NIWA
Taihoro Nukurangi

The effects of groundwater discharge on the algal, invertebrate and fish communities of the Waikanae River

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Executive summary

1. The Kapiti Coast District Council (KCDC) has been investigating ways to supplement water supply for its district. Following detailed investigations of six short-listed options, 2 options were finalised to meet the growing demand for water: River Recharge with Groundwater (RRwGW), and a dam on the lower Mangakotukutuku . A central issue to be considered for the RRwGW option is changes in water chemistry of the Waikanae River as a result of the ground water discharge, and potential effects of these changes on the river's ecology. The groundwater discharge will only occur during periods of low river flow in summer and early autumn - during a 50 year return period drought and with demand at 32,000 m^3/day the longest period of continuous discharge is predicted to be 59 days. NIWA was contracted to assess the magnitude of these effects. This report was conducted to assess potential effects of groundwater discharge on water chemistry, as well as key ecosystem components of the Waikanae River from the bores: algae, invertebrates and fish. We also assessed potential ecological effects of extracting groundwater on three nearby wetlands.

2. A series of longitudinal baffles were constructed along the true right side of the river, and bore water was discharged into this area at a rate of approximately 120 L per second. These baffles were designed to give approximately 70% dilution of groundwater within this experimental channel - close to predicted scenarios of a one in 50 year low flow period, combined with maximum projected population related demand in the year 2060 (without universal metering). Another set of baffles was constructed along the true left-side of the river to act as a control channel. This setup allowed us to monitor aspects of the river's ecology before and after groundwater discharge commenced, in the control and experimental (or impact) channels. This Before-After, Control-Impact (BACI) design represented a robust methodology to test the effect of groundwater discharge.

3. Changes to water chemistry were monitored by deploying data sondes in each channel to monitor temperature, dissolved oxygen, conductivity and pH. Spot measurements of conductivity were also made at regular intervals down the experimental channel to investigate mixing and dilution patterns. Water samples were also collected on three occasions and analysed for a range of nutrients, ions and cations.

4. We monitored algal, invertebrates and fish communities within the control and experimental channels before and after addition of groundwater. We sampled on two occasions prior to the discharge of groundwater, and on three occasions after the discharge had commenced. The discharge ran from February 18 until April 15 2011. We also monitored algal and invertebrate communities in detail at two sites well below the channels.

5. Algal communities were measured by a combination of algal biomass (as chlorophyll), and percent cover. Nutrient diffusing substrates were also deployed to assess the degree of nutrient limitation of algae. Invertebrate samples were collected from all sites to obtain information on species composition and abundance. Invertebrates were also collected from three nearby wetlands to

characterise their communities. A series of biotic metrics were calculated that described different components of the invertebrate community. Responses of fish to groundwater discharge were assessed by three independent methods. Firstly, fish numbers in control and experimental channels were quantified by electric fishing before and after the study. Secondly, three species of fish (Longfin eel, redfin bully, and inanga) were placed in specially designed fish cages that were deployed in the control and experimental channels. These species were chosen as they were commonly found in the Waikanae River. Fish survival and growth were monitored over time. Thirdly, a series of aquaria were constructed into which the three fish species were placed. Fish were subject to one of three treatments: 100% river water, 70% groundwater, and 100% groundwater, and monitored throughout the experiment.

6. Flows during the study were generally low, reflecting the natural summer pattern, although and a number of small, short-lived flushing flows occurred. These would have overtopped the baffles on three occasions, and further diluted the groundwater. Conductivity records from data sondes showed that dilution of groundwater averaged 63%, but this increased to 67% when the diluting influence of floods was removed - close to the predicted 70:30 mix with a 1 in 50 year low flow and a population demand in the year 2060. Although pH, temperature and dissolved oxygen differed between the experimental and control channels as a result of the discharge, the magnitude of this difference was small, and unlikely to have demonstrable impacts on ecological values. Examination of conductivity along the channel showed that dilution was close to the target of 70% within the experimental area.

7. Analysis of water chemistry data showed clear differences in the chemical signatures between groundwater, and water in the experimental and control channels, with the water in the experimental channel having an overall chemical signature closer to that of spring fed rivers. Water chemistry data was compared to that of other rivers from the Wellington region, and the North Island. Although water chemistry in the control channel was not distinguishable from these other rivers, water chemistry in the experimental channel was clearly different. These differences were driven by differences in concentrations of sodium, calcium, bicarbonate, magnesium and pH.

8. Chlorophyll biomass increased more in the experimental channel than the control channel, most likely reflected the coarser substrate size there. However, higher chlorophyll biomass was observed on large substrates in the main river adjacent to the experimental channel, and in the lower sites not subject to the influence of groundwater discharge. Such increases most likely reflect a normal summer increase in algal cover. Thus, the high biomass observed in the experimental channel and the lower sites was unlikely to have been caused by the groundwater. Proliferations of cyanobacterial mats naturally develop in the Waikanae River, and other rivers in both the Wellington and Canterbury regions, some of which have lower nutrient status than the Waikanae. These proliferations have been attributed to extended periods of low flow, and the absence of flushing flows. The effect of nutrient enrichment in exacerbating these proliferations in

these other rivers is regarded as minor. It is highly likely that this is also the case in the Waikanae River.

9. A flushing event near the conclusion of the study decreased algal biomass at all sites, but the magnitude of this reduction was less in the experimental channel. This reflected the fact that the wooden baffles would have ameliorated the effects of this flood by reducing instream velocities. Results of the nutrient diffusing substrates also suggested that algal growth in the Waikanae River was not limited by nutrients. Based on these observations, algal biomass will not increase as a result of discharging phosphorus rich groundwater more than it naturally does during summer low flow periods.

10. A diverse invertebrate fauna was found in the Waikanae River, which was dominated by mayflies, midges, caddisflies and beetles. Calculation of biotic metrics suggested that this river was in good ecological condition. Analysis of changes to invertebrate communities over time in the experimental and control channels showed little differences in their behaviour. Although conditions in the experimental channel appeared to have become less favourable to invertebrates such as the mayfly *Deleatidium*, and the caddisfly *Olinga*, these differences most likely reflected the effects of increased algal biomass that occurred during the summer low flow period, and not to any effects of groundwater on invertebrates. Addition of groundwater to the experimental channel therefore appeared to have little demonstrable effect on invertebrate communities. Strong relationships existed between chlorophyll biomass and invertebrate metrics, and densities of selected taxa. Total abundance, and abundance of midges and the caddisfly *Pycnocentrodes* increased with increasing biomass, whereas two biotic metrics, and densities of mites decreased. These strong relationships emphasise the close link between invertebrate communities and algal biomass, which naturally increases during the summer.

11. Fish density increased in both channels over time, and no differences in fish lengths were detected between control and experimental channels. No significant differences were apparent in growth rates of fish in cages in either channel. Results from the aquaria test showed that growth rate of inanga were unaffected by groundwater, even at 100%. These results suggest that groundwater had no adverse effects on fish survival or growth, and that fish were not moving away from areas where groundwater was being discharged into.

12. The invertebrate fauna found in the three wetlands was typical to that of other wetlands in the North Island. The fauna was dominated by the freshwater mud snail (*Potamopyrgus antipodarum*), oligochaete worms, the small pee-clam *Sphaerium*, and the common freshwater shrimp *Paracalliope.* All the animals encountered are common and widespread, and no animals of particular conservation interest or concern were identified. Many of the animals encountered were highly mobile and capable of rapidly recolonising wetlands that had dried up, and been re-wet. A separate groundwater modelling study suggests that the additional drawdown effect on the shallow aquifer (and therefore the wetlands) from the RRwGW option in a 50 year drought with demand at 32,000 m^3/day will be less than natural seasonal variation, so it is

highly unlikely that the proposed activity would have any particular adverse effect on these communities.

13. The practical implications of this study suggest that a number of adaptive management responses may be required. These include a revised hierarchy of bore preference to use those bores with the lowest phosphorus concentrations in the first instance. Secondly, a monitoring regime in the river should be established to observe whether the river recharge is causing any cyanobacterial growth over and above that occurring naturally in the river. Finally, while this report suggests this will not be the case - in the event there is excessive cyanobacterial growth downstream of the discharge, a range of active management techniques could be included as a resource consent condition in future. Such techniques could include active removal of cyanobacterial blooms from the rocks immediately downstream of the discharge, or some other means of mitigating the bloom.

14. Although the resultant water chemistry of the Waikanae River below the water treatment plant will change as a result of the groundwater discharge, the scientific studies undertaken at this stage conclude that the effects of RRwGW option are likely to be acceptable from an ecological perspective.

1 Introduction

The Kapiti Coast District Council (KCDC) has been investigating ways to supplement the water supply for the Waikanae, Paraparaumu and Raumati areas. A Kapiti water supply project was created to focus on this issue which narrowed down 40 district wide options to a shortlist of six in catchment options. These options are:

- Aquifer Storage and Recovery (Option 27)
- Groundwater River Recharge (Option 29)
- Waikanae Borefield and Storage (Option 23/38)
- Kapakapanui Dam (Option 12)
- Ngatiawa Dam (Option 13)
- Lower Maungakotukutuku Dam (Option 18).

Following consultation with CH2M Beca and other technical experts and the Technical Advisory Group, the KCDC agreed to use River Recharge with Groundwater (RRwGW; Option 29) as their preferred option to meet the growing demands for water, subject to further investigation. Briefly, the RRwGW option is based on pumping ground water into the river below the water treatment plant (WTP) to augment river flows, but only when they fall below the current minimum flow (750 L/s). Under a scenario of a 1 in 50 year drought and a forecast water demand of 32,000 m^3/day in the year 2060, approximately 70% of the river flow below the water treatment plant would be from groundwater, and this would last for a 2 month period during summer. However, this scenario is extreme, and the actual amount of groundwater pumped into the river at any one time would typically be much less, and would occur only when the difference between river flow and demand exceeds the current minimum flow. The main issues to be considered for this option are changes in water chemistry of the Waikanae River as a result of the ground water discharge, and potential effects of these changes on the river's ecology. Changes to the water chemistry of the Waikanae River will depend mainly on the relative volume of groundwater discharged into the river, its dilution, and the time that the discharge occurs for.

As part of the initial assessment phase, the KCDC arranged for a series of short-term pump tests from bores (K4 and Kb4) to discharge groundwater in the Waikanae River. Suren et al. (2010) investigated the short term impacts of these discharges as part of the initial assessment of the different options. Data sondes (recording water temperature, pH, dissolved oxygen and conductivity) were deployed at two sites: one above the water treatment plant, and one below the discharge of ground water. Water samples were collected during groundwater discharge tests from the same sites. A dye test was also conducted to demonstrate the flow and dilution dynamics of the groundwater discharge. Suren et al. found that the dye plume became more diluted as it moved downstream, and complete mixing was observed after 100 m. However, a similar dilution would not occur during drier conditions, and the 1.50 year in 2060 scenario would be groundwater comprising *ca*. 70% of river flow below the discharge at the WTP, and all the way to the estuary. Subsequent modelling showed that groundwater augmentation to the Waikanae River under this low flow, high water demand scenario would increase conductivity, alkalinity, dissolved calcium and hardness, pH, ammonium-N and DRP, based on current chemical properties of bores K4 and Kb4. Concentrations of the latter were predicted to approximately double under this scenario. Increased DRP concentrations, when combined with stable low flows that are common in summer raised the concern that there may be undesirable periphyton growth below the discharge. Because there is significant variability in water chemistry bore by bore (Suren et al 2010), the final choice of bores used for the RRwGW option may alter the overall changes to water quality in the river. Given the strong effect that periphyton proliferations can have on stream ecosystems (e.g., Suren et al. 2003, Kilroy et al. 2009, Suren and Riis 2010), it is imperative to determine whether groundwater discharge would cause excessive algal proliferations over and above what normally accrues during summer.

In addition, work by Death and Joy (2004) showed that the invertebrate communities of 187 minimally impacted streams throughout the Manawatu-Whanganui region were largely controlled by alkalinity and conductivity - two parameters that are likely to change as a result of the discharge. Streams with very low alkalinity were dominated by mayflies, caddisflies and stoneflies, whereas these were gradually replaced by snails, crustaceans and midges as alkalinity increased.

The fauna of the Waikanae River is currently dominated by mayflies (*Deleatidium* and *Coloburiscus*), six species of caddisfly (*Pycnocentrodes, Olinga, Psilochorema, Helicopsyche, Aoteapsyche* and *Costachorema*), elmid riffle beetles, the snail *Potamopyrgus* and orthoclad midges. Many of these taxa (e.g., *Helicopsyche*, *Olinga*, *Coloburiscus* and *Deleatidium*) are indicative of streams in good condition, with low-nutrient water (Stark 1993). Whether this fauna would be affected by short-term (i.e., up to a maximum of 60 days) increases in alkalinity and conductivity arising from RRwGW during summer low flow was unknown.

Discharging groundwater into the Waikanae River may also lower water tables in the area, with possible effects on nearby wetlands. Changes to hydrological regimes, and in particular lowering of water tables, can have a dramatic effects on wetland condition (Mitsch and Gosselink 2000, Clarkson et al. 2003, Wissinger et al. 2009). Lowered water tables may reduce the extent of water-logged soils, leading to colonisation by invasive weedy species such as willows, blackberry and gorse. Lowering water tables may also affect the aquatic components of wetlands as these habitats may dry or decrease in size. A number of small ponds and wetlands exist in the Waikanae area (e.g., Totara Lagoon, Nga Manu Wildlife Reserve) and these may be susceptible to any potential lowering of water tables. As such, these needed to be monitored to determine firstly whether any changes in water level could be detected as a result of discharging groundwater from the aquifer, and secondly to characterise the invertebrate communities in these wetlands.

The earlier ecological assessment of Waikanae River by Suren et al (2010) showed that the Waikanae River has high ecological and fisheries values. As such, there may be concern at the use of groundwater to replenish river flows, especially during severe summer droughts. This is particularly pertinent given the perception of many residents in the area that groundwater is of potentially "inferior" quality because it is not regarded as suitable for drinking water. Given these concerns, it was necessary to conduct a robust experiment to determine whether the discharge of ground water would:

1. result in an increase in algal biomass;
2. alter the composition or abundance of invertebrate communities as a result of changes to water chemistry, or by any increases in algal biomass;
3. affect resident fish communities, including whitebait and eels (tuna), by changes to water chemistry; or
4. affect the ecology of nearby wetlands.

This assessment was therefore designed to monitor changes to 3 important components of the Waikanae River ecosystem, and to nearby wetlands. Any adverse effects in one or more of these different components were expected to have been detected by the devised monitoring program.

2 Methods

2.1 Channel construction

The effects of a trial 56-day discharge of groundwater into the Waikanae River was assessed by constructing an experimental channel using a series of longitudinal plywood baffles along the true right of the river. These baffles were extended to meet the bank at the point where the groundwater was discharged through a series of large boulders. This was done to minimise the dilution of groundwater by natural river water to meet the target of approximately 70% groundwater within the experimental channel. Plywood sheets were bolted together using untreated timber (75 mm x 50 mm) and fastened into warratahs driven into the streambed. Small gravels were placed along the bottom of each sheet to minimise leakage and cross-contamination of ground water and river water. These baffles were cut to a height so that at low flow they stood slightly above the water, while at higher flows they became submerged, thus minimising forces on them. The height of the baffles meant that water would overtop them at a stage greater than 2370 mm, or 3009 L/s – slightly higher than the long term median flow. Bore water from bores K4 and K6 was discharged into the river at a rate of ca. 120 L/s. As part of the discharge process, all bore water went through the aeration tower at the WTP, and was then discharged directly to the river via the stormwater system.

Another series of baffles was constructed on the opposite bank through which natural river water would flow, which acted as a control channel (Figure 2-1). This control channel was located behind an exposed boulder bar that extended into the river channel, downstream of which was a substrate dominated by small cobbles and gravels. The channel was placed in this location as further upstream was an area of deep, slow water, and further downstream was too bouldery, fast flowing and shallow to match the experimental site. In addition, studies by Suren et al (2010) showed that groundwater became mixed beyond the right angled bend in the river that was below this fast flowing section. The discharge commenced on February 18th at 1530h, and continued for 8 weeks until April 15th when the discharge was also terminated at 1530h.

Concentrations of groundwater were confirmed by deploying data sondes in each channel that recorded temperature, dissolved oxygen, conductivity and pH every 15 minutes during the trial. These were placed in the middle of each channel, and at 20m above the downstream end of each channel. We also measured instantaneous conductivity on three occasions at cross-sections every 5 m along the experimental channel from the origin of the discharge. At each cross-section we measured conductivity at 25%, 50% and 75% across the channel width. Water samples were also collected on three occasions from the discharge point (i.e., 100% groundwater), the experimental and control channels and analysed for a range of determinants such as nutrients, ions, pH, and hydrogen sulphide.

Substrate size in each channel was quantified using the Wolman technique (Wolman 1954). Here, 100 sediment particles were randomly selected from each channel and particle width measured according to the standard Wolman size classes. A series of gaugings were also conducted in both experimental channels as well as the main channel in order to obtain information on depth, velocity and flow.

Figure 2-1: Photo of the baffles in the experimental channel (foreground) and control channel (background).

2.2 Biological sampling

Sampling in the experimental channel was restricted to areas 20 m below the upstream discharge point to avoid sampling in undiluted groundwater. Sampling was done along the entire length of the control channel. We monitored algal, invertebrate, and fish communities within the experimental channels, both before and after[1] the addition of groundwater. In this case, the proposed monitoring represents a Before After Control Impact (BACI) study: the most robust form of study to detect changes as a result of human activities (Underwood 1991). We collected samples from the experimental channels on two occasions prior to the discharge of groundwater and on three occasions during the discharge. This data formed the basis for our first analyses, which examined in detail the effects of groundwater discharge on biological communities in the channels.

We also monitored algal and invertebrate communities at two sites well below the discharge: one below the State Highway 1 road bridge (1.2 km below the treatment plant), and one at the lower end of the Jim Cooke Park at the end of Greenwaway Rd (4.4 km below the treatment plant – Figure 2-2). The Waikanae River loses water to groundwater between State Highway 1 and the Jim Cooke Park site, and then gains water below this area. These sites were chosen to act as downstream “controls” in the absence of groundwater inputs. Although we acknowledge that these sites would have displayed a slight increase in nutrients from the upstream groundwater discharge, the magnitude of this increase was very little, in the order of 7.5%[2]. .. These sites were only sampled on three occasions, once before the

[1] Note that, strictly speaking, the "After" time period actually referred to the period **during** which groundwater was discharged. The term "After" has been used consistently through this report to be consistent with the modern terminology of the Before, After, Control, Impact (BACI) design used in this experiment.

[2] The bore water was discharged at 120 L/s into the river, which maintained a median flow of 1593 L/s. Thus, assuming complete mixing and no loss of the mixed river water to ground water below the WTP, the median dilution would be 7.5%. This

discharge, and on two occasions during. Data from all four sites (the experimental and control channels, and the State Highway and Jim Cooke Park sites) on three sampling occasions formed the basis for the second analyses, comparing biological communities in the greater river during the summer period.

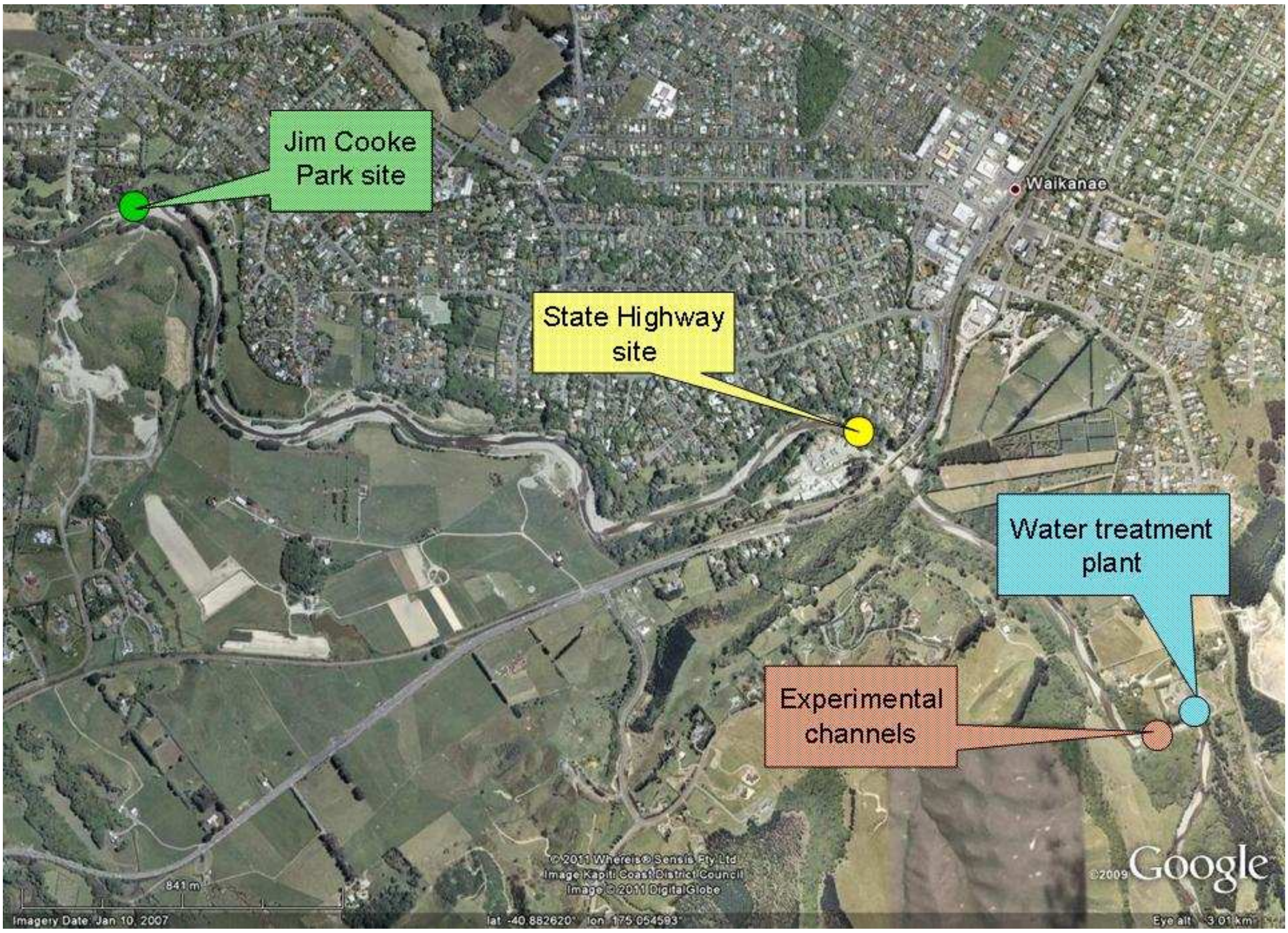

Figure 2-2: Map showing the locations of the study sites in the Waikanae River (Map from Google Map.

