6.81	5.90	0.91
Site 3	10/10/19	0.7	833	0	0.17	0.41	0.0037	0.15	6.87	5.92	0.96
Site 4	10/10/19	2.2	1633	133	0.010	0.32	0.0054	0.30	7.08	5.88	1.20
Site 5	10/10/19	1.8	900	33	0.037	0.25	0.0017	0.26	5.79	5.09	0.70
Site 6	10/10/19	0.8	533	133	0.044	0.097	0.0017	0.37	5.64	4.89	0.75
Site 7	10/10/19	2.6	3100	233	0.035	0.033	0.00067	0.22	5.91	4.61	1.30
Site 1	9/01/20	0.8	8450	50	0.0085	0.028	0.033	0.08	11.69	7.56	4.13
Site 2	9/01/20	1.5	3750	100	0.0052	0.20	0.0040	0.06	7.63	6.66	0.97
Site 3	9/01/20	1.2	1300	50	0.0024	0.20	0.0032	0.08	6.94	5.77	1.18
Site 4	9/01/20	1.2	4550	150	0.0052	0.18	0.0069	0.29	8.04	5.90	2.14
Site 5	9/01/20	0.6	6500	350	0.0057	0.15	0.0023	0.23	8.58	7.28	1.30
Site 6	9/01/20	0.2	7900	1400	0.011	0.095	0.0069	0.18	6.59	3.73	2.86
Site 7	9/01/20	0.8	600	0	0.00095	0.074	0.0030	0.28	6.18	4.83	1.36

Table 6.5: Acid soluble trace elements. Where results were below detection limits, <DL is used.

Site	Al	As (V)	B	Cd	Co	Cr (VI)	Cu	Fe	Mn	Mo	Ni	P	Pb	S	V	Zn
	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L
$S1_{May}$	10.2	<DL	28.1	<DL	<DL	<DL	<DL	13.1	0.9	1.2	<DL	69.2	<DL	130	8.7	5.1
$S1_{July}$	8.2	6.7	22.4	<DL	<DL	1.1	<DL	<DL	0.6	4.8	<DL	40	<DL	1930	10	<DL
$S2_{May}$	8.4	<DL	29.7	<DL	<DL	<DL	<DL	7.4	0.7	1.0	<DL	13.5	<DL	151	5.1	4.7
$S2_{July}$	11.9	<DL	18.1	<DL	<DL	<DL	<DL	<DL	1.5	1.9	<DL	<DL	<DL	2000	10	<DL
$S3_{May}$	6.5	<DL	27.9	<DL	<DL	<DL	<DL	8.4	0.7	1.3	<DL	8.15	<DL	172	<DL	10.4
$S3_{July}$	8.7	<DL	16.9	<DL	<DL	<DL	<DL	<DL	1.3	1	<DL	<DL	<DL	2000	<DL	<DL
$S4_{May}$	9.6	<DL	27.5	<DL	<DL	<DL	<DL	88.8	20.5	1.5	<DL	13.5	<DL	195	6.1	4.9
$S4_{July}$	10.8	6.3	16.3	<DL	<DL	<DL	<DL	<DL	18	1.1	<DL	<DL	<DL	1980	<DL	<DL
$S5_{May}$	13	<DL	28.2	<DL	<DL	<DL	1.2	21.5	1.1	1.2	<DL	15.7	<DL	232	<DL	3.6
$S5_{July}$	16.5	<DL	15.0	<DL	<DL	<DL	<DL	<DL	0.7	1.2	<DL	<DL	<DL	2140	10	<DL
$S6_{May}$	35.8	<DL	27.6	<DL	<DL	<DL	2	51.8	1.3	1.6	<DL	9.8	<DL	252	<DL	3.6
$S6_{July}$	13.3	<DL	11.6	<DL	<DL	<DL	<DL	<DL	0.1	<DL	<DL	<DL	<DL	2000	<DL	<DL
$S7_{May}$	5.8	<DL	26.5	<DL	<DL	<DL	<DL	8.0	0.4	1.1	<DL	10.7	<DL	274	<DL	4.8
$S7_{July}$	6.2	<DL	7.6	<DL	<DL	<DL	<DL	<DL	<DL	0.9	<DL	<DL	<DL	1900	10	<DL
ANZECC 99%	27	0.8	90	0.06		0.01	1		1200		8		1			2.4
ANZECC 95%	55	13	370	0.2		1	1.4		1900		11		3.4			8
ANZECC 90%	80	42	680	0.4		6	1.8		2500		13		5.6			15

Table 6.6: Dissolved trace elements from May and July 2019. Where results were below detection limits, <DL is used.