2.2.1 Algae

A key requirement of this assessment was to determine whether nutrients associated with the groundwater discharge might enhance algal cover or biomass. The effects of the groundwater discharge on algal communities were thus assessed by three methods. Firstly, we collected algal samples from cobbles in four locations in the Waikanae River: the control and experimental channels, and from the SH1 and Jim Cooke site. We selected 5 transects in each channel, and collected three cobbles at ca. 25%, 50% and 75% across each transect. All algal material within a 40 mm circle was removed for analysis using a combination of scrubbing brushes or scalpel blades. All removed material was frozen within 1h of collection and sent to the laboratory where algal biomass (as measured by the plant pigment chlorophyll *a*) was measured using standard techniques (Biggs and Kilroy 2000).

is about 8 times greater than the dilution of groundwater experienced in the experimental channel. The effects of this are considered to be negligible

Secondly, algal cover was visually assessed at the same four locations used for the chlorophyll analysis, as well as at two other locations: above the water treatment plant weir (and therefore above the groundwater discharge), and below the sharp bend in the river, some 30 m below the end of the experimental channel. Algal cover was assessed at between 2 and 5 transects at each location. At each transect, observations were made at five equidistant points across the channel of the percentage of algal cover in a 0.25 m^2 area of streambed. Algae were assigned to one of the six categories: no algae; thin diatom films; medium blue green mats; medium diatom films; filamentous green algae; and thick diatom films (see Biggs and Kilroy 2000). Assessments were made at all sites on April 5 (before a large flood), and on the last sampling occasion (April 14).

Finally, an instream nutrient assay (Francoeur et al. 1999, Biggs and Kilroy 2000) was conducted, as this is regarded as being the least ambiguous approach for establishing whether algal communities are nutrient limited. Here, porous filter papers are placed over containers of agar enriched with nutrients. A combination of phosphorus alone, nitrogen alone, phosphorus plus nitrogen, and a control of no nutrients were used. The nutrients slowly diffused from the agar and into the substrate (in this case the filter paper), providing a localised nutrient source. Algae grow on the filter papers, and if algal growth is limited by either nitrogen or phosphorus then it will show enhanced growth on the agar enriched with these nutrients. If ambient nutrient levels in the water are already high and not limiting algal growth, then no enhancement of algae will be found on the nutrient-enriched substrates. After a pre-determined time, the filter papers are removed and analysed for chlorophyll biomass using standard methods (Biggs and Kilroy 2000). Duplicate steel tray nutrient diffusing substrates (see Biggs and Kilroy 2000) were placed in the experimental and control channels on 22 March, and collected approximately 3 weeks later on 14 April.

If nutrients, and in particular phosphorus, were limiting periphyton growth, then we would expect to find higher chlorophyll biomass in the phosphorus enriched treatments, particularly in the control channel where phosphorus concentrations were already low. Such an increase in chlorophyll biomass would theoretically not be as high in the experimental channel, as this channel contained water with higher phosphorus load. In this way, we would be looking for a significant interaction effect between nutrient treatment, and channel type.

2.2.2 Invertebrates

Quantitative invertebrate samples were collected from all sites to obtain information on species composition and abundance. Samples were collected with a circular Hess sampler (area=0.76 m^2; 300 µm mesh) which was driven slightly into the streambed and all gravels within the sampler disturbed to a depth of 100mm below the streambed surface. All dislodged organic material was subsequently collected in a downstream net. Where large cobbles precluded the use of the Hess sampler, samples were collected using a “rock rolling” technique where large cobbles were simply lifted into a downstream net and all invertebrates clinging to these were dislodged (Death and Winterbourn 1994, Suren and Lambert 2006). The dimensions of each cobble (x, y and z) were measured to the nearest mm, and surface area calculated using standard equations (see Biggs and Kilroy 2000).

Six replicate samples were collected before the groundwater was discharged into the experimental channel, and then after a period of 20, 31 and 56 days. Collecting samples at successive times during the discharge allowed us to determine whether any short-term changes were occurring to the invertebrate community, as well as ascertaining the effect of increasing the duration of time that the groundwater was discharged for.

All invertebrate samples were processed using a modification of Protocol P3: full count (Stark et al. 2001), where all material trapped on a small sieve (0.5 mm mesh size) was examined for invertebrates under a dissecting microscope, instead of removing invertebrates by eye from a sub-sample. Invertebrates were identified to as low a taxonomic resolution as possible; usually to Family, Order, or Genera, and counted. Some of the larger insects (e.g., Trichoptera) could be identified to species, while other insects were either too small to identify to species, or could not be identified due to lack of suitable identification keys.

2.2.3 Fish

The impacts of groundwater discharges on fish communities were assessed by three independent methods. Firstly, fish numbers in both channels were quantified by electric fishing both before, and at the completion of the groundwater discharge. This was done to see whether any differences in fish found within each area could be detected, and may have shown whether fish moved away from the experimental channel. Secondly, a series of fish cages were placed in each channel, into which five fish were added. These cages (Figure 2-3) were constructed from 250 L plastic barrels cut in half lengthwise and fitted with steel netting (5 mm mesh size) to the upstream and downstream ends (Suren and Lambert 2006). The upstream end of the cage was hinged to facilitate the easy removal of fish for observational purposes. Cobbles were added to each cage for fish to seek cover amongst, and PVC pipes (20 mm diameter) added to cages supporting eels. Each cage was left in situ for the experiments duration. The large mesh of the end panels was assumed to have allowed invertebrate drift into the cages, so no additional feeding was conducted. This assumption was confirmed at the end of the experiment when invertebrates were observed within the cages.

Figure 2-3: Fish cages deployed in the channels to assess the effects of groundwater discharge in fish survival and growth.

Fish chosen for this experiment included long finned eels (tuna: *Anguilla dieffenbachii*), redfin bullies (*Gobiomorphus huttoni*), and inanga (whitebait: *Galaxias maculatus*). These fish were chosen as they represented some of the most common species in the Waikanae River. Eels and whitebait also have highly significant social and recreational values. Eels and red-fin bullies were obtained from the upper reaches of the Waikanae River by electric fishing, while inanga were obtained from small tributaries flowing into the lower reaches of the Waikanae using fine seine nets. All fish were allowed to recover from electric fishing in buckets, and only healthy looking and normally behaving fish were selected for the experiment. Prior to being placed into each cage, all fish were anaesthetised using 2-phenoxyethanol and their lengths (nose to tail fin) measured to the nearest mm. All fish were allowed to recover again in a bucket of river water prior to being placed into the cages. Fish were re-measured twice during the experiment by carefully removing the cobbles and PVC tubes from the cages, and then opening the hinged end of the cage and tipping the fish into a bucket (Figure 2-4). All measurements were made on fish that were anaesthetised. By comparing survival and growth of these three species in the control and experimental channels, we were able to determine whether exposure to groundwater affected short-term fish survival for the length of the expected RRwGW discharge, approximately 60 days, or growth rates.

Figure 2-4: Fish were examined on 2 occasions to assess growth and mortality. At each time, cobbles (or PVC pipes) in the cages were removed and the hinged front of the cage opened. The cages were upended and all fish inside the cages were tipped into a collecting bucket.

Thirdly, a series of aquaria were constructed to assess the effect of groundwater on fish survival. Use of aquaria safeguard against potential loss of cages from unpredictable high flows that may occur during the experiment. Each aquaria consisted of a 57L plastic bucket fitted with an inlet pipe and outlet tap, and placed on a series of three elevated platforms (Figure 2-4). Water flowed from a header tank above the uppermost platform into the upper tank, where it overflowed into the middle tank, which then overflowed into the lower tank prior to being discharged. All inflow and outflow pipes were sealed with fine mesh to prevent fish from escaping. Small air-stones were installed in each tank to maximise aeration and water movement.

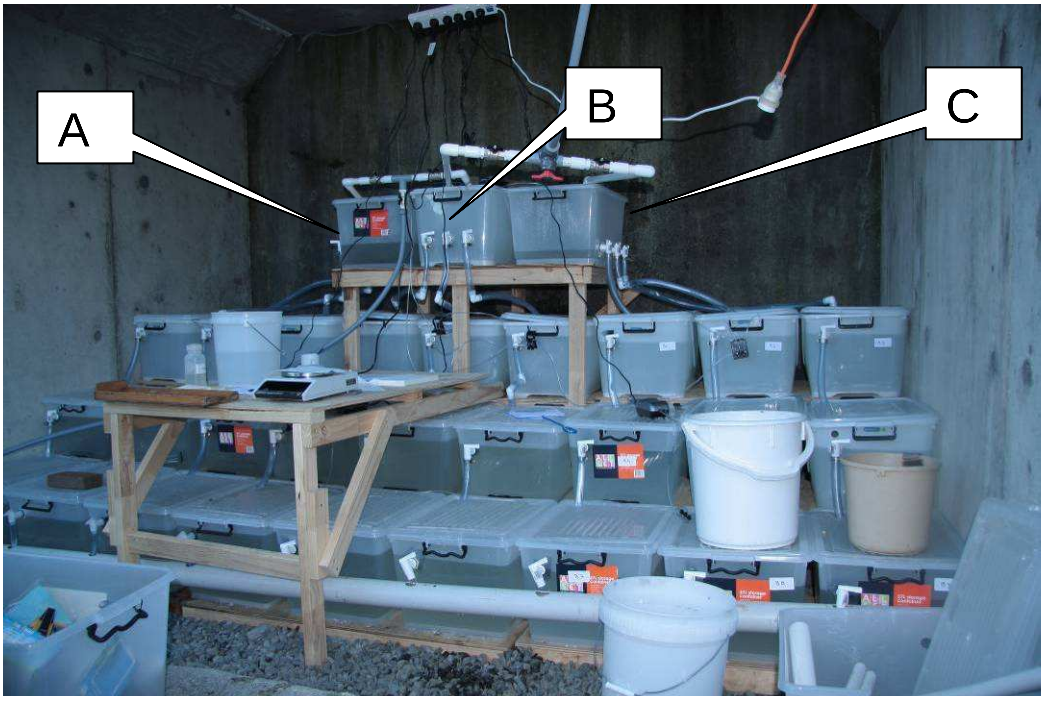

Figure 2-5: Fish tanks constructed from plastic containers and placed on a platform to allow water to flow from the upper to lower container, before being discharged to an overflow pipe. The three header tanks consisted of A) 100% river water; B) 30% river water and 70% groundwater; C) 100% groundwater.

The same fish species as used in the cage study were used in this experiment, and all fish were obtained from within the Waikanae catchment. Five individual fish were anaesthetised with 2-phenoxyethanol and their lengths measured prior to being placed in a tank. Cobbles were placed in each fish tank for shelter, as well as PVC pipes for eels. Fish were fed every second day with commercially available fish food (either pellets or frozen blood worms). Excess food was removed from the tanks every second day as well. Fish were subject to one of three treatments: 100% river water; 70% groundwater; and 100% groundwater. They were monitored throughout the experiment with notes made on mortality and escaping individuals at regular intervals. All dead fish were removed from tanks as soon as they were noted. No replacements were made for loss due to mortality or escapes. Spot measurements of water chemistry (pH, conductivity, temperature and dissolved oxygen) were made at intervals during the trial.

2.2.4 Wetlands

Three wetlands were selected for the study in the general area of the borefield (Figure 2-5). Wetland “A” and Wetland “C” (Totara lagoon) were deemed to be down gradient from the borefield, while wetland “B” (Nga Manu) was up gradient from the borefield.. Water levels in the Nga Manu wetland are currently monitored by the Greater Wellington Regional Council, allowing us to determine if any effects of the bore discharge could be detected. Water level recorders were also placed in the other two wetlands to measure water levels at 15 minute intervals, and left for the duration of the bore pumping, as well as an additional month afterwards.

Knowledge of wetlands invertebrate communities within New Zealand is in its infancy (Suren and Sorrell 2010). In particular, very little is known as to the effect of lowered water tables on wetlands invertebrate communities, despite the fact that wetland hydrology, and in particular the propensity of a wetlands to dry, has very large effects on invertebrate communities (Wissinger 1999). Furthermore, it is very difficult to sample wetland invertebrates quantitatively, meaning that any changes to density as a result of lowered water levels will be difficult to detect. For this study wetland invertebrates were sampled semi-quantitatively using a kick net (300 µm mesh) that was repeatedly jabbed through emergent vegetation, the water column and the substrate. Duplicate samples were collected from separate water bodies in each wetland and processed as outlined in Suren and Sorrell (2010) in order to characterise their biodiversity values.

2.3 Data analysis

2.3.1 Effects on water chemistry in river

All spot conductivity readings were converted to a percentage of the groundwater discharge, and a two-way analysis of variance (ANOVA) was used to see how conductivity changed along the length of the experimental channel, and across the width of the channel. A useful feature of the 2-Way ANOVA is the interaction term, which in this case tested whether conductivity changed in a consistent manner along the channel at different widths. A significant length × width interaction effect suggested that the pattern of conductivity along the channel differs across the width. Water quality data obtained from the data sondes was analysed by paired t-tests to determine whether dissolved oxygen, conductivity, pH or temperature differed between the control and experimental channels. All water chemistry data was analysed by principal component analysis (PCA) to determine differences in the overall nutrient and ionic signatures of the groundwater, the mixed water in the experimental channel, and the background river water. The % dilution of the groundwater in the experimental channel was calculated for the different nutrients and ions, as was the magnitude of change between the experimental channel and the control channel.

In addition to the measurements made during the groundwater discharge trial, we used predicted water chemistry downstream of the groundwater discharge under different discharge scenarios to assess the potential changes in water quality. Predicted water chemistry was reported in Suren et al. (2010) for discharges from different bores and under two different discharge rates (resulting in 27% and 70% dilution).

Figure 2-6: Map showing the location of the three wetlands examined for potential effects of pumping groundwater from the aquifer field. The wetlands were either up gradient (green box), or down gradient (red boxes) of the aquifer bore field.

To examine the significance of the measured and predicted changes to water chemistry we obtained major cation and anion data for multiple water sources including: sea water; spring sources from the literature (Michaelis 1977, Cowie and Winterbourn 1979), as well as from Environment Waikato, Environment Canterbury, and the Taranaki Regional Council; rainwater from three sites in NZ (Kelburn in Wellington; Puruki in inland North Island and Lauder in inland South Island; Nichol et al., 1997); and for a range of New Zealand rivers (Close and Davies-Colley 1990).

This data was used in two ways:

- Piper diagrams to display and compare the relative concentrations of major anions and cations to distinguish between different types of water. Major cations are plotted on one trilinear plot and major anions on another. The points are projected onto the central diamond which shows the overall chemical character of the water.
- PCA to determine differences in water chemistry, including nutrient concentrations between the measured and predicted Waikanae River under two different discharge rates and other natural water bodies.

2.3.2 Effects on ecological values

The following biological indices were calculated from the invertebrate data: total density; taxonomic richness; the hard-bottomed macroinvertebrate community index (MCI); its quantitative variant, the QMCI; the number of Ephemeroptera, Plecoptera, and Trichoptera taxa (i.e., EPT); and % EPT. The MCI represents a useful index describing overall invertebrate community “health”, with high scores (e.g., MCI > 120) indicating pristine waters, while scores < 80 indicate “probable organic enrichment”, and streams in poor condition (Stark 1993). The number and % of EPT taxa also conveys information about overall invertebrate community composition and condition. For example, as nutrient or algal biomass increase, the number or % of EPT taxa often declines. These different indices are useful for assessing both the current condition of the invertebrate community, as well as for monitoring changes to the community over time as a result of any activities in the catchment.

All biological data were checked for normality and fourth-root transformed where necessary. All data was then analysed using a two-way analysis of variance (ANOVA) to determine whether there were any differences in measured parameters describing invertebrate communities as a result of either time, or exposure to groundwater. The treatment effects in the ANOVA model were “time” (i.e., before and after), and “treatment” (i.e., control or experimental channel). We used the different sampling dates to represent replicates over time (Stewart-Oaten and Murdoch 1986), so that the first 2 samples were collected “Before” the groundwater discharge, and the last 3 samples were collected “After” the discharge. The time × site interaction term tested whether there was any difference in the behaviour of a particular metric over time between the different treatments. This would be expected to happen if the biological communities in the channels were not responding in a similar manner over time. A two-way ANOVA was run firstly for the control and experimental channels sampled on five occasions to see how biotic metrics changed before and after the discharge of groundwater. A second two-way ANOVA was run on the data collected from the channels and the lower sites sampled on three occasions to see how these metrics changed throughout the whole river. Although we were performing multiple statistical comparisons with the data, we did not employ any Bonferroni type adjustments, and therefore were more likely to reject the null hypothesis (H_0) that we were testing. In this case, the H_0 being tested was "*there was no impact of groundwater on biotic metrics*". Rejecting a true H_0 meant that we were saying "*there was a significant impact of groundwater on biotic metrics*". In other words, we were being more conservative than strictly necessary as a rejected H_0 meant that we detected an effect when in fact there was not. This is better than saying that there was not an effect, when indeed there was. This extra degree of conservatism was warranted in lieu of the risks of falsely saying that groundwater discharge has no effect on the river's ecology when indeed it does. In this way, a robust assessment was made as to whether discharging groundwater into the Waikanae River at concentrations typical to that of a low-flow 2060 scenario will indeed be having adverse ecological effects on the biota of the Waikanae River.

An ordination Non-metric Multi-Dimensional Scaling (NMDS) was then performed on the data (Clarke and Warwick 2001). This statistical technique graphically represents the location of samples based on their invertebrate communities, such that samples with similar communities are grouped on an *x-y* graph, while samples with different communities are far apart. Another statistical technique, Analysis of Similarities (ANOSIM), was also used to test whether there were any significant differences between two or more groups of sampling units. In this case, the groups represented samples collected from different sites (either the control or experimental channel, or the State Highway and Jim Cooke site) and at different times (before and after the groundwater discharge). ANOSIM produces a statistic, R, which indicates the magnitude of difference among groups of samples. An R of 1 indicates that the communities completely differ among defined groups, and an R of 0 indicates no difference among groups. The statistical significance of R was tested by Monte Carlo randomization. ANOSIM was done on both the data obtained from the five sampling occasions from the control and experimental channels, as well as the data obtained on three occasions from these channels and the two lower sites. The main comparisons of interest were the R statistic for samples collected within a particular site before and after groundwater was discharged. Thus, if the groundwater discharge was having a significant effect on invertebrate community composition, then the R-statistic for invertebrate community composition in the experimental channel over time would be higher than that in either the control channel, or the lower sites.

Both ANOSIM and NMDS rely on calculations of similarity between samples, so the Bray-Curtis similarity of all pairwise sample comparisons was calculated. This distance measure has been shown to have high statistical power in BACI analyses (Faith et al. 1991). The Bray-Curtis similarity is used to quantify the compositional similarity between two different sites. It is bound between 0 and 1, where 0 means the two sites have no species in common, while 1 means the two sites have the same composition (that is they share all the species).

NMDS and ANOSIM were also done on the invertebrate data collected from the three wetlands in the area to see how unique the fauna from these habitats was in comparison to that of other wetlands from throughout the North Island.

All fish data (i.e., numbers present in the channels, survival and growth in the cages, and aquaria) were also analysed by 2-Way ANOVA.

2.3.3 Interaction between algae and invertebrates

Recent studies have emphasised the importance of algal communities in influencing invertebrate communities (e.g., (McAuliffe 1983, Suren et al. 2003, Kilroy et al. 2009, Suren and Riis 2010). Indeed, such interactions were first postulated over 60 years ago by Allen (1951), who observed shifts in the invertebrate fauna of the Horokiwi Stream, near Wellington, over a nearly 20-year-period of agricultural development in the catchment. When the Horokiwi riverbed was covered with thin diatom layers, caddisflies and mayflies dominated the benthos, but when the riverbed was covered by filamentous green algae, elmid beetles, snails, midges, and ostracods dominated. Studies since then (e.g., Quinn and Hickey 1990, Harding and Winterbourn 1995, Quinn and Cooper 1997, Quinn 2000) have confirmed this general trend. Given such a close link between algal and invertebrate communities, interactions between chlorophyll biomass and invertebrate communities in the channels and the lower sites were investigated. Because paired samples of chlorophyll biomass and invertebrates were not collected, the means of chlorophyll biomass and of the various biological metrics describing the invertebrate communities at each site on each sampling occasion were calculated. In this way, we were able to determine whether biotic metrics were related to chlorophyll biomass. Scatter plots were made relating the different biotic metrics to chlorophyll biomass and appropriate regression techniques (linear or non-linear) were used to describe these relationships.

3 Results

3.1 Physical and chemical conditions

3.1.1 Physical conditions

Analysis of the particle size data showed that substrates in the experimental channel were significantly larger (median = 64 mm) than those in the control channel (median = 16 mm: Figure 3-1). This may have affected the degree to which algal communities colonised the channels, as larger, more stable particles are colonised more by algae than smaller, more easily moved particles. Moreover, the larger substrates in the experimental channel may have influenced invertebrate and fish communities as large substrates tend to have more spaces between them, which provides good habitat for these animals.

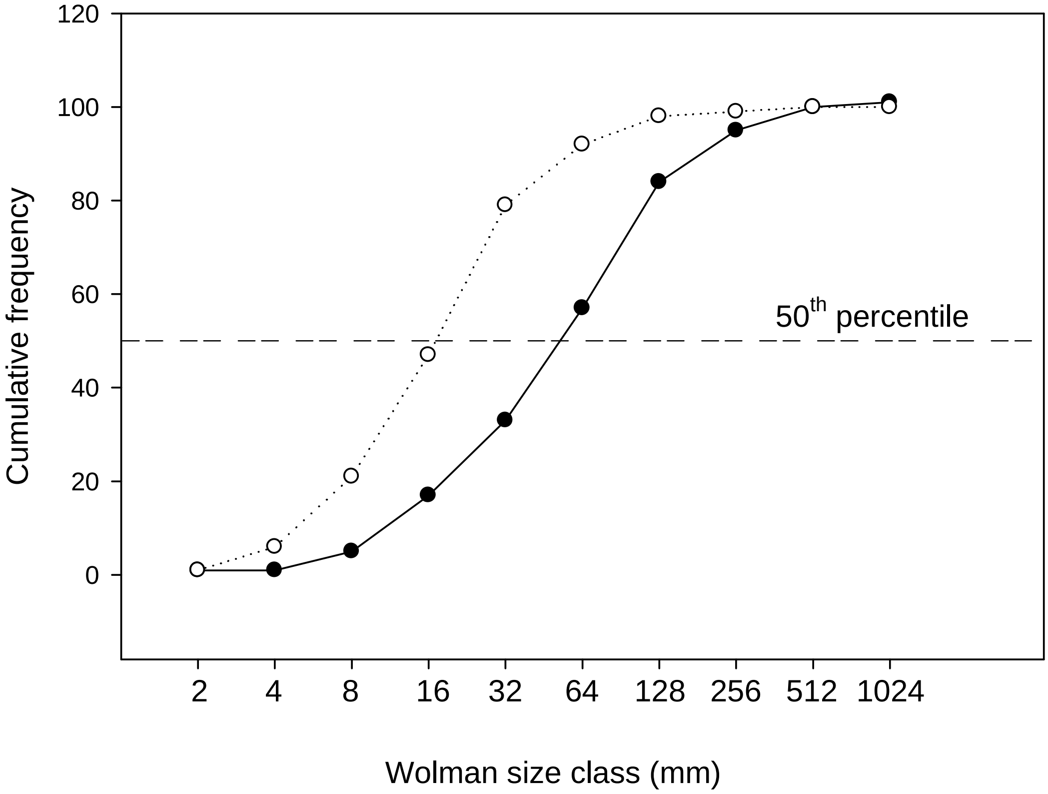

Figure 3-1: Stream substrate size class distribution in control channel (open symbols + dotted line) and experimental channel (filled symbols + solid line). Also shown is the 50th percentile line.

Two gaugings were done of flows in the experimental and control channels, as well as the unconfined river between these. In both cases, flows in the study channels were approximately 20% that of total river flow. Average flow in the experimental channel was slightly higher (c. 10%) than flow in the control channel. Conversely, average velocity in the experimental channel (mean = 0.39 ms^{-1}) was slightly slower than that in the control channel (mean = 0.56 ms^{-1}). These small differences were not expected to have influenced resultant benthic communities. Velocities in both channels were considerably slower than those in the main unconfined river (mean = 0.89 ms^{-1}).

3.1.2 Hydrological conditions

Flow data from the duration of the study was obtained from the Greater Wellington Regional Council (GWRC) hydrological site just above the water treatment plant. This data is used to generate a number of relevant flow statistic parameters for the Waikanae River (Table 3.1), including parameters describing floods (maximum and mean annual flood), and low flows (mean annual low flow or MALF, the 7-day MALF, and the minimum flow). Another statistic (FRE3) was used to describe the frequency of floods greater than three times the median flow. We also created a synthetic flow record of the site below the water take, based on the average daily take. As expected, the synthetic flow record below the take displayed reduced low flow statistics.

The average flow during the study was 2418 L/s, and median flow was 1593 L/s - almost half the river's natural mean and median flow. The average 7-day low flow was only 966 L/s, much higher than the existing 7-day MALF below the current take (Table 3-1). The hydrograph was also examined to determine how many flood events occurred during the study. We defined a flood event as times when flow in the river increased over time before decreasing again. If two flood peaks were observed within a 24-hour period, these were regarded as the same flood. A total of 8 floods were observed, with the highest peaking at 44.1 cubic meters per second on 5 April. The height of the baffles meant that water would overtop them at a stage (= water level above a known datum) greater than 2370 mm, or 3009 L/s – slightly higher than the long term median flow. Analysis of the hydrograph showed that this happened on three occasions (Figure 3-2). The other five floods resulted in only a slight increase in stage but without overtopping.

Table 3-1: Hydrological statistics for the unmodified flows at the water treatment hydrological site and for the various options downstream of the take.

Waikanae at water treatment		1976-2009	Current take
Maximum flood	$m^3\ s^{-1}$	381	381
Mean annual flood	m^3 s	155	155
Mean flow	m^3 s	4.79	4.53
Median flow	m^3 s	2.97	2.71
7 day MALF	m^3 s	0.923	0.785
MALF	m^3 s	0.928	0.792
Minimum flow	m^3 s	0.539	0.539
FRE3	Exceed y^{-1}	11.3	11.8

The average flood duration was 12 hours 30 minutes, but floods ranged from 3.5 hours to 104 hours in duration. The river was in flood for a total of 245 hours, or c. 18% of the time during the experiment.

3.1.3 Continuous water chemistry

Conductivity in the control channel changed little over time, and averaged at 100 µS cm^{-1} (Figure 3-2). Conductivity in the experimental channel was significantly higher (median = 539 µS cm^{-1}; paired t-test t = 248.4, $P < 0.001$), and more variable (Figure 3-2). Conductivity and flow data showed opposite trends, with conductivity decreasing with increasing flows. There was a significant negative linear relationship between flow and conductivity, explaining 72%

of the variability (Figure 3-3). Although five small floods were identified that did not overtop the baffles, they still reduced conductivity (Figure 3-2), which most likely reflects ingress of river water into the experimental channel through either joins between the plywood sheets, or along the river bed.

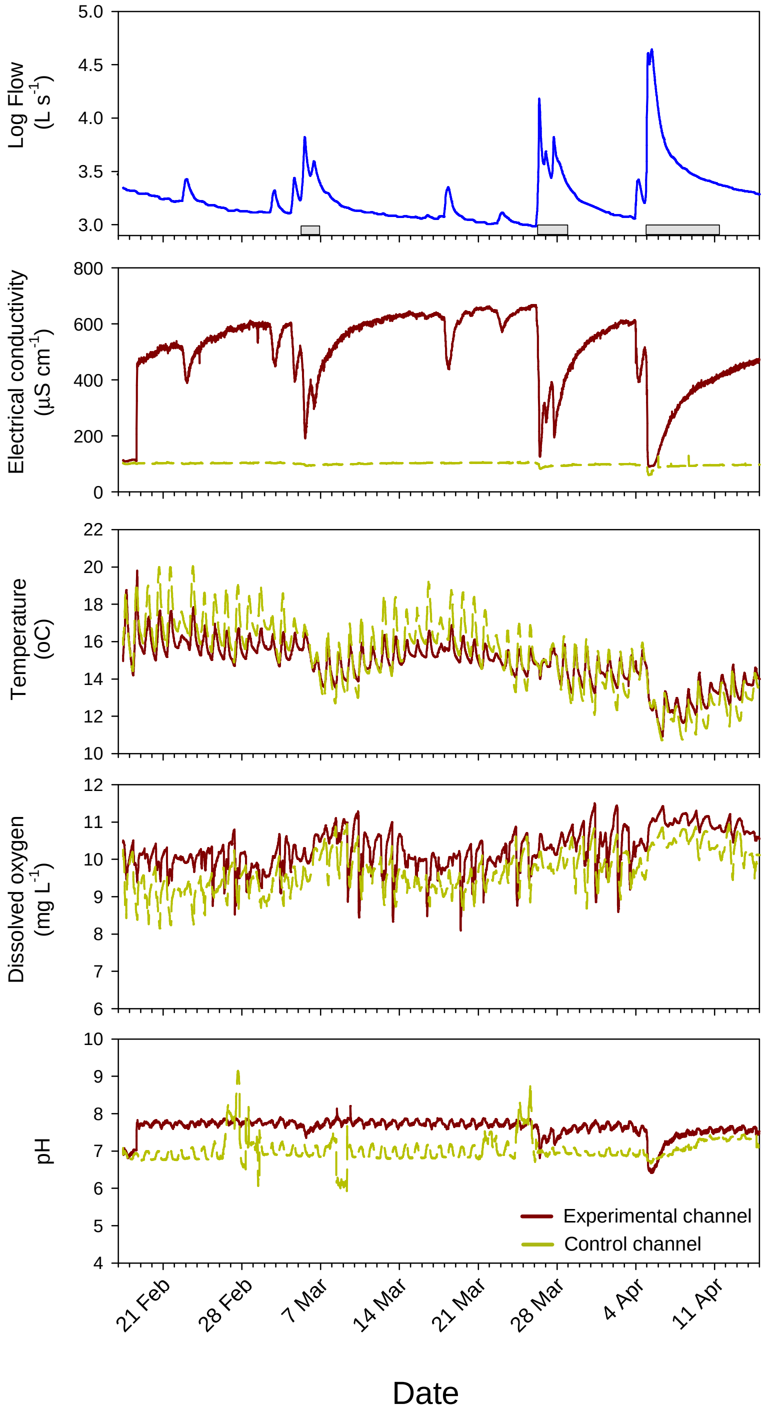

Figure 3-2: River water flow (in log scale for clarity), conductivity, river water temperature, dissolved oxygen and pH at sites within the experimental and control channels in the Waikanae. (Flow data kindly supplied by Wellington Regional Council.) Grey bars indicate times that the river flow overtopped the experimental channel.

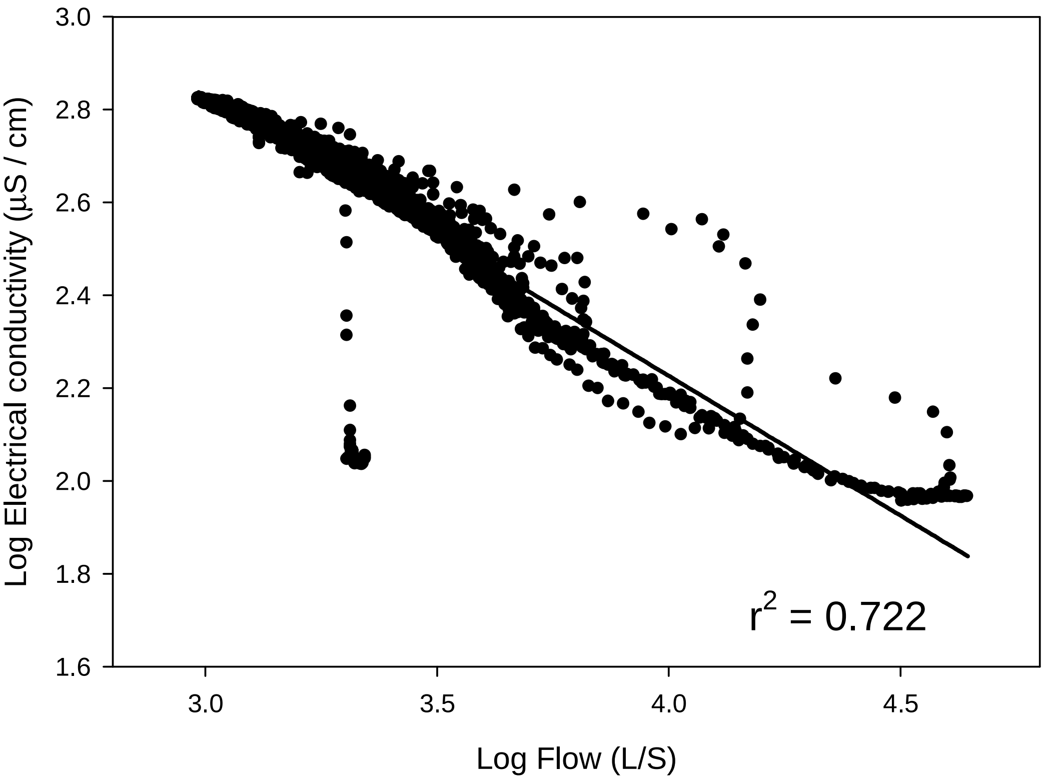

Figure 3-3: Relationship between river flow and electrical conductivity (log scale).

Under the predicted 2060 scenario of low river flows and high water demand, groundwater would constitute *ca*. 70% of river flow below the discharge (Suren et al 2010). The median conductivity of groundwater being discharged into the experimental channel from bores K4 and K6 was c. 850 µS cm^{-1}. The median conductivity in the experimental channel during the experimental period (including floods) was equivalent to 63% of the groundwater discharge, close to the predicted 2060 scenario dilution. When the diluting influence of floods was removed, the median conductivity was higher and close to the predicted 2060 scenario (67%), although variable (ranging from 38% to 78% dilution).

The median pH of water in the control channel was 6.9, and generally ranged from 6 to 7 (Figure 3-2). Daily pH fluctuations were most likely attributable to algal respiration at night increasing dissolved carbon dioxide and decreasing pH. There were two periods where there were increases in water pH greater than the daily fluctuations: on 26-27 February pH reached 9.1 and on 24-25 March pH reached 8.3. These changes do not appear to be due to changes in stream flow or temperature and may be due to the presence of algae on the sensor. Water pH was slightly higher in the experimental channel (median = 7.7; paired *t*-test = 122.2, P < 0.001), but had a similar range (min = 6.41, max = 8.21 excluding 3 outliers). There was a highly significant correlation between pH in both channels, with pH in the experimental channel being on average 0.7 units higher than in the control channel.

The influence of the discharge on water temperature (Figure 3-2) was less dramatic than for conductivity and pH. Median temperature in the experimental channel was 15.1^{o}C, slightly lower than in the control channel (15.4^{o}C). Although this difference was highly significant (paired t-test = 30.2, P < 0.001), it was unlikely to have any ecological significance. Water temperature varied diurnally at both sites, with similar daily minima and maxima.

Dissolved oxygen (DO) was significantly higher in the experimental channel (mean = 10.3 mg l^{-1}) than the control channel (mean = 9.7 mg l^{-1}), but again these differences were unlikely to have any ecological significance. DO also varied diurnally (Figure 16e), and was generally lowest in the early morning and highest in the early evening. Daily fluctuations are caused by photosynthesising algae producing oxygen during sunlight hours, and consuming oxygen during respiration at night-time. Dissolved oxygen concentrations in both channels were at all times well above minimum recommended concentrations of 6.0 g m^{-3} to protect fish early life stages (Franklin, 2010) and also above the guideline of 80% saturation at all times. Daily mean DO saturation exceeded 90% at all times in both the experimental and control channels, indicating no risk to fish life, including trout.

3.1.4 Longitudinal patterns

Significant differences in conductivity were found along the experimental channel (Figure 3-4). Conductivity decreased from its highest value (c. 90% groundwater) at 5 – 10 m from the start of the channel to an average of 64% groundwater between 20 and 9m along the channel. Conductivity was also significantly lower at locations 75% across the channel and close to the plywood baffles next to the river water. These differences were particularly noticeable in the first 5 m of the channel, before with groundwater could fully mix with the river water. Within the experimental section of the channel (i.e., below the upstream 20 m mark), there were no significant differences in conductivity either along or across the channel, which averaged 70% that of groundwater.

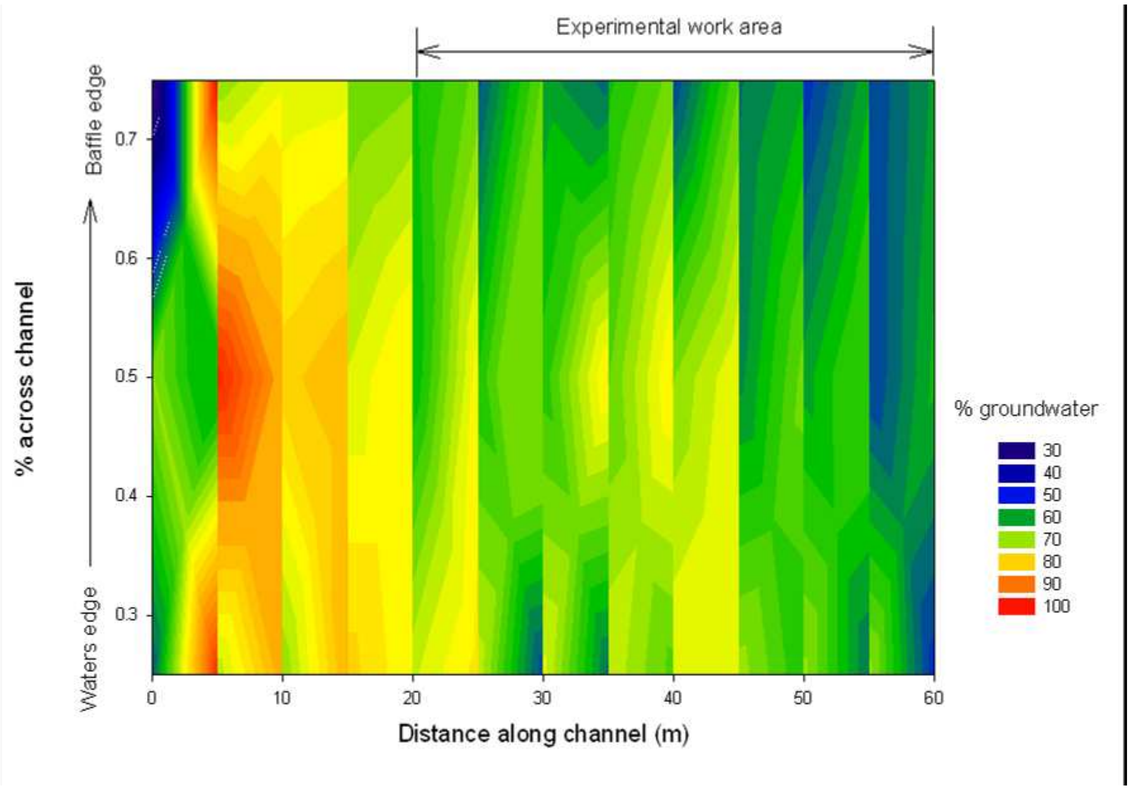

Figure 3-4: Measured conductivity (expressed as the % dilution of the groundwater) along the experimental channel at locations 25%, 50% and 75% across the width of the experimental channel.

3.1.5 Spot water chemistry

Water samples were collected on three occasions during the study: at the commencement of the discharge (18 February, 2011), on the third sampling trip (10 March 2011), and at the conclusion of the study (15 April 2011). Analysis of the percentage dilution of the groundwater showed that bicarbonate, conductivity, alkalinity and ions were diluted to approximately 60% in the experimental channel (Table 3-2). Ammonia and dissolved reactive phosphorus were also diluted by 44 and 67% respectively in the experimental channel. However, nitrate and nitrite concentrations were higher in the control channel when compared to the experimental channel, reflecting the higher nitrogen concentration in the river water (Table 3-2).

Comparison of water chemistry in the experimental channel to that of the control channel showed that bicarbonate, conductivity, alkalinity, and three ions (calcium, magnesium and iron) were five times higher in the experimental channel, while total and dissolved manganese were 78 and 150 times higher, respectively. Concentrations of ammonia and dissolved reactive phosphorus were approximately 1.5 and 5.7 times higher, and the concentration of nitrite was six times. In contrast, the concentrations of nitrate was half as high in the control channel.

Table 3-2: The average concentrations of ions, nutrients and other determinants as collected on three occasions from the groundwater discharge, the control channel, and experimental channel. Also shown is the percentage dilution of groundwater in the experimental channel, and the ratio of determinants in the experimental to the control channel.

Determinant	Groundwater discharge	Control Channel	Expt Channel	% dilution Of bore	Ratio of expt to control
Ionic composition					
Total Alkalinity (g m^{-3} as $CaCO_3$)	187	20.5	113	61	5.5
Bicarbonate (g m^{-3} at 25 ^{o}C)	223	24.7	137	61	5.6
Conductivity mS m^{-1}	85	10.6	53	63	5.0
Dissolved Sodium	129	11	76	59	6.9
Dissolved Potassium	4.4	0.8	2.8	65	3.7
Dissolved Calcium (g m^{-3})	18.9	5.1	12.7	67	2.5
Dissolved Magnesium (g m^{-3})	9.2	1.8	5.9	64	3.4
Dissolved Manganese (g m^{-3})	0.10	0.0003	0.06	53	150
Total Manganese (g m^{-3})	0.12	0.001	0.06	55	79
Total Iron (g m^{-3})	0.11	0.01	0.07	63	4.7
Nutrients					
Ammoniacal-N (g m^{-3})	0.17	0.05	0.07	45	1.5
DRP (g m^{-3})	0.071	0.008	0.048	67	5.7
Nitrate-N (g m^{-3})	0.05	0.14	0.08	138	0.5
Nitrite-N (g m^{-3})	0.002	0.001	0.01	286	6.7
Dissolved Inorganic N	0.22	0.19	0.15		
DIN : DRP ratio	3.1	24	3.3		
Other					
pH	8.0	7.8	7.7	-	-
Temperature (^{o}C)	16.7	16.2	16.4	-	-

A PCA of the water chemistry data showed clear differences in overall chemical signatures between groundwater, and water in the experimental and control channels (Figure 3-5). The experimental channel had an overall chemical signature between that of groundwater and control channel. There was also considerable temporal variability in the chemical signatures of both the groundwater and experimental channel, and much less in the control channel. Despite this variability, at no time did the chemical signatures converge.

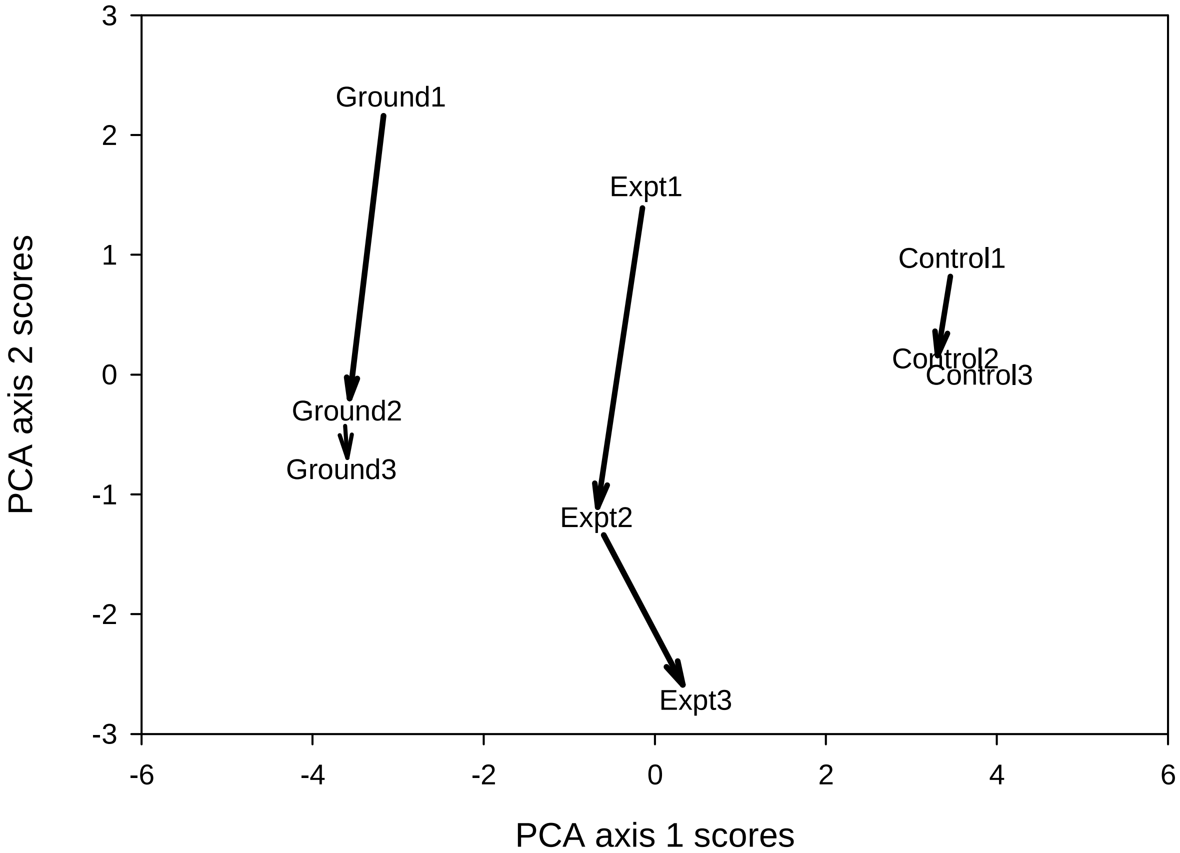

Figure 3-5: Results of a PCA ordination of the water chemical data collected from the groundwater, experimental channels, and control channels on three occasions during the experiment.