Site	Al	As (V)	B	Cd	Co	Cr (VI)	Cu	Fe	Mn	Mo	Ni	P	Pb	S	V	Zn
	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L
S1$_{May}$	6.3	<DL	28.0	<DL	<DL	<DL	<DL	8.8	0.5	1.1	<DL	63.4	<DL	139[1]	4.2	1.5
S1$_{July}$	6.1	<DL	20.1	<DL	<DL	<DL	<DL	<DL	0.3	2.5	<DL	40	<DL	1900	<DL	<DL
S2$_{May}$	7.0	<DL	29.3	<DL	<DL	<DL	<DL	3.6	0.7	1.0	<DL	13.2	<DL	151	<DL	0.9
S2$_{July}$	7.7	<DL	17.9	<DL	<DL	<DL	<DL	<DL	1.2	1.5	<DL	<DL	<DL	2000	<DL	<DL
S3$_{May}$	5.3	<DL	27.9	<DL	<DL	<DL	<DL	3.7	0.5	1.2	<DL	8.2	<DL	172	<DL	0.6
S3$_{July}$	7.0	<DL	16.5	<DL	<DL	<DL	<DL	<DL	1	<DL	<DL	<DL	<DL	2020	<DL	<DL
S4$_{May}$	6.5	<DL	27.5	<DL	<DL	<DL	<DL	58.7	19.2	1.1	<DL	13.5	<DL	195	<DL	0.7
S4$_{July}$	8.7	<DL	16.0	<DL	<DL	<DL	<DL	<DL	16.2	1.1	<DL	<DL	<DL	1950	<DL	<DL
S5$_{May}$	7.7	<DL	27.6	<DL	<DL	<DL	0.6	8.8	0.7	0.8	<DL	14.7	<DL	232	<DL	0.5
S5$_{July}$	11.4	<DL	14.5	<DL	<DL	<DL	<DL	<DL	0.7	1.0	<DL	<DL	<DL	2130	<DL	<DL
S6$_{May}$	15.0	<DL	27.0	<DL	<DL	<DL	1.1	17.7	0.8	1.4	<DL	7.9	<DL	252	<DL	0.5
S6$_{July}$	8.8	<DL	10.6	<DL	<DL	<DL	<DL	<DL	<DL	<DL	<DL	<DL	<DL	1990	<DL	<DL
S7$_{May}$	5.0	<DL	26.5	<DL	<DL	<DL	<DL	6.7	0.3	1.1	<DL	7.9	<DL	274	<DL	1.5
S7$_{July}$	5.8	<DL	7.0	<DL	<DL	<DL	<DL	<DL	<DL	0.8	<DL	<DL	<DL	1900[1]	10	<DL

Table 6.7: Major ions from July 2019. Where results were below detection limits, <DL is used.

Site	Ca	K	Mg	Na	Br	Cl	DIC as HCO_3	F	SO_4
	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
S1	13.3	0.64	1.34	3.37	>DL	1.44	49.4	0.040	5.26
S2	14.3	0.79	1.54	3.53	>DL	1.41	51.3	0.039	5.21
S3	13.8	0.84	1.60	3.76	>DL	1.37	49.7	0.046	5.10
S4	14.1	0.91	1.80	4.06	>DL	1.26	49.0	0.052	5.12
S5	12.4	0.78	1.50	3.08	>DL	1.70	42.7	0.073	4.70
S6	11.6	0.71	1.31	2.58	>DL	1.21	36.4	0.073	5.27
S7	10.6	0.56	1.12	2.47	>DL	1.10	36.2	0.076	5.26

Table 6.8: Trace elements in sediment from May 2019.