The water chemistry of the groundwater, control and experimental channels was compared to river water from the Wellington region and around the North Island (Figure 3-6). This showed that the water chemistry of the control channel was not distinguishable from rivers around Wellington and the North Island generally. The water chemistry of the experimental channel was; however, different to that of most other rivers and was more similar to the Tarawera River, which has significant geothermal input.

Based on the eigenvector coefficients for the PC2 axis, where the greatest difference is seen between the water chemistry of the experimental chan nel compared to other waters (Figure 3-6), the variables driving these differences appear to be sodium, calcium and pH. Bicarbonate and Magnesium ions appeared to be important variables on the PC1 axis.

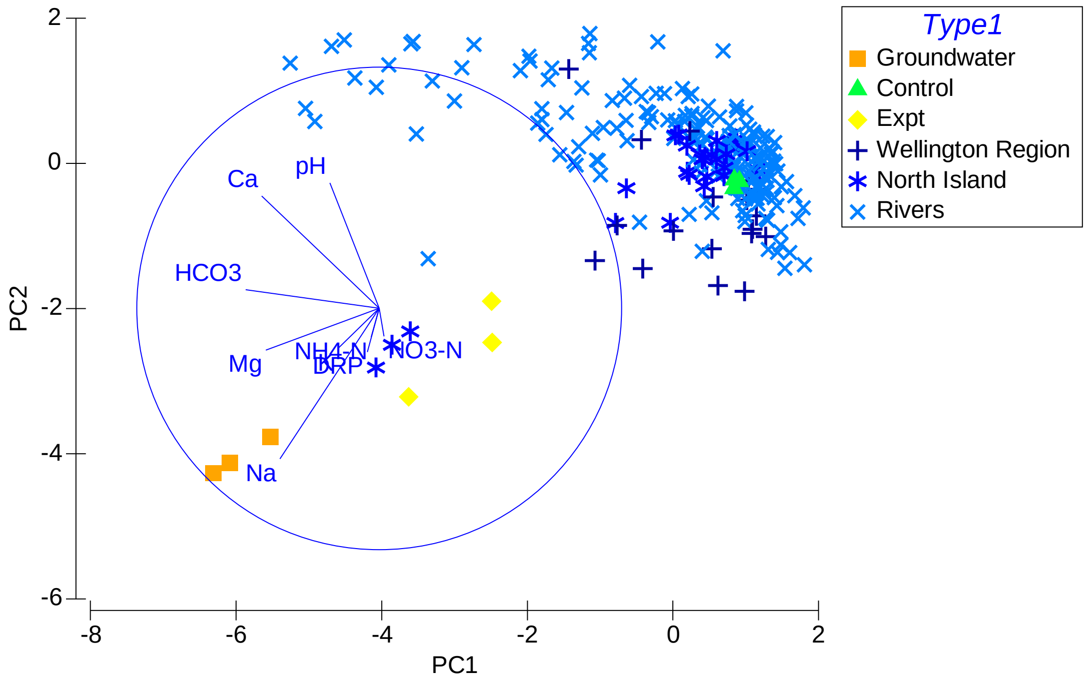

Figure 3-6: PCA of water chemistry of groundwater, experimental and control channels compared to Wellington, North Island and other New Zealand rivers.

3.1.6 Comparison of predicted anion-cation composition to other natural waters

The anion-cation composition of the Waikanae River downstream of the bore water discharges is shown graphically in a Piper diagram in Figure 3-7. Predicted anion-cation composition in the Waikanae River, with the groundwater recharge is shown by crosses for a low-use scenario of groundwater recharge (27% bore water, 73% river water) and stars for the a high-use recharge scenario (70% bore water, 30% river water) expected in a 1 in 50 year drought coupled with a forecast demand in 2060 of 32,000 m^3/day. This shows that under either scenario, the anion-cation composition of the Waikanae River will be altered to become more like the groundwater of the area.

Anion-cation composition is predicted to change the most with input from K4, which has water chemistry least like the Waikanae River; while it is predicted to change the least with input from Kb4, where water chemistry is closer to that of the river.

This predicted Waikanae River anion-cation composition is compared to river water and springs from 13 rivers in the North Island; 5 of which were in the Wellington Region (Figure 3-8). This shows that there is a wide range in the anion-cation composition for the different rivers and springs. In terms of cations, the predicted chemistry of the Waikanae River is not unlike some spring waters. Overall, the anion-cation composition is most like Pauatahanui River in Wellington.

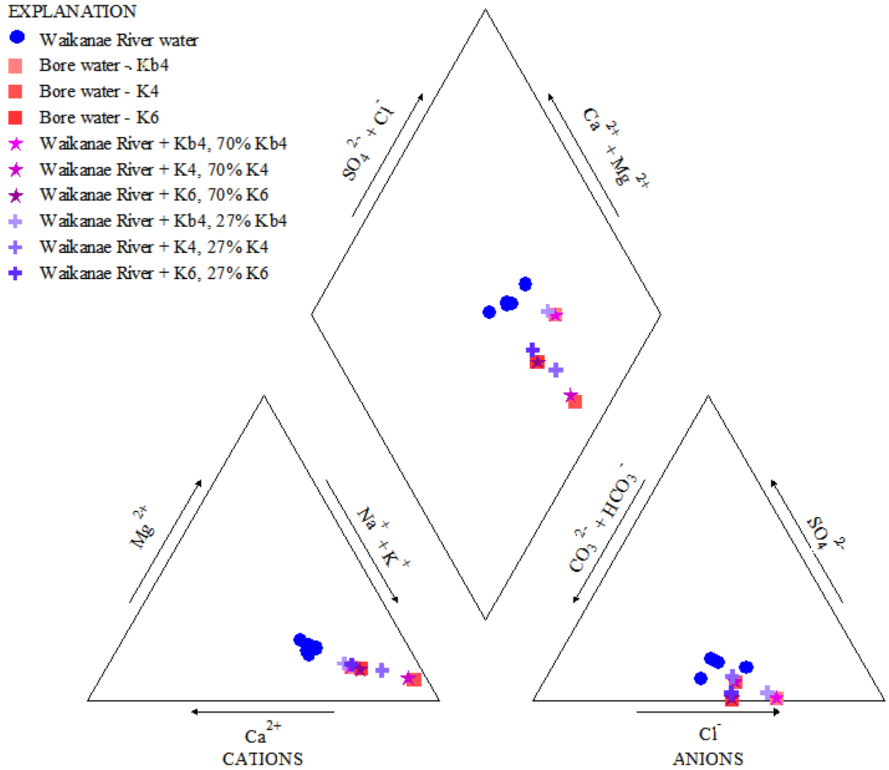

Figure 3-7: Piper diagram showing anion-cation composition of water chemistry from Waikanae River, bore water and river water under two different recharge flow scenarios.

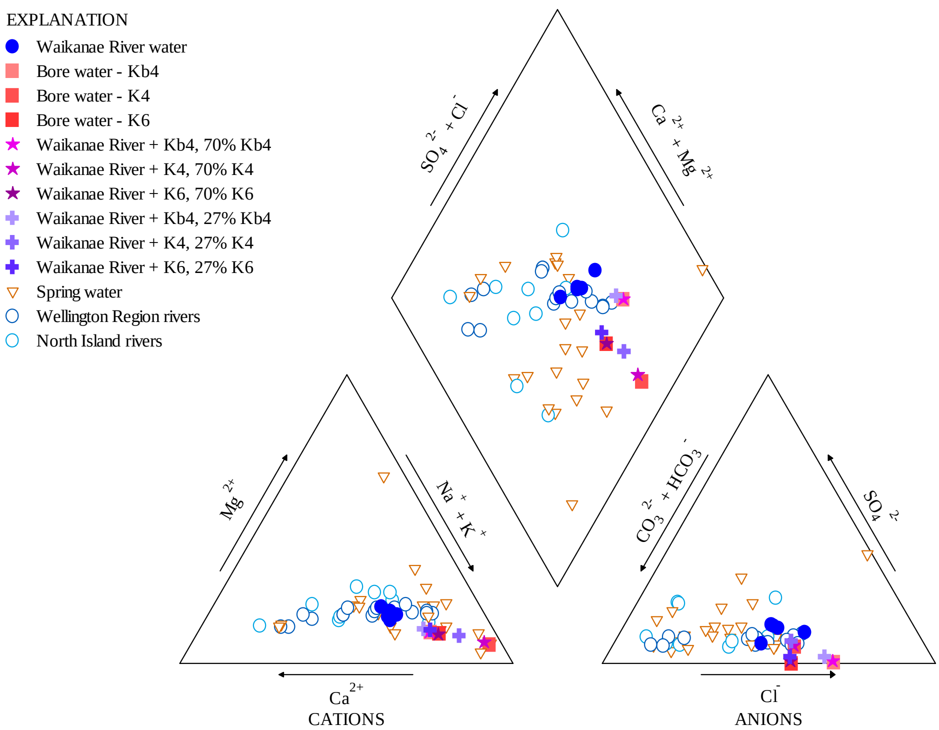

Figure 3-8: Water chemistry from Waikanae River, bore water and river water under two different recharge flow scenarios, compared to North Island rivers and New Zealand springs.

3.2 Algal communities

3.2.1 Chlorophyll biomass

Two-way ANOVA showed that chlorophyll biomass was significantly higher in the experimental channel than the control channel during the experiment (Figure 3-9). Biomass differed significantly over time, and increased considerably in the experimental channel after the commencement of the groundwater discharge. Maximum biomass was reached in the experimental channel on the fourth sampling trip (March 22, 2011) before decreasing prior to the last sampling trip (April 14, 2011). The reduction in biomass reflected a large flood event (44 cubic meters per second) that occurred on April 5.

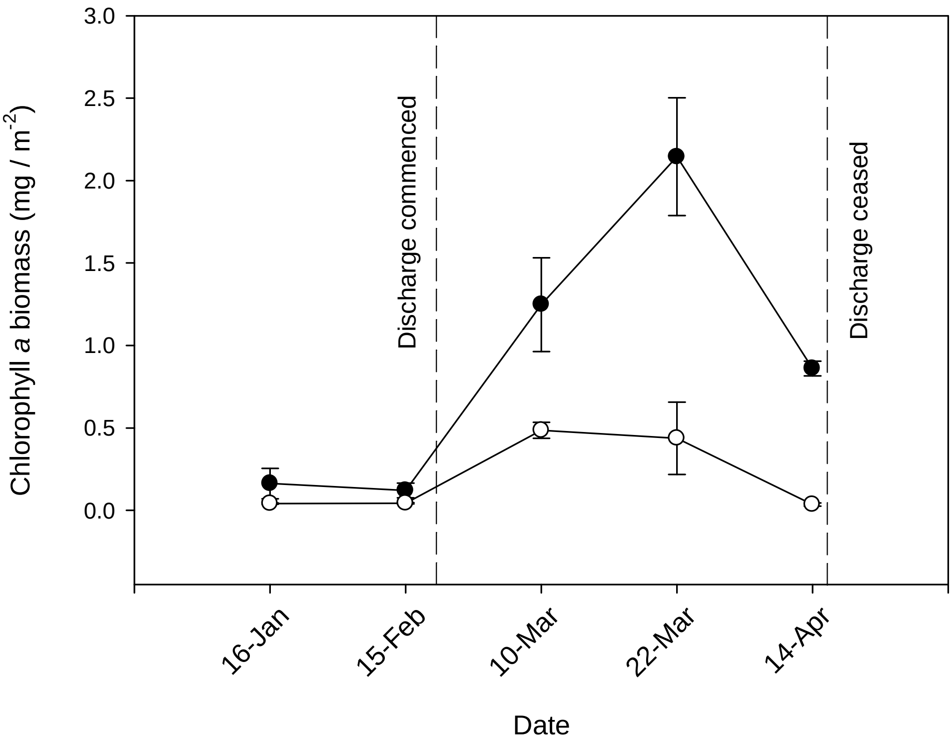

Figure 3-9: Chlorophyll-a biomass (mean ±1 SE) from six randomly selected cobbles in the control (open circles) and experimental (closed symbols) channels collected on five occasions before and during groundwater discharge.

The significant trip × site interaction effect suggested that the temporal trend of chlorophyll biomass was inconsistent between the two channels. Examination of the data showed a much higher increase in the experimental channel on trips 3 and 4 than in the control channel (Figure 3-9), despite starting off with a similar biomass. This increased biomass in the experimental channel compared to the control channel could have been caused by either the larger substrate size in the experimental channel (see Figure 3.1), or as a result of the ground water discharge. The experimental channel also had a much larger reduction in biomass as a result of the flood in early April, reflecting the greater degree that thick algal mats are sloughed from the streambed following floods than thinner diatom films.

Visual examination of algae in the experimental channel showed that this biomass was dominated by a mixture of cyanobacterial mats and filamentous green algae (Figure 3-10). Much of this mat material within the channel was covered with a thin grey film of fine sediment. This was most likely associated with fine particles of silt that were discharged in the first few days of pump operation. This silt discharge appeared short lived however, and was not visible for the remainder of the experiment.

Although these results may have implied that algal cover, and biomass in the experimental channel increased as a result of the discharge of groundwater, a high cover of algae was observed on large stable substrates in the main river outside the experimental channel (Figure 3-11). Detailed conductivity measurements from the river side of the baffle into the main river above these thick algal mats showed no evidence that groundwater was mixing with the river water, as conductivity was always low (11.2 mS m^{-2}) and stable in this part of the channel.

This observation of high algal cover outside the experimental channel was confirmed by comparison of chlorophyll data collected from the experimental channel, and the main river on the third sampling trip (March 10). Chlorophyll biomass was obtained from five stones randomly selected from both the experimental channel and river, and compared using a *t*-test. This showed that chlorophyll biomass was almost double in the main channel (2.6 mg cm^{-2}) than in the experimental channel (1.2 mg cm^{-2}; $t = 2.58$; $P < 0.03$). The high cover of cyanobacteria mats in the experimental channel during the time that groundwater was discharged thus appears unrelated to the discharge per se, but more to the combination of low, stable flows, and large substrates upon which the cyanobacteria could grow.

Chlorophyll biomass at the two lower sampling sites followed the same temporal trend: viz, low biomass in January, increasing to high biomass in late March, followed by a reduction in early April as a result of the flood event (Figure 3-12). A two way ANOVA showed no significant difference in chlorophyll biomass in the experimental channel, the site below State Highway 1, or the site at Jim Cooke reserve. These lower sites were well below the discharge point, and unlikely to be influenced by the effect of the groundwater discharge. The high biomass at these sites simply reflected the normal summer increase in cover of cyanobacteria. This increase was found throughout the river, and did not appear to be caused by groundwater discharge.

Interesting comparisons existed in the percentage change of chlorophyll biomass between the 1st and 4th sampling occasion, and this and the last sampling occasion in the experimental channel and the two lower sites. Biomass increased from January 16 to March 22 by approximately 90% in both the experimental channel and the State Highway site, and by 80% at the Jim Cooke site. Following the flood on April 5, biomass declined at the lower sites by approximately 20 times, whereas biomass declined in the experimental channel by only approximately 2.5 times. This was shown by a highly significant site × time interaction effect in the 2-WAY ANOVA. The much lower biomass reduction in the experimental channel most likely reflected the effect of the baffles on flow regimes in the channel. This structure would undoubtedly have reduced instream velocities and bed shear stress, meaning less algae was scoured away. This was why relatively more algae was found in the experimental channel at the end of the experiment, and was highly unlikely to have reflected any biomass increase as a result of the groundwater discharge.

Figure 3-10:Photograph of cyanobacterial mats and filamentous green algae as found growing in the experimental channel. Note the amount of fine grey sediment trapped amongst the cyanobacterial mats.

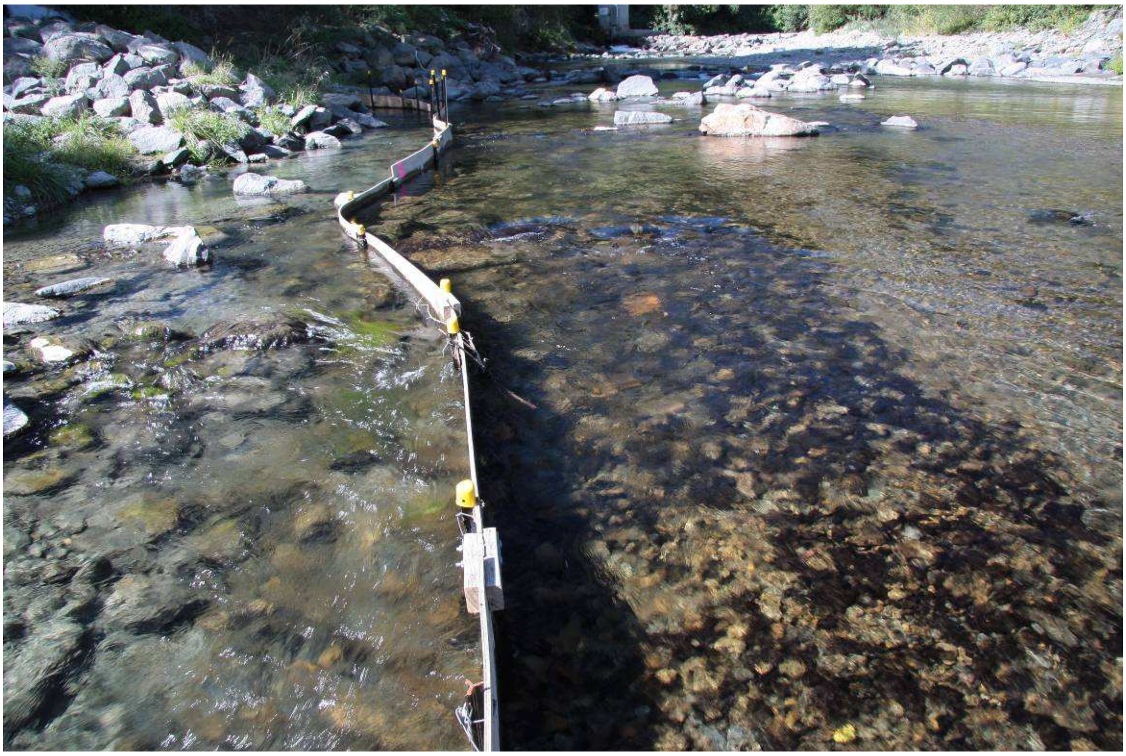

Figure 3-11:Photo of the upstream part of the experimental channel (left-hand) and the main river channel (right-hand) showing prolific cyanobacterial growth in the main channel, in the absence of groundwater discharge. Chlorophyll biomass here was almost double that in the experimental channel, despite not being exposed to any groundwater.

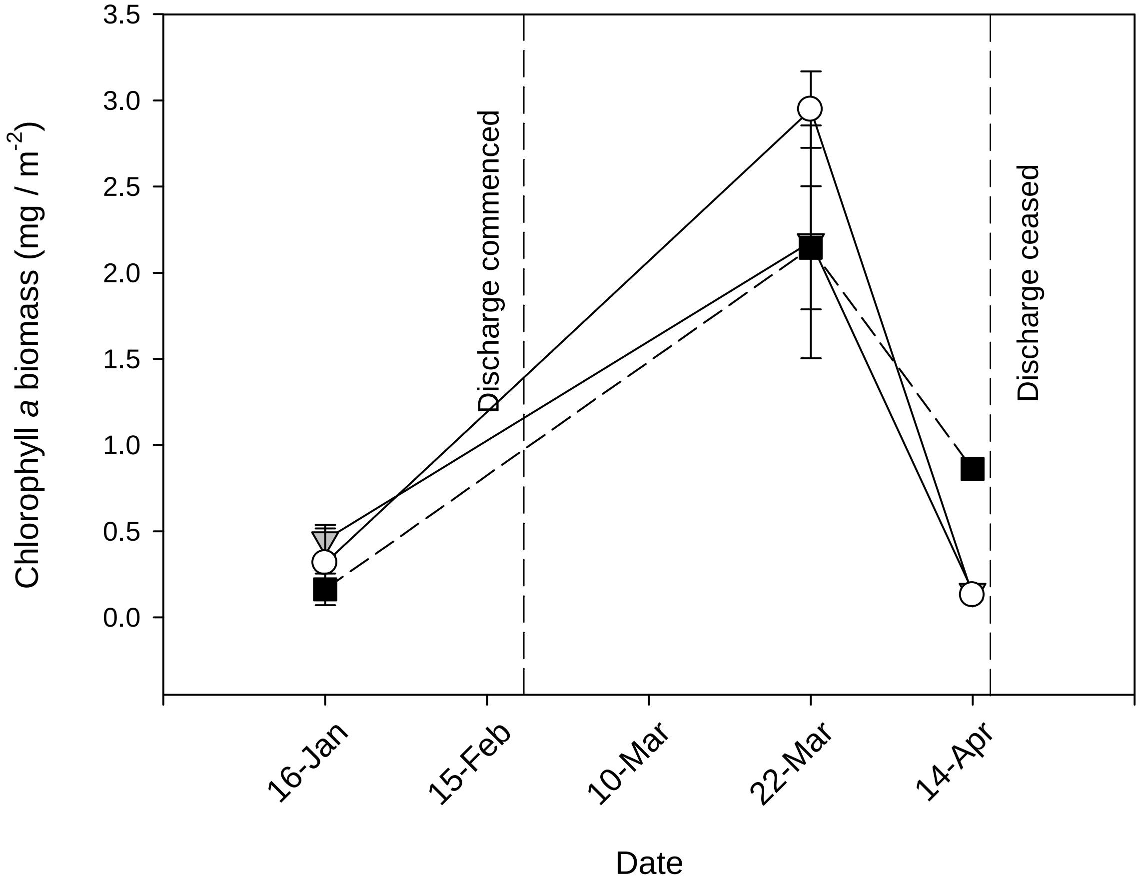

Figure 3-12:Chlorophyll-a biomass (mean ±1 SE) from six randomly selected cobbles collected from the Jim Cooke Park site (open circles) and State Highway 1 site (grey triangles) and from the experimental channel (black squares), before and during groundwater discharge. Note that some symbols are obscured.

3.2.2 Percentage cover

Algal cover was dominated by thin diatom films (51% of observations), followed by medium cyanobacterial mats (21% of observations). The next most common observations were of stones covered by either medium diatom films or no algal cover (both *c.* 12% of observations), followed by filamentous green algae (c. 4% of observations). Two-way ANOVA showed significant differences in cover of thin diatom films and cyanobacterial mats between the different sites. Cover of thin films was highest at the Jim Cooke Park, as well as the control channel and downstream site, and lowest in the experimental channel. In contrast, cyanobacteria cover was highest in the experimental channel, the State Highway 1 site, and the downstream site (average = *c.* 40%), and lowest at the Jim Cooke site, the control channel in the upstream site (average = *c*. 10%). There was no difference in cover of thin films between the two observation periods (before and after a flood on April 5 2011), whereas the cover of cyanobacterial mats had decreased significantly following the flood from approximately 45% cover to 10%.

There was a significant interaction effect on the cover of thin algal films between the locations over time. For example, cover of thin diatom films increased at all sites between observation periods except in the experimental channel, where it declined. This suggested that cyanobacterial mats had sloughed from the streambed at most sites following the flood and been replaced with thin diatom films. Examination of the data showed a large reduction in cyanobacteria cover at the State Highway and Downstream sites, where cover had decreased from approximately 90% to 10% (State Highway) and 50% to 10% (downstream site). Although cyanobacteria cover decreased at the other sites, the magnitude of this reduction was not as great. For example, cover of cyanobacterial mats in the experimental channel decrease by twofold (56% to 22%), and in Jim Cook Park by threefold (19% to 6%).

3.2.3 Nutrient diffusing substrates

Some filter papers on top of the agar containers were washed away, presumably by the flood on April 5. However, algae had subsequently grown on the tops of the agar (Figure 3-13), therefore this was scraped off and analysed for chlorophyll. To determine if agar alone interfered with the chlorophyll determination, we also ran this analysis using agar blanks of the control and nutrient combinations. Intact filter papers were also removed and the adhering algae was analysed for chlorophyll.

Figure 3-13:Representative photograph of one of the nutrient diffusing substrate trays placed in the Waikanae River for three weeks. Note the loss of filer papers from many of the agar containers. Where this had occurred, algae was scraped from the tops of the agar, and analysed for chlorophyll.

Loss of the filter papers on top of some of the agar containers meant that we could not use the chlorophyll data from all nutrient treatments in a single analysis, and so we were unable to compare the effect of nutrient treatments within each channel. In other words, we could not unequivocally determine whether periphyton growth in the Waikanae River was nutrient limited. However, there were enough samples of different nutrient treatments (control, phosphorus, and nitrogen plus phosphorus) from both the control and experimental channels to determine whether algal biomass was responding to nutrient treatments in the different channels, because we could compare chlorophyll biomass on the no-nutrient agar containers in the control and experimental channels. This allowed us to determine whether there were differences in chlorophyll growing in the control channel (low nutrients) and experimental channels (high nutrients). If algal growth was nutrient limited, then we would expect more algal growth in the experimental channel.

A paired t-test showed no significant difference in chlorophyll from algae scraped off the unenriched agar placed in either the control (mean = 0.060 mg cm^{-2}) or experimental channel (mean = 0.054 mg cm^{-2}), suggesting that algal growth in the Waikanae River was not strongly limited by nutrients.

3.2.4 Algal Communities: summary

Algal biomass increased in both the experimental channel and the main river, despite the absence of groundwater in the latter. Cover and biomass also increased in the lower sites, despite being exposed to only very diluted groundwater. The nutrient diffusing study also implied that algal growth was not strongly nutrient limited. High algal cover thus appears unrelated to groundwater discharge, but more to a combination of low, stable flows, warm temperatures, and large stable substrates. The effects of a autumn flood in reducing cyanobacterial cover were also clearly evident, part of the river's natural cleansing process.

3.3 Invertebrate communities

A total of 74 invertebrate taxa were recognised during the survey. The fauna was dominated by the mayfly *Deleatidium* (24% total abundance) and Orthocladinae midges (21%), as well as *Tanytarsus* midges (15%), Elmidae riffle beetles (13%), and the caddisfly *Pycnocentrodes* (11%). Other common taxa (abundances between 1 and 5%) included oligochaete worms, the common freshwater snail *Potamopyrgus*, aquatic mites, and the caddisflys *Olinga* and *Bareoptera*. Average invertebrate densities were relatively high (c. 14,000 individuals per square metre), and samples contained an average of 22 different types of animals (i.e., taxa). Calculated biotic metrics were all high, with the average MCI (110) and average QMCI (5) suggesting streams in good ecological condition (Stark 1993). There were on average 11 EPT taxa per sample, and the average percentage EPT was 47%. Such a diversity of EPT, and high relative abundance of EPT again highlights the generally good ecological condition of the Waikanae River at the treatment plant, as well as at the State Highway 1 and Jim Cooke sites, as highlighted by Suren et al (2010).

3.3.1 Control and experimental channels

Two-way ANOVAs were conducted on six biotic metrics, as well as densities of the 10 most common taxa. Of the 16 analyses done, significant site × time interactions were observed in only 4 instances (Table 3-3), indicating that only a few metrics behaved differently in the control and experimental channel over time. The number of EPT taxa, and densities of *Deleatidium* and *Olinga* increased significantly in the control channel, but did not change over time in the experimental channel (Figures 3.14, 3.15, 3.16). The percentage EPT did not change in the control channel over time, but decreased more in the experimental channel after addition of groundwater. These results tended to suggest that conditions in the experimental channel had become less favourable to taxa such as *Deleatidium* and *Olinga*, and to EPT metrics during the discharge of groundwater.

Table 3-3: Results of two-way ANOVAs for the six biotic metrics, and densities of the 10 mot common taxa, showing significant site effects (Expt = experimental channel, Cont = control channel) and significant timing effects. The table also shows the nature of significant Site x Time effects. N.s. = not significant.

Biotic metric	Site effect	Timing effect	Site x Time internation effect
Invertebrate densities	n.s	After > Before	n.s
Taxonomic richness	Expt > Cont	After > Before	n.s
MCI	n.s	Before > After	n.s
EPT	Expt > Cont	After > Before	Expt: n.s Cont: After > Before
QMCI	Cont > Expt	Before > After	n.s
Percentage EPT	Cont > Expt	Before > After	Exp: Before > After Cont: n.s
Acarina	Cont > Expt	n.s	n.s
Beraeoptera	n.s	n.s	n.s
Deleatidium	Cont > Expt	n.s	Expt: n.s Cont: After > Before
Elmidae	Cont > Expt	n.s	n.s
Oligochaetes	n.s	After > Before	n.s
Olinga	n.s	n.s	Expt: n.s Cont: After > Before
Orthocladinae	Expt > Cont	After > Before	n.s
Potamopyrgus	Expt > Cont	n.s	n.s
Pycnocentrodes	Expt > Cont	n.s	n.s
Tanytarsus	Expt > Cont	After > Before	n.s

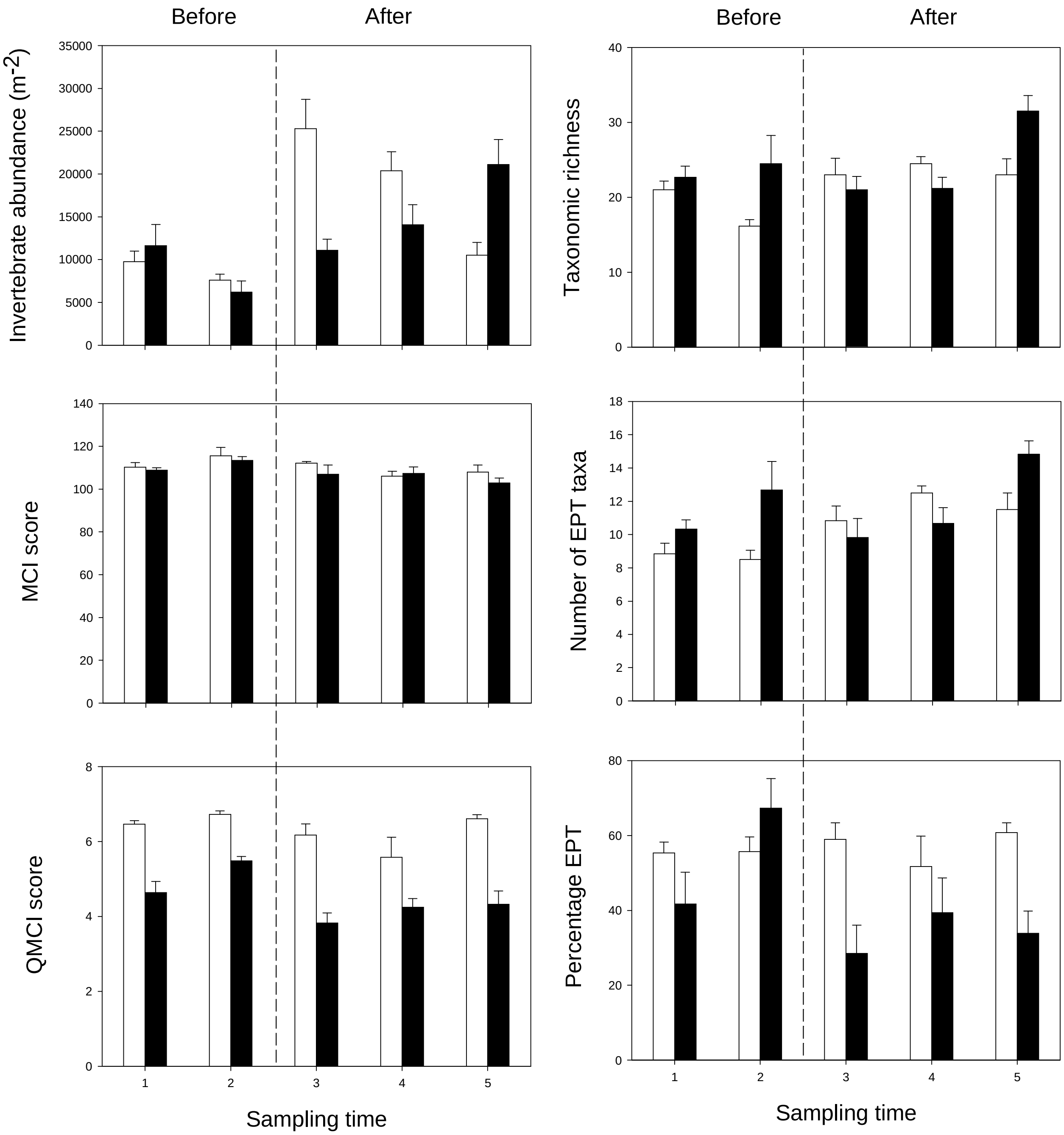

Figure 3-14: Total invertebrate abundance, taxonomic richness and calculated biotic metrics summarising the invertebrate communities collected from control (open symbols) and experimental (closed symbols) channels in the Waikanae Before and After addition of groundwater (x ± 1se, n = 5).

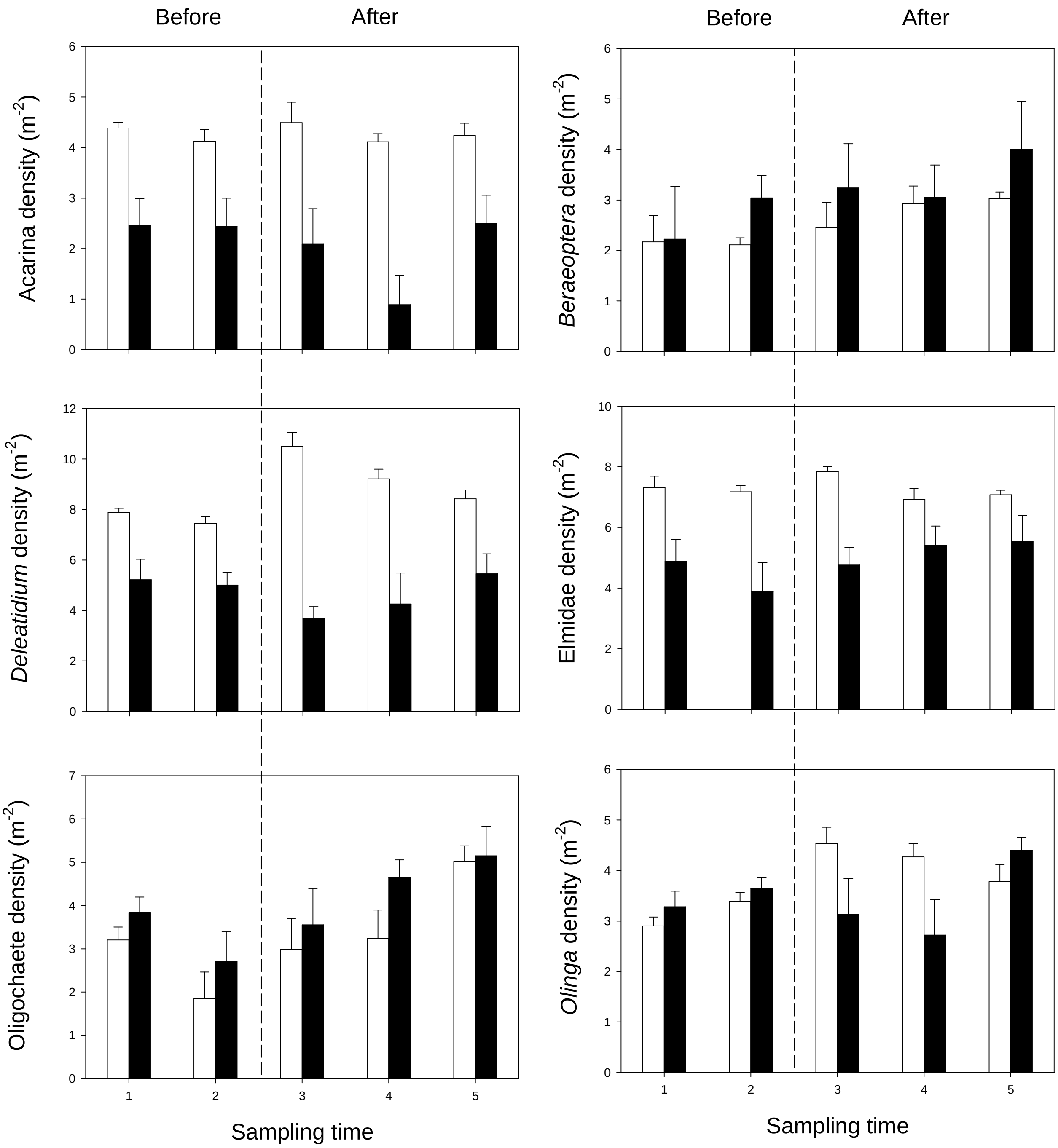

Figure 3-15: Densities of the most common taxa collected from control (open symbols) and experimental (closed symbols) channels in the Waikanae Before and After addition of groundwater (x ± 1se, n = 5). Note that data for the densities of specific taxa have been fourth-root transformed for clarity.

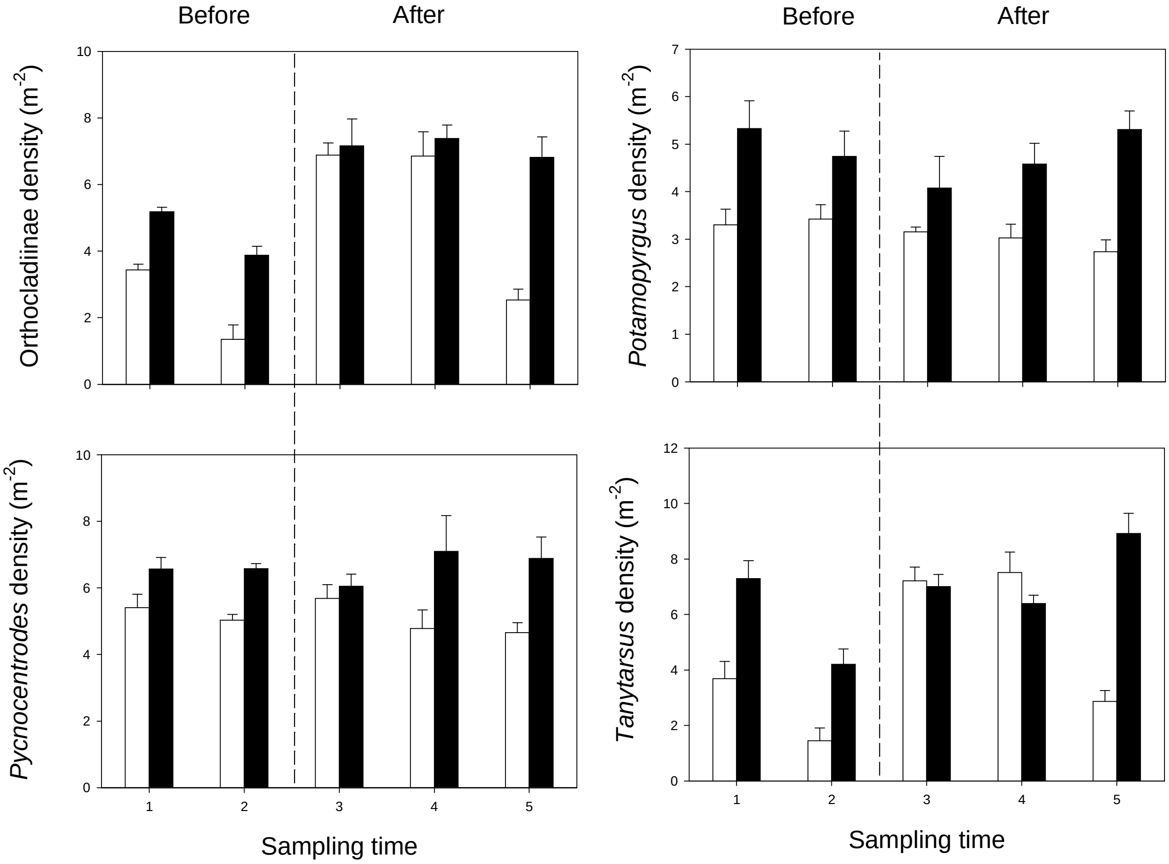

Figure 3-16: Densities of the most common taxa collected from control (open symbols) and experimental (closed symbols) channels in the Waikanae Before and After addition of groundwater (x ± 1se, n = 5). Note that data for the densities of specific taxa have been fourth-root transformed for clarity.

Despite the apparent reduction of some EPT taxa in the experimental channel over time, results of the NMDS ordination showed that behaviour of the overall invertebrate community composition in the control and experimental channels appeared similar. The ordination showed that community composition differed between the experimental and control channels before the discharge of groundwater (Figure 3-17), and community composition changed over time in both channels. Visual observation showed that the magnitude of this change appeared relatively constant between the control and experimental channels (Figure 3-17).

This conclusion was supported by results of the ANOSIM. Here, sites were allocated to one of 4 groups based on their location in either Control or Experimental channels, as well as collection time either Before or After the groundwater discharge. ANOSIM showed the magnitude of difference in the pairwise comparisons of samples collected from these 4 groups. Within-channel similarity over time was relatively high and comparable in both the control and experimental channels over time, as shown by the low R-statistic (Table 3-4). This suggests that the invertebrate communities in both the control and experimental channels changed by a similar amount over time. In other words, the addition of groundwater to the experimental channel appeared to have little extra effect on the behaviour of the invertebrate communities, which changed to the same extent as those in the control channel. Larger differences occurred in community composition between the two channels, both before the groundwater discharge, and after (Table 3-4). Such differences most likely reflect inherent differences in environmental factors between the 2 channels such as the smaller substrate size, shallower water and slower velocities in the control channel.

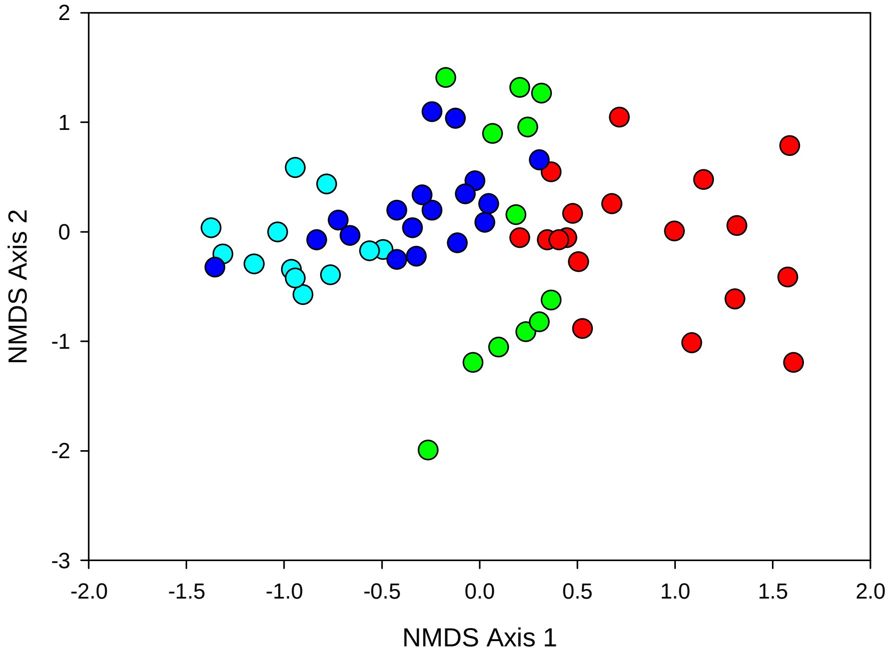

Figure 3-17:NMDS ordination of invertebrate data collected from the control channel before (light blue symbols) and after addition of groundwater (dark blue symbols), and from the experimental channel before (green symbols) and after (red symbols).

Table 3-4: Results of ANOSIM showing the calculated *R* statistic, indicating the magnitude of difference among pairwise groups of samples collected from control and experimental channels, before and after addition of groundwater. An *R* of 1 indicates that the communities completely differed among groups and an *R* of 0 indicates no difference among groups.

Groups	*R*-statistic	Comparison
BEF-CON,AFT-EXP	0.764	
BEF-EXP,AFT-CON	0.578	
BEF-CON,BEF-EXP	0.57	Between channels, BEFORE
AFT-CON,AFT-EXP	0.45	Between channels – AFTER
BEF-CON,AFT-CON	**0.382**	**Within Control channel over time**
BEF-EXP,AFT-EXP	**0.31**	**Within Experimental channel over time**

3.3.2 Upper and lower sites

Two-way ANOVA conducted on the sixteen biotic metrics showed significant site-effects for 11 metrics, highlighting natural differences in abundance of some invertebrates in the four sites. For example, the control channel had the highest or equal highest values for six metrics (Richness, Percentage EPT, Acarina, *Beraeoptera*, *Deleatidium* and elmid riffle beatles), and equal lowest densities of *Pycnocentrodes* and *Tanytarsus*. In contrast, the Jim Cooke Park site had the highest, or equal highest values for percentage EPT and Orthocladinae densities, and lowest or equal lowest values of taxon richness, and densities of *Beraeoptera*, elmid riffle Beatles, *Pycnocentrodes* and *Tanytarsus*. Nine of the calculated metrics showed differences in timing, with five metrics being more abundant after the groundwater discharge had commenced, and four metrics being more abundant before. Only three significant site × time interactions were observed (Table 3-5). In all cases, the behaviour of the communities in the Jim Cooke Park site differed to those at the other sites. Values of the QMCI and percentage EPT declined at this site after the discharge had commenced, whereas these remained relatively unchanged at the other sites. Densities of oligochaetes also increased at this site, and at the State Highway site, but remained unchanged at the other sites.