Site	Al	As	B	Cd	Co	Cr	Cu	Fe	Mn	Mo	Ni	P	Pb	S	V	Zn
	wt%	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	wt%	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
S1	0.70	1.9	5.2	1.9	0.6	11.0	5.8	0.91	197	0.3	7.5	521	9.2	113	151	70.0
S2	0.78	2.2	4.8	1.9	0.5	12.2	31.2	1.00	167	0.3	7.8	562	61.6	159	170	48.3
S3	0.94	2.7	5.2	1.8	0.8	14.1	8.6	1.17	207	0.4	9.9	422	23.7	85	208	58.9
S4	0.80	2.9	5.0	1.6	0.7	11.7	9.3	1.00	186	0.2	7.5	480	31.7	43	200	40.8
S5	0.93	2.3	4.7	1.7	0.6	13.1	11.3	1.11	176	0.2	10.7	378	13.9	36	199	66.9
S6 (Marginal)	1.38	4.6	8.2	1.7	1.2	21.2	345.8	1.63	267	0.9	13.3	959	134.1	258	306	67.7
S6 (Instream)	0.65	1.7	4.1	3.1	0.4	10.2	7.1	0.77	133	0.8	6.2	428	6.4	196	136	34.6
S7	1.91	26.9	44.0	4.5	1.1	59.0	380.6	2.56	289	5.2	34.2	933	211.5	2719	853	379.9
ANZECC Low		20		1.5		80	65				21		50			200
ANZECC High		70		10		370	270				52		220			410

Table 6.9: Grain size composition from May 2019.

	>2 mm	1-2 mm	0.5-1 mm	0.25-0.5 mm	0.125-0.5 mm	0.063-0.125 mm	<0.063 mm
S1	17.7	2.2	4.5	42.3	19.4	4.1	9.8
S2	0.3	0.3	0.7	18.3	54.0	19.7	6.6
S3	46.9	8.2	0.1	18.5	15.8	3.3	7.2
S4	1.6	4.4	4.9	54.3	28.4	5.8	0.6
S5	19.1	0.6	2.4	19.0	29.3	16.9	12.7
S6 (Marginal)	19.0	12.5	8.4	5.2	5.2	7.5	42.2
S6 (Instream)	5.9	6.6	11.2	47.2	7.7	13.4	8.1
S7	18.8	5.0	4.7	10.9	21.5	8.0	31.1

Table 6.10 Invertebrate abundances as measured on one occasion in July 2019.

Higher Taxonomic Group	Taxon	Species	Abundance									
			Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10
Acarina	Acarina		8	6	3		13	7	14	89	4	
Annelia	Oligochaeta		58	24	17	3	18	69	11	11	3	13
Coleoptera	Elmidae		38	26	14		83	5	10	72		7
Collembola	Collembola				1		4			3		
Crustacea	Amphipoda				1		4		14			
Crustacea	Cladocera					2	1					
Crustacea	Copepoda				7	2			4	1		1
Crustacea	Ostracoda		9	318	18	47	8	771	610	232	72	44
Crustacea	Paracalliope	Paracalliope fluviatilis		76	1	383	3			47	115	
Diptera	Austrosimulium		1	4				1	10	4		27
Diptera	Corynoneura		7						110			76
Diptera	Empididae				1							
Diptera	Lobodiamesa	Lobodiamesa campbelli		6	5	1	1	4		2		
Diptera	Mischoderus			1	1		1		1	4		
Diptera	Orthocladiinae		250	219	28	30	10	115	292	22	23	172
Diptera	Paradixa											9
Diptera	Tanyderidae	Mischoderus					1					
Diptera	Tanypodinae		11	2						4	1	
Diptera	Tanytarsini							1				
Diptera	Tipulidae	Paralimnophila		1						1		
Ephemeroptera	Coloburiscus	Coloburiscus humeralis		16	8	4	33			6	21	
Ephemeroptera	Deleatidium		1	585	1229	197	689		18	1268	601	19
Hemiptera	Sigara							2			1	
Mollusca	Gyraulus		2	2		4		3			2	7