A NMDS ordination showed that temporal behaviour of the overall community composition in the four sites appeared similar. Despite differences in community composition between four sites, all sites moved in a similar direction and with a similar magnitude before and after the discharge of groundwater (Figure 3-18). ANOSIM was conducted on all possible combinations of paired differences between the four different sites, before and after discharge of groundwater. This showed that relatively large differences existed between the four sites before the discharge (Table 3-6), with the greatest difference occurring between the control and the State Highway 1 sites. Samples collected after the discharge had slightly lower between-site differences, with the least difference occurring between the control site, and the State Highway 1 site (Table 3-6). Examination of within site differences over time showed that the calculated R statistic was relatively small, suggesting that the invertebrate communities within a site did not change much over time, and less than between sites. Of interest was the observation that the communities in the experimental channel had changed the least over time, as shown by their lower R statistic.

Table 3-5: Results of two-way ANOVAs for the six biotic metrics, and densities of the 10 most common taxa, showing significant site effects (Exp = experimental channel, Cont = control channel, SH1 = State Highway 1; JC = Jim Cooke Park) and significant time effects. Similar number superscripts indicate sites with similar means (Post-hoc test, $P > 0.05$). The table also shows the nature of significant (Site x Time effects. n.s. = not significant.

Biotic metric	Site effect	Timing effect	Site x Time internation effect
Invertebrate densities	n.s	After > Before	n.s
Taxonomic richness	Exp^{1} > Cont^{1,2} > SH1^{1,2} > JC^{2}	After > Before	n.s
MCI	n.s	Before > After	n.s
EPT	n.s	After > Before	n.s
QMCI	Cont > all others	Before > After	JC: Before > After Other sites: n.s
Percentage EPT	Cont^{1} > JC^{1,2} > SH1^{2,3} > Exp^{3}	Before > After	JC: Before > After Other sites: n.s
Acarina	Cont^{1} > SH1^{2} > JC^{2,3} > Exp^{3}	Before > After	n.s
Beraeoptera	SH^{1} > Exp^{1} > Cont^{1} > JC^{2}	n.s	n.s
Deleatidium	Con^{1} > SH^{2} > JC^{2} > Exp^{3}	n.s	n.s
Elmidae	SH1^{1} > Con^{1} > Exp^{2} > JC^{2}	n.s	n.s
Oligochaetes	n.s	After > Before	JC: After > Before Other sites: n.s
Olinga	n.s	n.s	n.s
Orthocladinae	SH1^{1} > JC^{1} > Exp^{1} > Con^{2}	n.s	n.s
Potamopyrgus	Exp^{1} > SH1^{2} > Con^{2} > JC^{3}	n.s	n.s
Pycnocentrodes	Exp^{1} > SH1^{1} > Con^{2} > JC^{2}	n.s	n.s
Tanytarsus	Exp^{1} > SH1^{1} > JC ^{2}> Con^{2}	After > Before	n.s

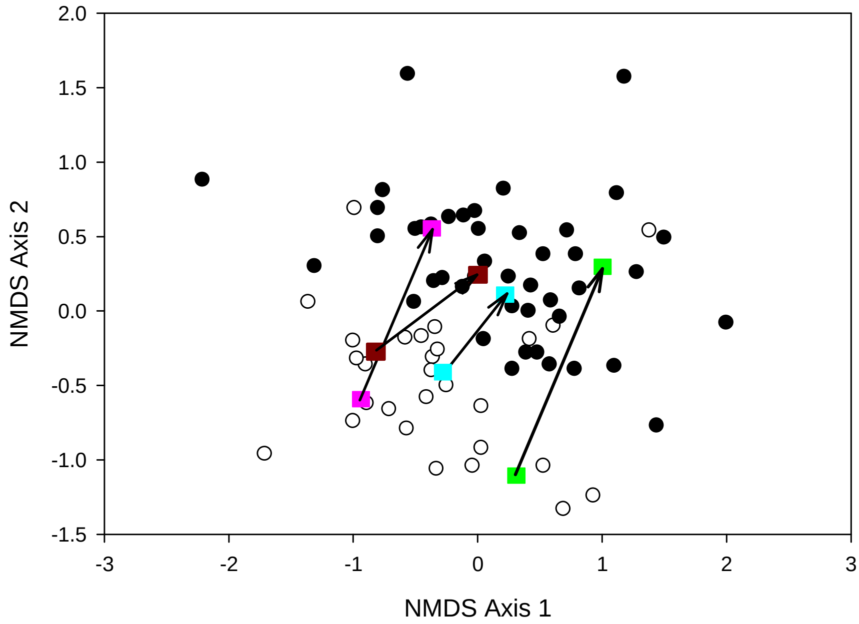

Figure 3-18:NMDS ordination of invertebrate data collected from the control and experimental channels, and from the State Highway and Jim Cooke Park sites before (open symbols) and durng discharge of groundwater (black symbols). Coloured squares show the centroid of samples collected from the control (blue) and experimental (green) channels, and from the State Highway (brown) and Jim Cooke (pink) sites. The direction of change between samples before and during groundwater discharge is shown (arrows).

Table 3-6: Results of ANOSIM of pairwise comparisons fo the four sites before and during the discharge of groundwater showing the calculated *R* statistic. Also shown is the average R statistic for between site comparisons before and during groundwater discharge, as well as within site comparisons over time.

Comparison	Sites	*R* statistics	*P* value	Average
Between site comparisons Before	Cont vs SH1	0.628	0.002	
	Cont vs Expt	0.552	0.002	
	Cont vs Jim Cooke	0.513	0.002	
	Expt vs Jim Cooke	0.513	0.002	
	Expt vs SH1	0.476	0.002	
	Jim Cooke vs SH1	0.465	0.002	0.5245
Between site comparisons After	Cont vs Expt	0.442	0.001	
	Cont vs Jim Cooke	0.391	0.001	
	Expt vs Jim Cooke	0.385	0.001	
	Expt vs SH1	0.324	0.001	
	Jim Cooke vs SH1	0.287	0.006	
	Cont vs SH1	0.215	0.012	0.341
Within site comparisons - Over time	Control	0.478	0.001	
	SH1	0.432	0.003	
	Experimental	0.375	0.004	
	Jim Cooke	0.368	0.001	0.413

3.3.3 Interactions between algae and invertebrates

Scatter plots showed linear relationships between chlorophyll biomass and invertebrate metrics. Regression analyses showed a highly significant positive relationship between chlorophyll biomass and invertebrate abundance, and highly significant negative relationships existed between chlorophyll biomass and the QMCI and percentage EPT (Figure 3-16). Significant positive relationships existed between chlorophyll biomass and densities of Orthocladinae midges, *Tanytarsus* midges, and *Pycnocentrodes*, and a significant negative relationship also existed between densities of mites (negative relationship) and chlorophyll biomass (Figure 3-19, 3-20).

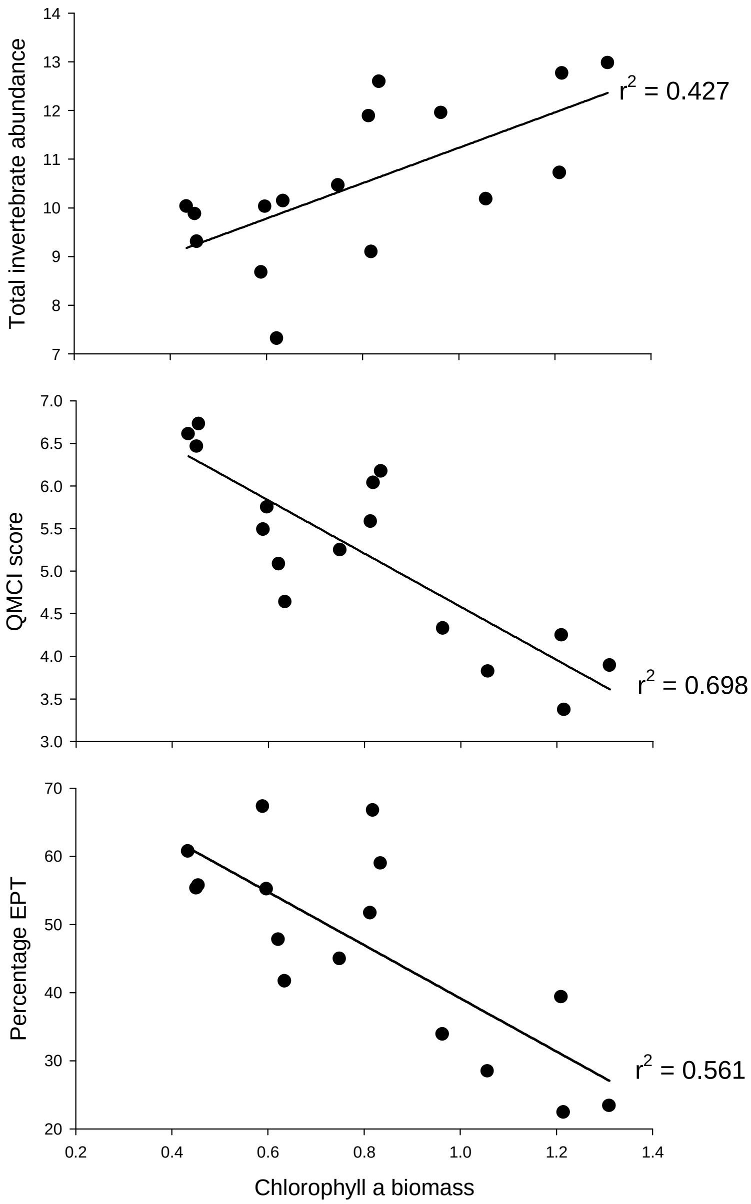

Figure 3-19:Relationships between chlorophyll-a biomass and invertebrate abundance (both 4th-root transformed), and calculated QMCI and percentage EPT. The correlation coefficient for each relationship is also shown.

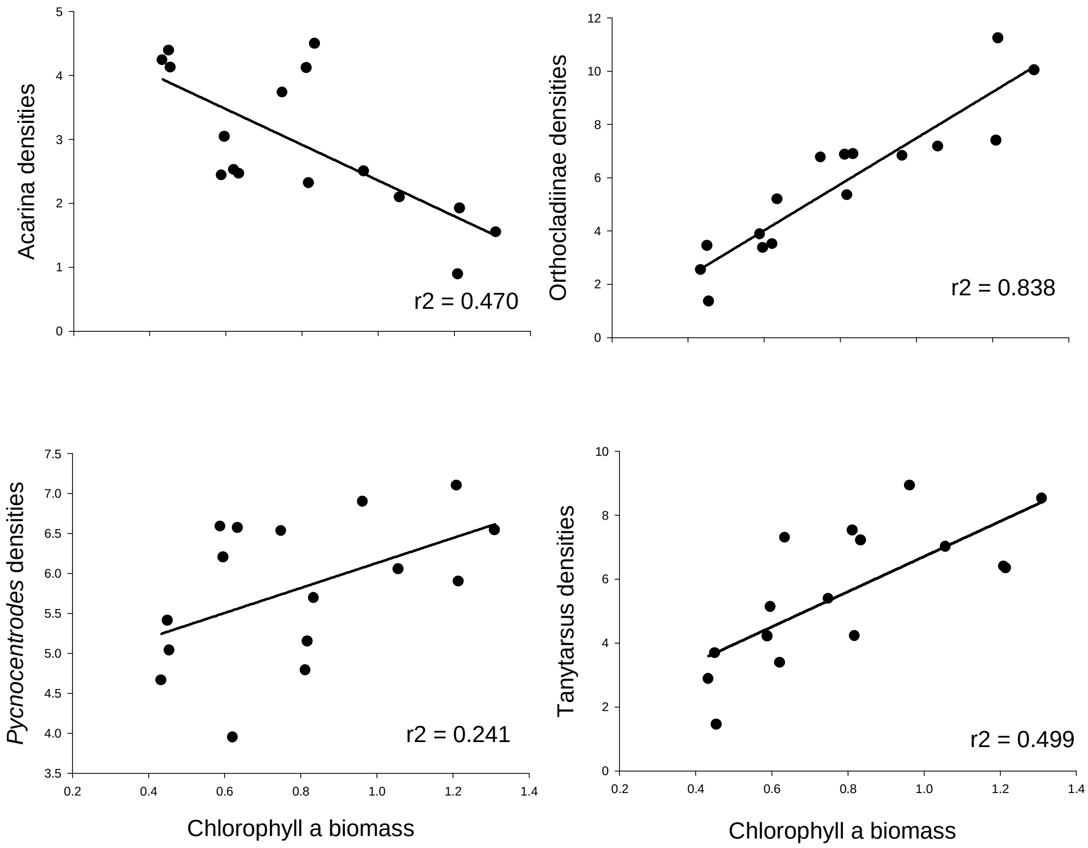

Figure 3-20:Relationships between chlorophyll-a biomass and densities of aquatic mites (Acarina), Orthocladinae, *Pycnocentrodes*, and T*anytarsus* (all 4th-root transformed). The correlation coefficient for each relationship is also shown.

3.2.5 Invertebrate Communities: summary

The invertebrate communities in the Waikanae River were indicative of a waterway in good condition. Addition of groundwater had little extra effect on the invertebrate communities in both channels, and the lower river. Changes to invertebrate communities appeared linked more to changes in algal biomass than to any effects of groundwater.

3.4 Fish

3.4.1 Electric fishing

A slightly larger area of channel was electric fished in the experimental reach (290 m^2) than the control reach (250 m^2). Three species of fish were caught by electric fishing: redfin bullies, longfin eels and torrent fish. Fish densities (numbers per 100 m^2) in the experimental channel were almost double those in the control channel, at both the commencement and conclusion of the experiment, possible reflecting the coarser substrate at the edge of the river where many small fish were found (Pers. Obs). Total fish density increased by 2.5 times in the control channel, and by 1.7 times in the experimental channel during the experiment. Longfin eel density was similar in the control channel over time, whereas density decreased from 12 fish per 100m^2 to only 7 in the experimental channel. Redfin bully density increased by c. 3-fold in both channels over time, presumably as a result of the upstream migration of juvenile bullies during the study. Torrent fish density was low and stable in the control channel, but had declined from 3 fish in the experimental channel at the start to only one fish (Table 3-7).

Table 3-7: Total numbers of fish found in either the control or experimental channel at the commencement of the groundwater discharge (15 February) or the end of the discharge (15 April).

Channel	Time	Area (m^2)	Longfin eels	Redfin Bully	Torrentfish	Grand Total
Control	Start	250	5	16	3	24
	Finish	250	6	49	4	59
Experimental	Start	290	35	31	3	69
	Finish	290	20	96	1	117

Two-way ANOVA showed that the average longfin eel length did not differ between channels, or over time (Table 3.8). There was no channel × time interaction effect either. Average redfin bully lengths were significantly smaller at the conclusion of the experiment than at the beginning, most likely reflecting the increased numbers of young fish that had presumably returned to the river after their brief period at sea as larvae. Average redfin bully lengths were higher in the experimental channel than the control channel. There was no channel × time interaction effect, suggesting that the bore water had no influence on the growth of these fish. There was no difference in the lengths of torrent fish in either channel, or over time.

Table 3-8: Results of 2-WAY ANOVA of average fish length obtained from electric fishing in control and experimental treatment channels over time in the study, and the interaction between treatment and time. Bold values are significant.

Species	Factor	F-ratio	P-value
Redfin Bully	Treatment	4.18	**0.04**
	Time	5.55	**0.021**
	Treatment × Time	3.10	0.082
Longfin eels	Treatment	0.001	0.991
	Time	0.005	0.946
	Treatment × Time	0.240	0.627
Inanga	Treatment	1.45	0.267
	Time	11.12	**0.013**
	Treatment × Time	0.755	0.414

3.4.2 Fish cages

Fish were placed in the cages on the February 18. They were first examined for mortality 20 days later (March 9 2011). Here, only a single bully and longfin eel had died in one of the experimental and control cages, respectively. A second (and final) measurement was made of fish in the cages 12 days after this (March 21), when the cage experiment was terminated due to the risk of forecast rainfall that may have increased river flows and potentially damaged the Fish cages. A further dead redfin bully was observed in an experimental cage on this last sampling trip. Some Inanga had escaped from the cages in both control and experimental cages, with 5 fish escaping from control cages, and 3 from experimental cages after the first sampling occasion. One more inanga had escaped from another experimental cage before the second sampling occasion.

Examination of the data showed no significant difference in growth rates of fish in cages in either the experimental or control channels. Redfin bullies were significantly larger in the experimental channel (Table 3-9), but lengths did not differ over time. Neither longfin eels nor inanga showed any differences between channels, or over time (Figure 3-21).

Table 3-9: Results of 2-WAY ANOVA of average fish length in fish cages placed in the control and experimental treatment channels over time in the study, and the interaction between treatment and time.

Species	Factor	F-ratio	P-value
Redfin Bully	Treatment	10.98	**0.003**
	Time	0.32	0.73
	Treatment × Time	0.44	0.64
Longfin eels	Treatment	2.34	0.14
	Time	3.72	**0.031**
	Treatment × Time	0.56	0.58
Inanga	Treatment	1.18	0.29
	Time	2.78	0.07
	Treatment × Time	1.14	0.33

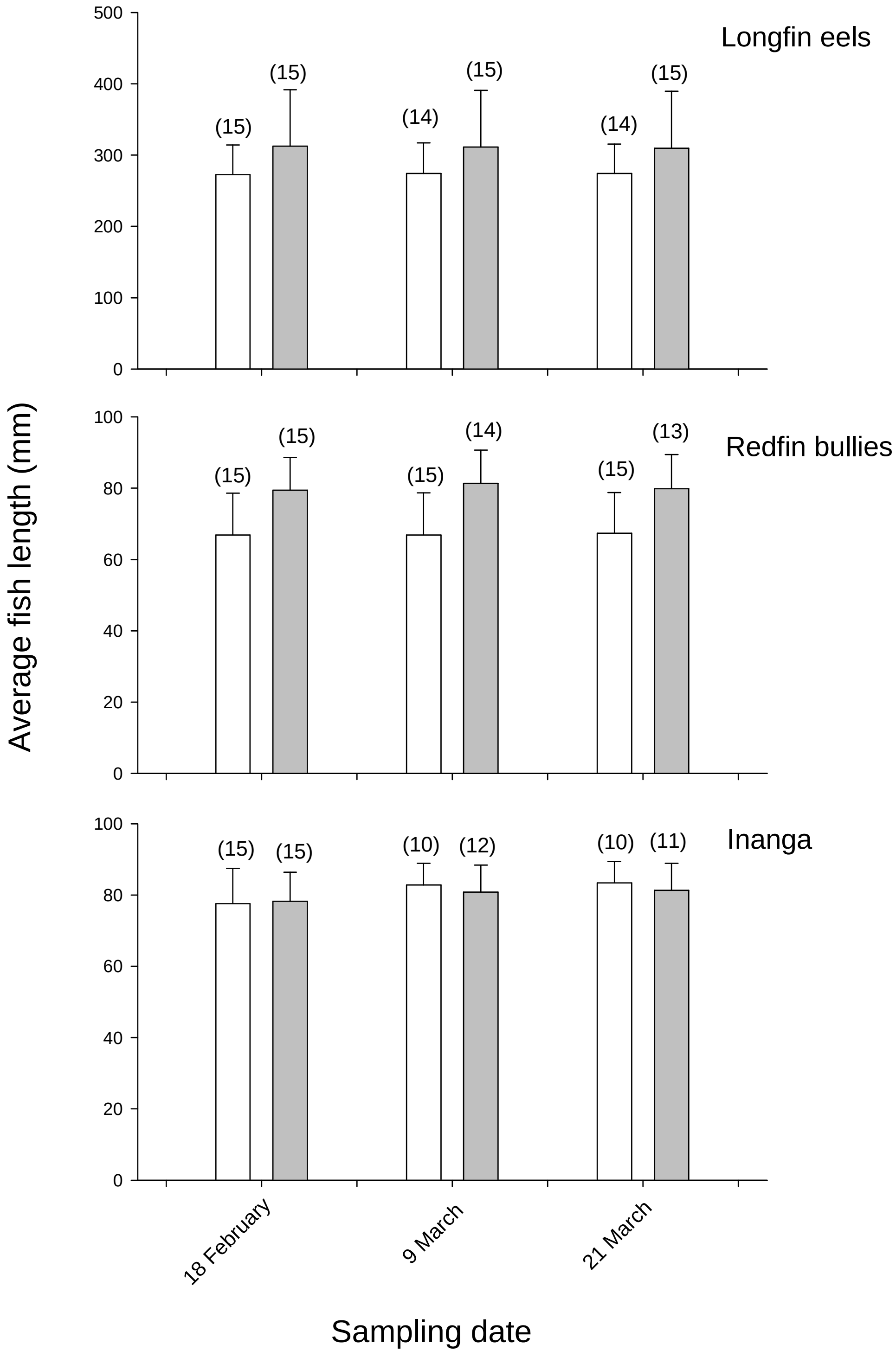

Figure 3-21:Mean lengths (mm) of the three test fish species placed in cages for 32 days in control (open bars) and experimental (shaded bars) channels. The number of fish measured is shown above each bar.

3.4.3 Fish tanks

The fish tank experiment commenced on January 23, and ran until the April 15. Spot measurements were made of dissolved oxygen, conductivity, pH, and temperature in all fish tanks at these times, as well as on March 9 and 23. As expected, significant differences existed in measured conductivity between treatments. Mean conductivity from the 100% groundwater was 852 µS cm^{-2}, whereas mean conductivity from the river water was eight times less (106 µS cm^{-2}). Mean conductivity in the fish tank receiving the mixed water was 735 µS cm^{-2}, or approximately an 85% dilution. Although dissolved oxygen, pH, and temperature differed significantly between treatments, the magnitude of these differences was thought to be ecologically insignificant. For example, water pH varied by less than 0.5 units between treatments, while temperature differed by only 1.2^{o}C, and was warmer in the river water (16.3^{o}C) than the tanks receiving groundwater (15.1^{o}C). Dissolved oxygen was also high in all tanks (average = 99%), and was lower in the bore water (93%) than either the mixed, or river water (101%).

Fish mortality and the number of escaped individuals were noted on three occasions: April 9 and 23 and at the conclusion of the experiment (15 April). The number of redfin bullies and longfin eels recorded in each tank declined dramatically over time, so that at the end of the experiment only a single redfin bully, and four eels remained (Figure 3-22). Loss of redfin bully was attributed mostly to mortality, which increased markedly after March 9. There was, however, no difference in mortality rates between the three water types (Figure 3-23). In contrast, loss of eels from cages was attributed mostly to them escaping, despite best attempts to minimise this (Figure 3-24). Due to the low numbers of redfin bully and eels remaining at the end of the experiment, it was not possible to determine the effect of the different water types on their growth rates.

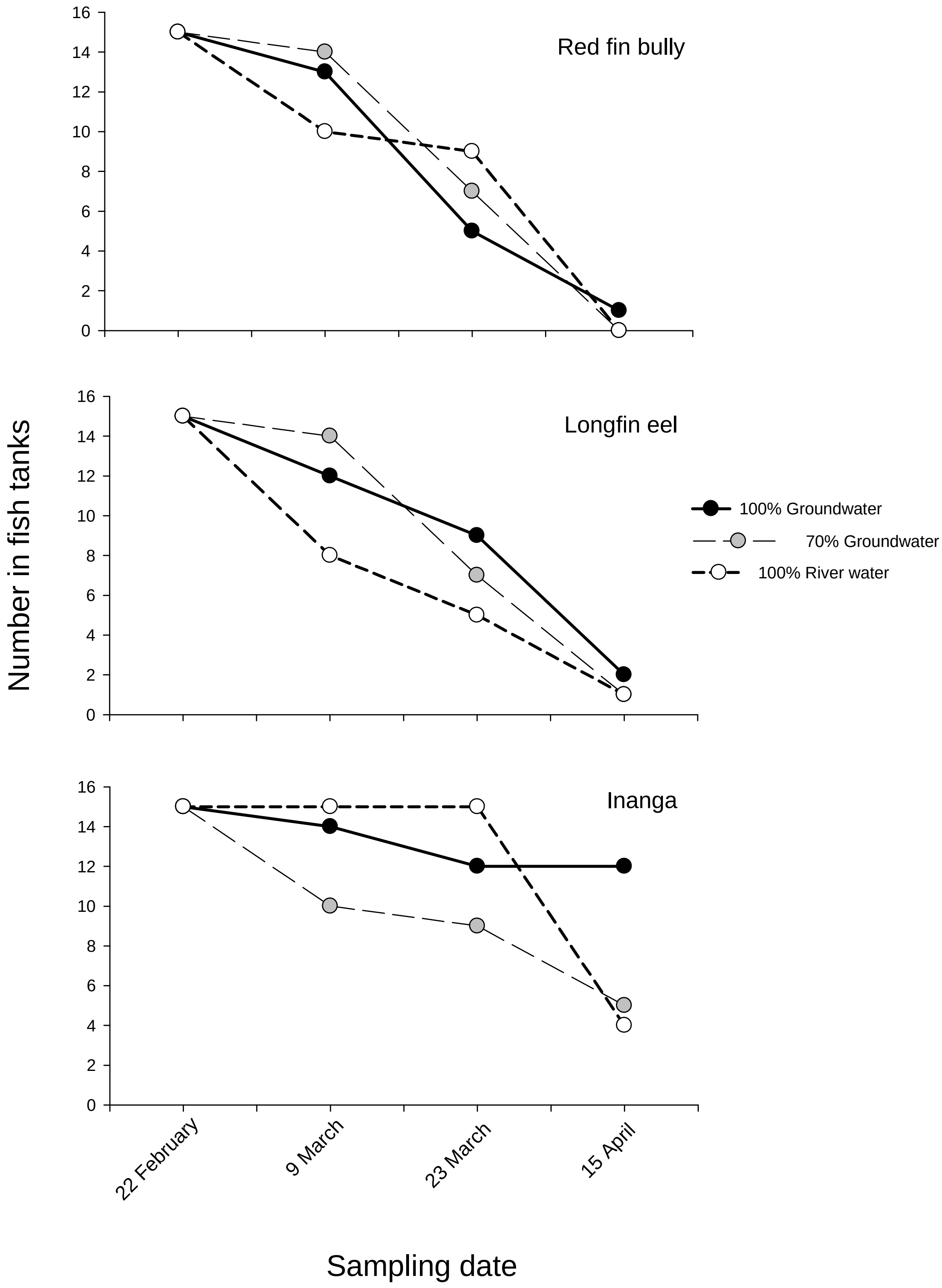

Figure 3-22:Number of fish alive in fish tanks exposed to the different water treatments during the course of the experiment.

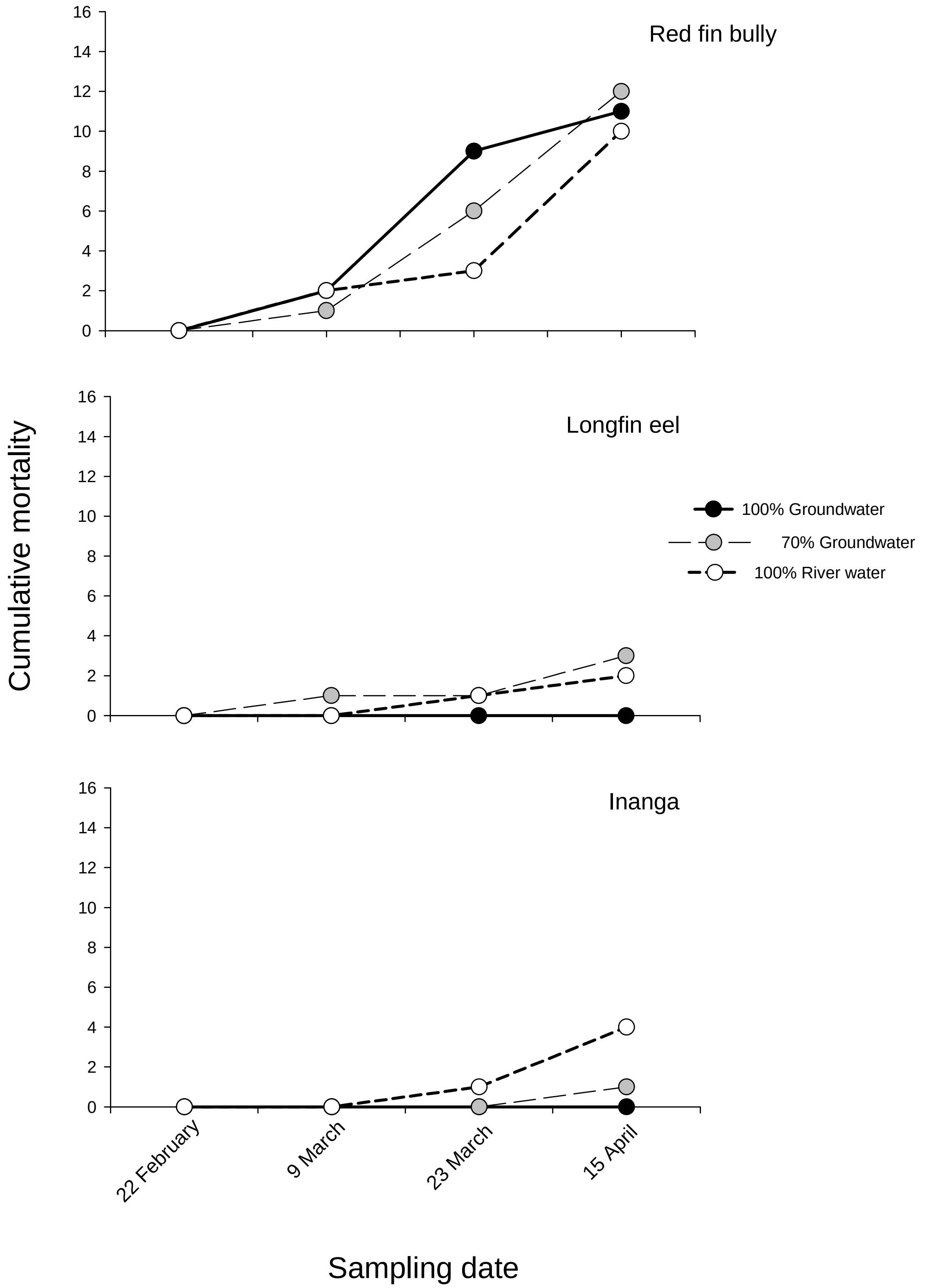

Figure 3-23:Cumulative mortality of fish in fish tanks exposed to the different water treatments during the course of the experiment.

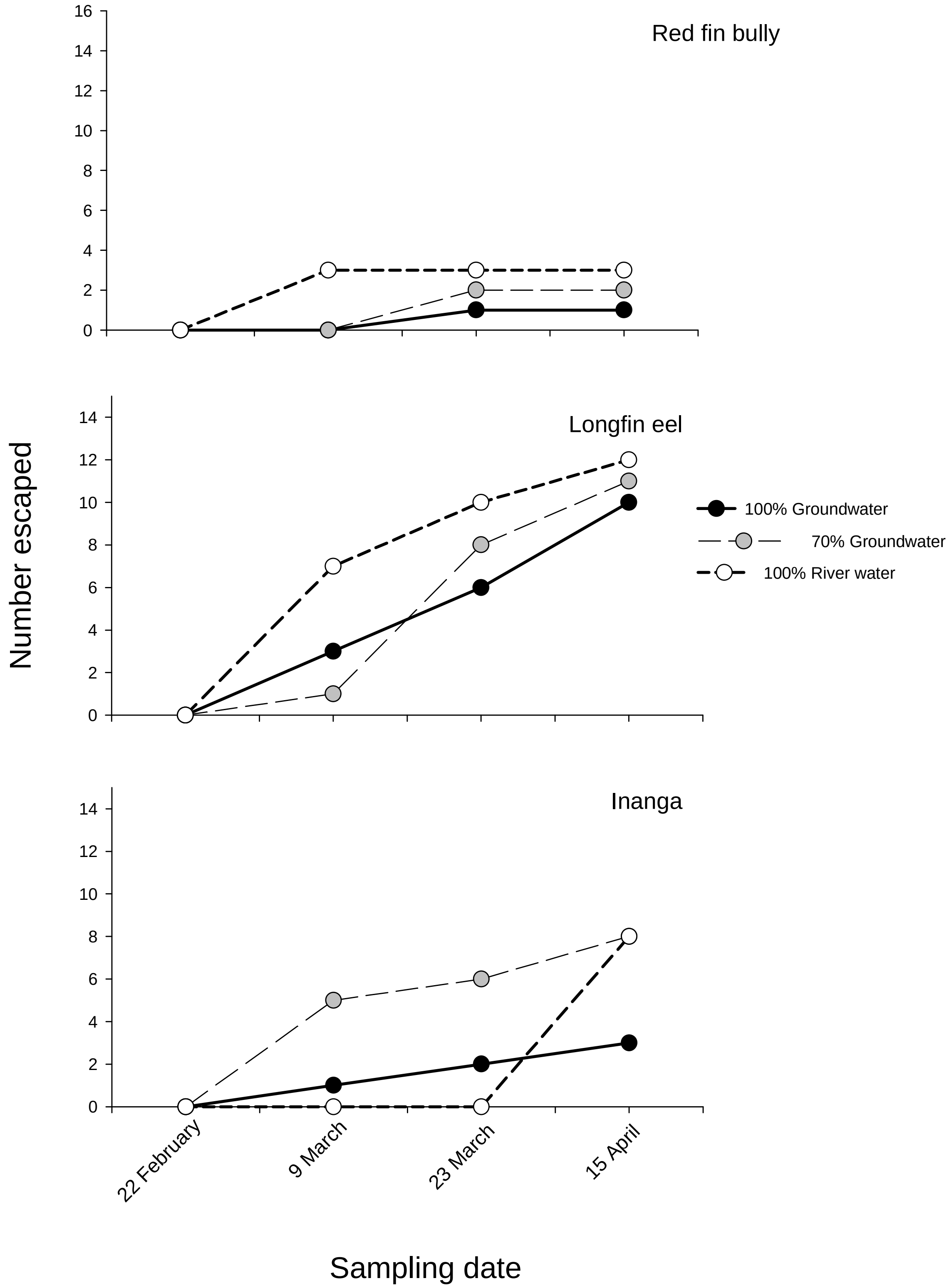

Figure 3-24:Numbers of fish escaped from fish tanks exposed to the different water treatments during the course of the experiment.

Sufficient Inanga remained to allow their growth rates to be compared between the three water types. No statistically significant differences were observed in Inanga length between the three water types, or over time. Furthermore, there was no treatment × time interaction effect on Inanga length. This result suggests that even in 100% groundwater, inanga survival and growth is unlikely to be adversely affected.

3.4.4 Effects on Fish: summary

Addition of groundwater had no demonstrable effects on fish in the experimental channel. No reductions in fish density were observed following groundwater discharge. Results of the fish tank experiment also showed no difference in inanga mortality in either 100% river water, a 30:70 mix of river and groundwater, or 100% groundwater. The RRwGW option is thus highly unlikely to have any effects on fish communities in the Waikanae River.

3.5 Wetlands

3.5.1 Wetland invertebrate communities – comparison to other wetlands

A total of 48 invertebrate taxa were collected from the three wetlands. This fauna was dominated by the common New Zealand freshwater mud snail (*Potamopyrgus antipodarum*), oligochaete worms, the small pee-clam *Sphaerium*, and the common freshwater shrimp *Paracalliope*. Other common invertebrates encountered included a variety of zooplankton such as ostracods, copepods, and *Daphnia*, as well as the common backswimmer *Anisops*. Examination of the 16 most common taxa encountered in this survey (i.e., those taxa with relative abundance greater than 1% total density) showed that 11 of these were also commonly found in other wetlands throughout the North Island.

NMDS ordination of the wetland invertebrate data showed that the invertebrate communities found in the three Waikanae wetlands did not appear particularly unusual when compared with communities found in other wetlands throughout the North Island (Figure 3-25). ANOSIM confirmed the lack of strong regional differences between wetlands, with a very low calculated *R*-statistic (0.08).

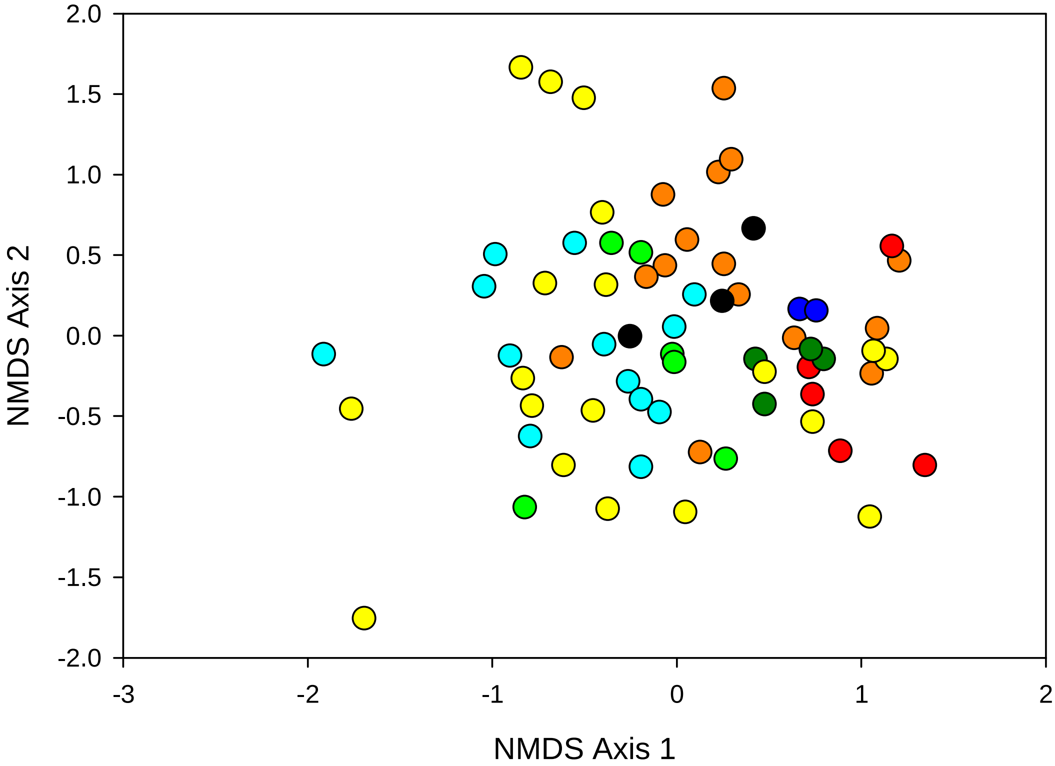

Figure 3-25:NMDS ordination of invertebrate data collected from 67 wetlands throughout the North Island, including the three wetlands sampled in the Waikanae region (black circles). Note the high degree of overlap between wetlands from different regions, and the location of the Waikanae samples and well within the sample clusters.

4 Discussion

4.1 Changes in water quality

4.1.1 Physico-chemical water quality

The continuous water monitoring data highlighted differences in conductivity, pH, temperature and dissolved oxygen between the experimental and control channels, indicating that groundwater discharge will change these water quality parameters in the Waikanae River below the discharge if the proposed option were to go ahead. The discharge resulted in slightly lower temperature in the experimental channel and also slightly higher DO concentrations, probably due to the ability of cooler water to dissolve more oxygen. In both cases these differences were not ecologically significant as the changes were minor (0.3°C, 0.6 g m^{-3} DO), less than normal daily fluctuations, and did not result in temperature or DO falling outside of acceptable ranges. Although oxygen levels can be very low in groundwater, re-oxygenation of the groundwater water appeared very rapid, so that by the time it was discharged into the river via the aeration tower and then by cascading over the large rip rap at the base of the outlet pipe, it was fully oxygenated. This finding suggests that no extra aeration devices would be needed should this current method still be used in the RRwGW option.

Water pH was about 0.7 pH units higher in the experimental channel than the control channel and reached a maximum of 8.2 excluding outliers. This pH is within the range normally recommended for stream water to protect fish health (6.5-9, USEPA 2009).

The conductivity in the experimental channel was considerably higher than the control channel, reflecting the higher concentrations of anions and cations in the groundwater. The potential effects of the increases in these parameters are discussed further below.

Under the Resource Management Act (1991), a number of specific water quality classes have been identified. A number of these classes may have relevance for the discharge of groundwater into the Waikanae River:

- Class AE Water (being managed for aquatic ecosystem purposes);
- Class F water (being water managed for fishery purposes);
- Class FS water (being water managed for fish spawning purposes);
- Class CR water (being water managed for contact recreation purposes);
- Class WS water (being water managed for water supply purposes)

Each of these classes has specific standards which apply after reasonable mixing with the receiving water. Examination of the standards for these five classes show the following requirements:

1. the natural temperature shall not be changed by more than 3^{o}C, and (for class F water) shall not exceed 25°C

2. the concentration of dissolved oxygen shall exceed 80% of saturation, OR, for Class WS, the concentration shall exceed 5 g per cubic metre (c. 50% saturation at 15^{o}C and at sea level pressure);
3. any pH change shall not have an adverse effect on aquatic life, OR, for class WS, the pH of surface waters shall be between 6.0 and 9.0;
4. there shall be no undesirable biological growths as a result of any discharge of contaminant;
5. the water shall not be tainted or contaminated so as to make it unpalatable for consumption by humans after treatment.

Monitoring of groundwater discharge during the study clearly showed that it would have met all these standards.

4.1.2 Major anions and cations

The groundwater discharge increases the sodium, magnesium, calcium and bicarbonate concentrations in the Waikanae River. Analysis of the water geochemistry using Piper diagrams indicates that the groundwater discharge changes the character of the river water chemistry, from something similar to other rivers around Wellington and the North Island, to something more like spring, or groundwater. This is not surprising. Our previous modelling (Suren et al. 2010) showed that water chemistry would change the most with the input from bore K4, which has water chemistry least like the Waikanae River; and would change the least with the input from bore Kb4, where water chemistry is closer to that of the river. This analysis suggests that the final choice of bore for the river recharge option may need to be made based on sustainable yield, as well as the different water chemistry. It may thus be possible to select a bore (or combination of bores) which would change water chemistry the least. However, this does not consider other factors, such as nutrient concentration in the bore waters or river water (and how these interact). A PCA analysis including nutrient concentrations confirms that water chemistry downstream of the groundwater discharge will be quite different to that upstream of it, and to that of other streams in the Wellington region.

Changes in water chemistry arising from the RRwGW are unlikely to have adverse effects on the biological communities. Measured water quality parameters that were predicted to significantly change (sodium, calcium, magnesium, bicarbonate) are of low toxicological significance, based on data from the Ecotox database (http://cfpub.epa.gov/ecotox; accessed June 2011). That is, only very high concentrations (typically > 1000 g m^{-3}) are expected to cause toxicity to freshwater organisms. These are much higher concentrations than can be expected as a result of the RRwGW option, which had concentrations ranging from seven times lower (for bicarbonate) to 170 times lower (for magnesium) these upper values.

4.1.3 Nutrients

The ground water has higher concentrations of DRP than the Waikanae River and discharges to the river were predicted to increase DRP downstream (Suren et al. 2010). In the current study, the DRP concentrations in the experimental channel (0.046 – 0.051 g m^{-3}) were substantially higher than in the control channel (0.008 – 0.009 g m^{-3}) and also higher than previously measured in the Waikanae River (Suren et al. 2010).

A region-wide analysis of river water quality (Perrie & Cockeram 2010) showed that the mean concentration of 12-monthly median DRP in 7 streams of type similar to Waikanae River (classified by REC as cool wet climate, low elevation source of flow, hard sedimentary geology and pastoral landuse) was 0.023 g m^{-3} with a maximum of 0.073 g m^{-3} measured in Mangatarere Stream at SH2. The concentration of 0.008 g m^{-3} measured in the control channel was thus well below the regional mean for rivers similar to the Waikanae River. In the experimental channel, the DRP increased to 0.051 g m^{-3}, still within the range observed for Wellington streams in the same REC class. Streams with the same climate, flow and geology characteristics but indigenous or forest land use typically had lower DRP concentrations. The effects of this increase in DRP on algal growth are discussed in Section 4.2.

4.1.4 Toxic contaminants

Table 4-1 shows the maximum concentrations detected in the ground water and downstream in the Waikanae River compared to trigger values to protect aquatic ecosystems. Monitoring data reported by Suren et al. (2010) showed that trace metal concentrations were very low in the ground water with most metals below detection limits of 0.00005 to 0.001 g m^{-3}, with the exception of manganese, measured at up to 0.19 g m^{-3} and total iron at up to 0.07 g m^{-3}. Subsequently downstream concentrations showed no increase in metals except manganese (Suren et al. 2010). The concentrations in the ground water discharges were all well below toxicant trigger values for protection of aquatic ecosystems based on protecting 95% of species (ANZECC 2000). For arsenic and chromium, the more stringent trigger values (to protect 99% of species) of 0.0008 and 0.0001 g m^{-3} were slightly lower than the detection limit used for these analyses (0.001 and 0.0005 g m^{-3} respectively). There is a chance that the arsenic and chromium may be found in the discharge at concentrations slightly above these trigger values for 99% protection, but still below the detection limits. However, as stated, the concentrations did not exceed the trigger values based on protection of 95% of species (0.013 and 0.001 g m^{-3} respectively) and once mixed with the Waikanae River water, the downstream concentrations would easily be below the more stringent trigger values.