Mollusca	Lymnaeidae											
Mollusca	Physella	Physella (Physa) acuta	248	15	6	7	8	60	13	3	3	31
Mollusca	Potamopyrgus	Potamopyrgus antipodarum	826	579	587	38	201	440	38	47	152	27
Mollusca	Sphaeriidae		1	158	8	56	12	7		1	147	9
Odonata	Xanthocnemis	Xanthocnemis zealandica					1	1	2			
Plecoptera	Zelandobius			1	1				2			
Trichoptera	Helicopsyche	Helicopsyche albescens			82	2	661	18		2	398	
Trichoptera	Hudsonema	Hudsonema amabile	67	336	37	23	8	24	32	116	81	17
Trichoptera	Hydrobiosis		46	52	21	1	1	14	1	35	9	46
Trichoptera	Hydropsyche-Aoteapsyche	Aoteapsyche colonica	7	75	159	2	149		27	76	2	
Trichoptera	Neurochorema	Neurochorema forsteri	1	2	1						1	1
Trichoptera	Oecetis	Oecetis unicolor		11	2			2			1	
Trichoptera	Olinga	Olinga feredayi	1	12	10	3	9	9	24	70	28	5
Trichoptera	Oxyethira	Oxyethira albiceps	25	40	2	9	4	97	98	10	6	31
Trichoptera	Polyplectropus					1		2	62			
Trichoptera	Psilochorema	Psilochorema bidens	35	129	41	10	37	5		35	44	98
Trichoptera	Pycnocentria		234	2464	2096	259	611	68	289	2336	1221	92
Trichoptera	Pycnocentrodes	Pycnocentrodes aureulus	286	48	505	41	65	30	19	83	30	
Trichoptera	Triplectides	Triplectides obsoletus						6	2			
Total Abundance			2162	5208	4892	1125	2636	1761	1703	4580	2966	732

Table 6.11: Macroinvertebrate indices from July 2019.

Site	Total Abundance	Taxonomic Richness	EPT Taxa	% EPT	SQMCI	MCI
S1	2162	22	10	45.5	4.2	91.8
S2	5208	28	12	42.9	5.7	99.3
S3	4892	29	14	48.3	5.9	100.7
S4	1125	23	10	43.5	5.8	105.2
S5	2636	27	11	40.7	7.7	103.7
S6	1761	25	10	40	3.3	98.4
S7	1703	24	9	37.5	3.6	90
S8	4580	27	11	40.7	6.7	103.7
S9	2966	24	13	54.2	6.8	104.2
S10	732	20	8	40	3.6	91

Table 6.12: SIMPER results for contributors to macroinvertebrate community differences between 2008, 2012, 2017 and 2019.

Year – Year Comparison	Taxa	Cumulative Percentage
2008 - 2012	Potamopyrgus	23.3
	Pycnocentrodes	36.0
	Pycnocentria	45.9
	Hydropsyche-Aoteapsyche	52.3
	Deleatidium	57.9
	Orthocladiinae	63.4
	Physella	68.2
	Oxyethira	72.8
2008 - 2017	Pycnocentria	29.5
	Potamopyrgus	44.3
	Deleatidium	51.9
	Pycnocentrodes	58.0
	Ostracoda	63.0
	Orthocladiinae	67.6
	Oxyethira	71.6
2008 - 2019	Pycnocentria	21.1
	Potamopyrgus	35.5
	Deleatidium	46.5
	Ostracoda	56.5
	Pycnocentrodes	63.0
	Orthocladiinae	67.5
	Helicopsyche	71.6
2012 - 2017	Pycnocentria	20.1
	Potamopyrgus	38.4
	Pycnocentrodes	48.1
	Deleatidium	53.9
	Hydropsyche-Aoteapsyche	59.2
	Orthocladiinae	64.1
	Ostracoda	68.5
	Physella	72.5
2012 – 2019	Potamopyrgus	18.2
	Pycnocentria	33.8
	Pycnocentrodes	44.8
	Deleatidium	53.7
	Ostracoda	61.1
	Hydropsyche-Aoteapsyche	66.0
	Orthocladiinae	70.1
2017 - 2019	Pycnocentria	26.6
	Potamopyrgus	38.6
	Deleatidium	49.1
	Ostracoda	58.0
	Pycnocentrodes	63.8
	Orthocladiinae	68.2
	Hydropsyche-Aoteapsyche	71.7

Appendix 3: Land Cover Classifications

Table 6.13: Descriptions of landcover classification hierarchy (LAWA, 2017).

Detailed Class	Class Description
Transport infrastructure	Artificial surfaces associated with transport such as arterial roads, rail-yards and airport runways. Skid sites and landings associated with forest logging are sometimes also included.
Surface mine or dump	Bare surfaces arising from open-cast and other surface mining activities, quarries, gravel-pits and areas of solid waste disposal such as refuse dumps, clean-fill dumps and active reclamation sites.
Sand or gravel	Bare surfaces dominated by unconsolidated materials generally finer than coarse gravel (60mm). Typically mapped along sandy seashores and the margins of lagoons and estuaries, lakes and rivers and some areas subject to surficial erosion, soil toxicity and extreme exposure.
Landslide	Bare surfaces arising from mass-movement erosion generally in mountain-lands and steep hill-country.
Gravel or rock	Bare surfaces dominated by unconsolidated or consolidated materials generally coarser than coarse gravel (60mm). Typically mapped along rocky seashores and rivers, sub-alpine and alpine areas, scree slopes and erosion pavements.
Permanent snow and ice	Areas where ice and snow persists through late summer. Typically occurring above 1800m but also at lower elevations as glaciers.
Alpine grass/herbfield	Typically sparse communities above the actual or theoretical treeline dominated by herbaceous cushion, mat, turf, and rosette plants and lichens. Grasses are a minor or infrequent component, whereas stones, boulders and bare rock are usually conspicuous.
Built-up area (settlement)	Commercial, industrial or residential buildings, including associated infrastructure and amenities, not resolvable as other classes. Low density 'lifestyle' residential areas are included where hard surfaces, landscaping and gardens dominate other land covers.
Urban parkland/open space	Open, mainly grassed or sparsely-treed, amenity, utility and recreation areas. The class includes parks and playing fields, public gardens, cemeteries, golf courses, berms and other vegetated areas usually within or associated with built-up areas.
Short-rotation cropland	Land regularly cultivated for the production of cereal, root, and seed crops, hops, vegetables, strawberries and field nurseries, often including intervening grassland, fallow land, and other covers not delineated separately.
Orchards, vineyards or other perennial crops	Land managed for the production of grapes, pip, citrus and stone fruit, nuts, olives, berries, kiwifruit, and other perennial crops. Cultivation for crop renewal is infrequent and irregular but is sometimes practiced for weed control.
Forest - harvested	Predominantly bare ground arising from the harvesting of exotic forest or, less commonly, the clearing of indigenous forest. Replanting of exotic forest (or conversion to a new land use) is not evident and nor is the future use of land cleared of indigenous forest.
Exotic forest	Planted or naturalised forest predominantly of radiata pine but including other pine species, Douglas fir, cypress, larch, acacia and eucalypts. Production forestry is the main land use in this class with minor areas devoted to mass-movement erosion-control and other areas of naturalised (wildling) establishment.
Deciduous hardwoods	Exotic deciduous woodlands, predominantly of willows or poplars but also of oak, elm, ash or other species. Commonly alongside inland water (or as part of wetlands), or as erosion-control, shelter and amenity plantings.
Indigenous forest	Tall forest dominated by indigenous conifer, broadleaved or beech species.
Broadleaved indigenous hardwoods	Lowland scrub communities dominated by indigenous mixed broadleaved shrubs such as wineberry, mahoe, five-finger, *Pittosporum* spp, fuchsia, tutu, titoki and tree ferns. This class is usually indicative of advanced succession toward indigenous forest.
Depleted grassland	Areas, of mainly former short tussock grassland in the drier eastern South Island high country, degraded by over-grazing, fire, rabbits and weed invasion among which *Hieracium* species are conspicuous. Short tussocks usually occur, as do exotic grasses, but bare ground is more prominent.
High producing exotic grassland	Exotic sward grassland of good pastoral quality and vigour reflecting relatively high soil fertility and intensive grazing management. Clover species, ryegrass and cocksfoot dominate with lucerne and plantain locally important, but also including lower-producing grasses exhibiting vigour in areas of good soil moisture and fertility.

Low producing grassland	Exotic sward grassland and indigenous short tussock grassland of poor pastoral quality reflecting lower soil fertility and extensive grazing management or non-agricultural use. Browntop, sweet vernal, danthonia, fescue and Yorkshire fog dominate, with indigenous short tussocks (hard tussock, blue tussock and silver tussock) common in the eastern South Island and locally elsewhere.
Herbaceous freshwater vegetation	Herbaceous wetland communities occurring in freshwater habitats where the water table is above or just below the substrate surface for most of the year. The class includes rush, sedge, restiad, and sphagnum communities and other wetland species, but not flax nor willows which are mapped as Flaxland and Deciduous Hardwoods respectively.
Flaxland	Areas dominated by New Zealand flax usually swamp flax (harakeke) in damp sites but occasionally mountain flax (wharariki) on cliffs and mountain slopes.
Herbaceous saline vegetation	Herbaceous wetland communities occurring in saline habitats subject to tidal inundation or saltwater intrusion. Commonly includes club rush, wire rush and glasswort, but not mangrove which is mapped separately.
Tall tussock grassland	Indigenous snow tussocks in mainly alpine mountain-lands and red tussock in the central North Island and locally in poorly-drained valley floors, terraces and basins of both islands.
Gorse and/or Broom	Scrub communities dominated by gorse or Scotch broom generally occurring on sites of low fertility, often with a history of fire, and insufficient grazing pressure to control spread. Left undisturbed, this class can be transitional to Broadleaved Indigenous Hardwoods.
Mixed exotic Shrubland	Communities of introduced shrubs and climbers such as boxthorn, hawthorn, elderberry, blackberry, sweet brier, buddleja, and old man's beard.
Manuka and/or Kanuka	Scrub dominated by mānuka and/or kānuka, typically as a successional community in a reversion toward forest. Mānuka has a wider ecological tolerance and distribution than kānuka with the latter somewhat concentrated in the north with particular prominence on the volcanic soils of the central volcanic plateau.
Matagouri or Grey scrub	Scrub and shrubland comprising small-leaved, often divaricating shrubs such as matagouri, *Coprosma* spp, *Muehlenbeckia* spp., *Casinnia* spp., and *Parsonsia* spp. These, from a distance, often have a grey appearance.
Fernland	Bracken fern, umbrella fern, or ring fern, commonly on sites with low fertility and a history of burning. Manuka, gorse, and/or other shrubs are often a component of these communities and will succeed Fernland if left undisturbed.
Sub alpine shrubland	Highland scrub dominated by indigenous low-growing shrubs including species of *Hebe*, *Dracophyllum*, *Olearia*, and *Cassinia*. Predominantly occurring above the actual or theoretical treeline, this class is also recorded where temperature inversions have created cooler micro-climates at lower elevations e.g. the 'frost flats' of the central North Island.
Mangrove	Shrubs or small trees of the New Zealand mangrove (*Avicennia marina* subspecies *australascia*) growing in harbours, estuaries, tidal creeks and rivers north of Kawhia on the west coast and Ohiwa on the east coast.
Lake or pond	Essentially-permanent, open, fresh-water without emerging vegetation including artificial features such as oxidation ponds, amenity, farm and fire ponds and reservoirs as well as natural lakes, ponds and tarns.
River	Flowing open fresh-water generally more than 30m wide and without emerging vegetation. It includes artificial features such as canals and channels as well as natural rivers and streams.
Estuarine open water	Standing or flowing saline water without emerging vegetation including estuaries, lagoons, and occasionally lakes occurring in saline situations such as inter-dune hollows and coastal depressions.