Ammoniacal-N and nitrate-N can also act as toxicants at elevated concentrations (ANZECC 2000). Concentrations of nitrate-N were well below recommended trigger values in the ground water and downstream during the experimental trial and during all previous monitoring. Ammoniacal-N concentrations were not elevated above the trigger values in the ground water or river when measured during the experimental trial (0.20 g m^{-3} in the discharge; 0.09 g m^{-3} in the experimental channel). However, ammoniacal-N concentrations in water from bore K6 have previously exceeded the most stringent trigger value, measuring a maximum of 0.34 g m^{-3} (KCDC monitoring data). Under a scenario the 1 in 50 year low flow, a high demand from an estimated population in 2060, plus the RRwGW using only water from K6 (i.e., the river comprising 72% ground water from bore K6), predicted downstream river concentrations of ammoniacal-N would be 0.25 g m^{-3}, still below the lowest trigger value for protecting the most sensitive species (99% protection). The other bores had substantially lower ammoniacal-N concentrations, with a maximum of 0.08 g m^{-3} measured from Kb4, well below the 99% trigger level. Given this, there appears to be little chance of ammoniacal-N concentrations reaching levels which would endanger ecosystem health within Waikanae.

Table 4-1: Trigger values for toxicants for protection of aquatic ecosystems compared to river water quality.

Parameter	Maximum measured in discharge ($g\ m^{-3}$)	Maximum measured or predicted downstream ($g\ m^{-3}$)	ANZECC guideline ($g\ m^{-3}$) Protection of 99% of species	ANZECC guideline ($g\ m^{-3}$) Protection of 95% of species
			0.0008	0.013
			0.00016	0.0005
Dissolved Chromium	< 0.0005	< 0.0005	0.0001	0.001
Dissolved Copper [a]	< 0.0005	< 0.0005	0.0025	0.0035
Dissolved Lead [a]	< 0.00010	< 0.00010	0.004	0.014
Dissolved Nickel [a]	< 0.0005	< 0.0005	0.020	0.028
Dissolved Zinc [a]	0.002	< 0.0010	0.006	0.020
Dissolved Manganese	0.19	0.078	1.2	1.9
Ammoniacal-N	0.34	0.25	0.32	0.90
Nitrate-N	0.087	0.17	4.9	7.2

Note: [a] Hardness dependent trigger value, trigger value used is for waters of moderate hardness (60 $g\ m^{-3}$ as $CaCO_3$).

No trigger value is provided by ANZECC (2000) for iron, however a guideline of 1 $g\ m^{-3}$ is suggested by USEPA (1976) and the Ministry of the Environment, British Columbia (Phippen et al. 2008) for total iron, while for dissolved iron the lower limit of 0.35 g^{-3} is recommended by Phippen et al. (2008). The maximum total iron concentration measured in the discharge during the experimental trial was 0.25 $g\ m^{-3}$ and when measured by Suren et al. (2010) it was 0.07 $g\ m^{-3}$, with both values well below the guidelines of 1 $g\ m^{-3}$. Higher concentrations have been measured in the groundwater discharges in the past during KCDC monitoring (up to 2.5 $g\ m^{-3}$). Under the 2060 scenario when groundwater represents about 70% of the flow, if iron was present at 2.5 $g\ m^{-3}$ in the ground water, there would be insufficient dilution in the river to reduce the concentration to below a guideline of 1 $g\ m^{-3}$ (estimated downstream concentration of 1.8 $g\ m^{-3}$). Despite this, adverse effects from elevated iron are not expected, as these elevated concentrations have been intermittent and therefore would not be expected to result in chronic effects in the Waikanae River. Dissolved iron was not detected in the discharge or downstream during monitoring above a detection limit of 0.02g m^{-3}, which is well below the guideline for dissolved iron (Phippen et al. 2008). The maximum dissolved iron concentration measured in the bores was 0.09 $g\ m^{-3}$ in Kb4, which remains well below the guideline of 0.35g m^{-3} (Phippen et al. 2008). In summary, this data shows that there is very low likelihood of toxic effects on the Waikanae River from metals, ammoniacal-N or nitrate-N.

4.2 Algae

A key requirement of this assessment was to determine whether the predicted increased nutrient load associated with the groundwater discharge might enhance algal cover or biomass in the river during times when the groundwater discharge would occur. This concern is particularly pertinent given the fact that excessive proliferations of toxic cyanobacterial mats have been common in the Wellington region. For instance, during the spring of 2005, thick mats of benthic cyanobacteria became established in several of the region's rivers, including the Otaki, Waikanae, Hutt, Mangaroa, Wainuiomata and Waipoua rivers (Milne and Watts 2007). Following the deaths of five dogs in the Hutt River catchment, an inter-agency response team consisting of the Greater Wellington Region Council (GWRC), regional public health, and local councils were set up to monitor the rivers, and issue appropriate media releases and warning signs. These blooms were thought to have occurred mainly as a result of favourable environmental conditions experienced during the spring of 2005. Such favourable conditions included a mixture of below average river flows, fewer "freshes" in the rivers that could have scoured excess growth from the river, and warmer water temperatures (Milne and Watts 2007).

Although nutrient concentrations are often regarded as a key variable controlling periphyton growth (e.g., Biggs 2000), Milne and Watts suggested that the nutrient status of rivers in Wellington region was unlikely to be as important a causal factor for cyanobacterial mat proliferations as elevated water temperatures, lack of flushing flows, and sustained low flow conditions. This contention was based on the fact that median nutrient concentrations of the Otaki, Waikanae, Hutt and upper Wainuiomata rivers are characterised by low-nutrient conditions - reflecting the dominance of indigenous forest in their catchments – yet cyanobacterial mats still proliferated in these rivers. Given that low flow and high water temperatures are part of a normal seasonal cycle, they are generally outside the realms of management intervention.

Heath et al (2011) examined spatial and temporal variations of cyanobacteria in the Hutt (6 sites) and Wainuiomata (2 sites) rivers for 12 months from December 2007. They monitored cyanobacterial cover, and collected water samples for nutrient analysis fortnightly during periods of high, variable flow (ie., during winter - spring), and weekly during times of stable flow (summer and autumn). They also continuously monitored stream flow and water temperature. Their observed nutrient concentrations in the Hutt and Wainuiomata Rivers were similar to those we observed in the Waikanae River control channel. Their observed DRP concentrations were mostly lower than what we found in the experimental channel, which was based on a predicted RRwGW scenario of low flow and high demand. Despite the low nutrient levels, Heath et al observed a very high cover of cyanobacteria at some sites. For example, the percentage cyanobacteria mat cover was often > 50% at four of the six Hutt river sites, with cover occasionally peaking to 70%. Heath et al used stepwise logistic regression to establish the best variables (nutrients, and the 5-day average of temperature and river flow preceding each sampling interval) to predict cyanobacterial cover. Their analysis showed that river flow and temperature were the only significant predictor variables correlated with the presence and cover of cyanobacterial mats. The importance of low sustained flows, and an absence of flushing flows, combined with high temperatures has also been shown to correlate well with increased cover of periphyton (including cyanobacterial mats) throughout New Zealand (Biggs and Price 1987; Biggs and Close 1989).

The importance of low flows is emphasised by the Greater Wellington Regional Council's decision to use periods of low stable flow - defined as a period of two weeks where flow has not exceeded three times the long-term median - as an early warning indicator of the strong likelihood of cyanobacterial mat proliferation. Heath et al (2011) also showed that the probability of mats formation increases dramatically when the river flow was half the yearly average and water temperature was greater than 14℃. They suggest that water temperature should also be monitored by the regional council to help them predict when proliferations would occur. We found that discharge of groundwater into the experimental channel under a predicted RRwGW scenario of low flow and high demand had little effect on water temperature - in fact temperature was slightly reduced in the experimental channel. Given that the RRwGW option will have no effect on the magnitude or duration of a summer low flow in the lower river, and the fact that water temperatures are highly unlikely to increase from groundwater discharge, the results of Heath et al support our contention that the RRwGW option will have no effect on the propensity of cyanobacterial mats to develop.

Finally, this argument is supported by findings that some of cyanobacteria have an ability to fix nitrogen (Bergman et al 2006), and that low levels of DIN (Dissolved inorganic Nitrogen - i.e., ammonia + nitrate) may favour some cyanobacteria species. Indeed, Borchardt (1996) has suggested that it is nitrogen, rather than phosphorus, that limits cyanobacterial growth. If this is correct, then the RRwGW option is very unlikely to increase DIN levels in the river as these were actually lower in the experimental channel (0.15) than in the control channel (0.19). The ratio of total nitrogen to total phosphorus (TN:TP) is often used to describe the nutrient status of water. Thus, waters with a low TN:TP ratio are more likely to be nitrogen limiting, while waters with a high TN:TP ratio are more likely to be P limited. The control channel had a high TN:TP ratio (23) whereas the experimental channel had a low ratio (c. 3). Six of the 7 sites monitored by Heath et al had similarly high TN:TP ratios as the Waikanae River control channel, yet three of these sites had the highest cyanobacteria cover they observed. One of their sites had a very low TN:TP ratio (2.6), and similar to that observed in the experimental channel under the RRwGW scenario. However, cyanobacterial cover was only moderate at this site.

The GWRC has been monitoring cyanobacterial blooms at 18 sites in nine rivers in the Wellington region since 2009, using methodologies outlined by Wood et al. (2009). The monitoring program is designed to manage risks to recreational users, and has developed an interim alert level framework. This alert level framework is based on assessments of the percentage of a riverbed that a cyanobacterial mat covers at each site. Three alert levels have been proposed based in part on cyanobacteria cover:

1. surveillance or green mode - up to 20% cover;
2. alert, or amber mode - 20 to 50% cover;
3. action, or red alert - either greater than 50% cover, or up to 50% cover where mats are visibly detaching from the substrate and accumulating a long the river's edge.

During the 2009-2010 summer, seven of these 18 sites had no observable cyanobacterial mats, while seven sites (including the Waikanae) had matt cover that placed them in the surveillance mode. A further three sites had maximum weekly matt cover placing them in the alert mode, and one site had maximum cover placing in the action mode (GWRC unpublished data).

Similar cyanobacterial proliferations have also been observed in rivers in North Canterbury, such as the Ashley, the Hurinui, and the Selywn. Cyanobacterial cover in these rivers have been monitored using recommended Ministry of Health interim guidelines (Wood et al. 2009), and the cover of cyanobacterial mats assessed using an underwater viewer at five points along 4 evenly spaced transects. Average cyanobacterial cover at these sites range from 11 to 20%, whereas cover within a particular transect ranged from 1 to 80% (ECan unpublished data). Mean cover of cyanobacterial mats at two sites in the Ashley River in March 2010 was approximately 20%, placing it in the alert level (or amber mode) framework as recommended by the Ministry of Health (Wood et al. 2009). Examination of long-term (6 + years) nutrient levels from these rivers showed that dissolved reactive phosphorus and total nitrogen concentrations were generally less than those found in the experimental channel, further reinforcing that high cyanobacterial cover can form even in the absence of nutrient enrichment, and that this cover is caused more by the lack of flushing flows and warm temperatures than by nutrients.

Our chlorophyll data showed clear temporal patterns in biomass, which increased during the summer and then declined following a flood in early April. Although the magnitude of the initial increase was highest in the experimental channel, it was also observed at the lower sites, particularly at the State Highway site where effects of the groundwater discharge were assumed to be negligible given the degree to which any groundwater was diluted to (c. 7.5% than of the experimental channel). Moreover, high chlorophyll biomass was observed on large stable substrates in the main river outside the experimental channel, even though this area received no groundwater. The high cover of cyanobacterial mats in the experimental channel during the time that groundwater was discharged thus appears unrelated to this discharge, or to any increased nutrients associated with this. Instead, this may have simply reflected the normal summer and early autumn increase of cyanobacteria, which was found on large stable substrates throughout the river. Indeed, independent monitoring of cyanobacterial matt cover by GWRC during the same time showed a similar increase in cover where maximum cover at the State Highway site increased from 4% (surveillance mode) to 42% (alert mode) of the streambed in March. This increase in cyanobacterial mat cover suggests that potential nutrient enrichment associated with the proposed groundwater discharge would be unlikely to increase the probability of a proliferation of cyanobacterial mats in the river. Instead, these appear to be dependent mainly upon periods of long stable flow, the absence of flushing flows that remove excess biomass accumulations,and warm summer temperatures.

Suren et al. (2010) showed that there would be no difference in the low flow hydrological statistics of the residual flows downstream of the take and below the groundwater discharge point with the groundwater recharge option. Furthermore, the maximum take (370 L/s) is such a small proportion of the flow that there is no difference in the high flow statistics. Thus, the proposed groundwater recharge option is highly unlikely to increase the period of stable flow, or reduce the chance or magnitude of unpredictable flushing flows.

4.2.1 Algae: conclusions

Algal biomass in the Waikanae River during summer and autumn appears to be controlled mainly by the persistence of periods of sustained low flow and warm temperatures. The RRwGW option will not affect these variables, so algal blooms (dominated by cyanobacteria) will still occur in the Waikanae River as they do presently. It appears highly unlikely that the groundwater discharge during low flows will increase the frequency or magnitude of these blooms. Under the RMA, discharges shall not result in any undesirable biological growths. It is evident that the proposed groundwater discharge will meet this requirement.

4.3 Invertebrates

The invertebrate fauna of the Waikanae River was dominated by EPT taxa, and indicative of a river in good ecological condition. This most likely reflects the relatively low-degree of catchment modification, and high degree of native bush throughout upper parts of the catchment (Milne and Watts 2007).

Analysis of invertebrate data showed at most, only a few small changes to some quantitative biotic metrics (QMCI and % EPT) in the experimental channel over time. These most likely reflected small changes in densities of the mayfly *Deleatidium* and the caddisfly *Olinga* between the channels. These animals both increased in abundance in the control channel over time, whereas their densities remained constant in the experimental channel. Despite these differences, examination of the overall community composition showed that the communities in both channels changed by roughly the same amount. This suggests that the RRwGW option will have negligible effects on the invertebrate community in the Waikanae River.

The relationships observed by Death and Joy (2004) between invertebrate communities and alkalinity or conductivity in streams throughout the Manawatu-Wanganui region were unlikely to reflect a direct physiological response. Although invertebrates such as molluscs can respond directly to chemical characteristics of water such as $CaCO_3$ for shell formation, Death and Joy stated that it was "hard to see why animals such as *Deleatidium* should be so tightly linked with alkalinity". They suggested that measures of conductivity and alkalinity may have acted as surrogate measures for algal productivity, geology or Landuse, which are known to regulate invertebrate communities. Short-term increases in alkalinity or conductivity associated with groundwater discharge are thus highly unlikely to have a direct effect on invertebrates.

Based on this, it is likely that the small changes observed in the invertebrate community in the experimental channel were not caused by groundwater. This contention is further supported by examination of the invertebrate communities in the lower sites (Jim Cooke Park, and State Highway 1). These communities also changed over time, but were only subject to a very diluted discharge of groundwater (c. 7.5%). Our findings showed that the invertebrate communities changed over time at all sites, but those communities in the control site and the State Highway site changed the most. In contrast, the invertebrate communities in the experimental channel appeared to have changed the least, despite being subject to the discharge of groundwater.

Examination of relationships between invertebrate communities and periphyton biomass showed clear and consistent relationships. Thus, many of the changes observed to the invertebrate communities appeared be related to the natural growth of algal communities. Our analysis and interpretation of the data suggests that invertebrate communities were not affected by the groundwater discharge, but instead by changes to the algal communities observed during the study. These changes reflected natural successional patterns associated with the warm water temperatures and mostly stable flows encountered during the study. This observation confirms a theoretical model developed to predict changes to invertebrates during low flows (Suren and Riis 2010). Here, natural successional changes in filamentous green algae or cyanobacterial mats will lead to potentially large alterations to instream habitat conditions during low flows, especially when these plants cover extensive parts of the stream bed. Under such conditions, the substrate surface will change from cobbles and gravels to thick algae. Although hydraulic conditions and substrate size might not have changed, overall habitat suitability for invertebrates such as mayflies or caddisflies would change as these thick algal communities develop (Hart 1985, Suren 2005). These changes in habitat condition may help explain the observations of reduced percentage EPT and QMCI in the experimental channel when compared to these metrics in the control channel where mat development was not as great.

4.3.1 Invertebrate communities: conclusions

The invertebrate fauna of the Waikanae River is typical of a river in good ecological condition, most likely reflecting the relatively low-degree of catchment modification throughout upper parts of the catchment. Groundwater discharge caused some subtle changes to the abundance of some taxa, but the overall community composition in the experimental channel remained the same. Invertebrate communities also changed at the lower sites, and the greatest degree of change was found in the control site and the State Highway site. This suggests that the RRwGW option will have negligible effects on the invertebrate community in the Waikanae River. Instead, changes to the invertebrate communities appear driven more by natural algal growth associated with warm water temperature and stable flows.

4.4 Fish

The effects of groundwater discharge on fish were assessed by three independent, but complementary methods: 1) quantifying fish densities in the two channels before and after the experiment; 2) measuring fish survival and growth in cages placed in each channel; and 3) monitoring survival of fish exposed to different treatments in aquaria. The first method allows us to determine whether fish would preferentially move from a channel subject to groundwater discharge. Our results showed no evidence of this: indeed total fish densities increased in the experimental channel over time, although there was a slight reduction in density of longfin eels. Some of this increase reflected enhanced densities of small redfin bullies, suggesting that groundwater was not interfering with the upstream migration of small individuals. Use of the fish cages in the second method allowed us to determine what the short to medium term effects of a mixture of 70% groundwater: 30% river water was on fish survival and growth, given their inability to move from the cages. No adverse effects were detected on any of the three fish species even after 32 days. The last method, using the fish tanks, more closely mimicked the second method in that it exposed fish to particular water chemistry, without their ability to swim away. This experiment had the added advantage of exposing fish to 100% groundwater, something which would never happen under a groundwater recharge option. We adjusted the header tank baffles to achieve a 70% groundwater mix, but the long-term average in the mixed tanks was 85%. Unfortunately, most of the redfin bullies died in all treatments during the experiment, despite constant attention to feeding and maintenance of clean water. This mortality most likely reflected their reluctance to feed on the variety of food items offered. Furthermore, most of longfin eels escaped, despite our best efforts to contain them. However, analysis of the inanga data showed no significant difference to their survival or growth in any of the three water treatments – even the 100% groundwater.

4.4.1 Effects on Fish: Conclusions

Results from three independent lines of enquiry generally suggest that groundwater has no adverse effect on fish survival or growth. Results of the electric fishing furthermore suggested that fish did not swim away from areas where groundwater was being discharged

4.5 Effects on wetland invertebrates

The invertebrate fauna found in the three wetlands appears typical to that of other wetlands in the North Island. All the animals encountered appear common and widespread throughout New Zealand, and no animals of particular conservation interest or concern were identified. Knowledge of invertebrate communities in wetlands is at its infancy (Suren and Sorell 2010), so little can be said on the potential effects of reduced water level in the wetlands that may occur as a result of the groundwater recharge option. However, this option will only occur during times of low rainfall, when the wetlands would naturally be drying out. All the open water bodies in each of the three wetlands were large and relatively deep, so it is felt that these would only dry in exceptional circumstances. Whether the abstraction of groundwater will hasten this drying process is not yet fully understood, although Michaelsen (2010) concluded that the Waimea aquifer is a relatively high yielding aquifer that can withstand groundwater abstraction of 32,000 m^3/day for 90 days over the summer period, and that the long term effects on the aquifer system of such pumping are less than minor. Moreover, they concluded that the drawdown effect on the shallow aquifer was less than natural seasonal variation. Given these findings, if an open water body in a wetland is likely to dry in a severe drought, it is likely that it would do so even in the absence of the groundwater recharge option. Many of the animals encountered were highly mobile and capable of rapidly recolonising wetlands once they water returned, so it is highly likely that the proposed activity would have less than minor effects on these communities.

A recent study documented the differences in the fauna of permanent and temporary wetlands in the South Island (Wissinger and McIntosh 2009). Although the fauna of permanent and temporary wetlands differed significantly, there was little evidence of organisms especially adapted to living in temporary environments: i.e., species in temporary wetlands were merely a subset of those in permanent wetlands. Chironomid midge larvae, water bugs, beetles, and crustacea dominated the fauna of temporary wetlands, whereas the fauna of permanent wetlands was dominated by molluscs, worms, caddisflys, dragonflies and damselflies. The fauna of the three wetlands in this study contained a mixture of these organisms, meaning many would be highly mobile, and able to recolonise areas that were to dry. Therefore, the effect of the groundwater option is thought to be insignificant on wetland invertebrates in the long term.

4.4.2 Effects on wetland invertebrates: conclusions

The invertebrate fauna found three wetlands near the groundwater recharge area is typical to that of other North Island wetlands, and all the animals encountered are common and widespread. Any potential drawdown effects of the RRwGW option on wetland water levels is less than natural seasonal variation. The effect of the RRwGW option is regarded as insignificant on wetland invertebrates in the long term.

5 The RRwGW option in a legislative context

The results gleaned from our investigations can be used to assess the implications of the RRwGW as it applies to the Waikanae River in the Regional Policy Statement (RPS) and Regional Plan. Tables 15 and 16 in Appendix 1 of the Proposed Wellington Regional Policy Statement 2009 identifies the Waikanae River as having significant recreation and amenity values (for fishing, swimming and camping) and indigenous ecosystem values . Policy 17 of the Proposed RPS directs Regional Plans to protect these values whilst Policy 42 requires these values to be considered by councils when considering resource consent applications which will potentially effect these significant values.

Consistent with Policy 17 of the RPS, the Wellington Regional Freshwater Plan 1999 confirms the Waikanae River as a water body with:

- Nationally threatened indigenous fish recorded in the catchment (Appendix 3);
- Important trout habitat (including spawning areas) - Water quality to be managed for fishery and fish spawning purposes (Appendix 4); and,
- Water quality to be managed for water supply purposes (above the intake at the water treatment plant) (Appendix 6).

To balance the demand for consumptive water from the Waikanae River and the significant recreation, amenity and ecological values afforded to it, the Regional Freshwater Plan imposes a minimum flow of 750l L/s (at the water treatment plant), below which point abstraction of water should cease.

The effects of the RRwGW option on the above identified ecological values are as follows.

- The ***minimum flow*** of 750 L/s which has been imposed to protect the river values will not be affected by RRwGW, indeed this has been introduced to ensure that the minimum flow is maintained whilst ensuring adequate water supply.
- The RRwGW option is highly unlikely to have any effect on ***nationally threatened indigenous fish*** such as Galaxiads or eels that are found throughout the catchment. Galaxiads such as giant kokopu (*Galaxias argenteus*) and koaro (*Galaxias brevipinnis*) migrate upstream (as whitebait) in spring (September – October), and adults spawn in the autumn and winter, where larvae then head go to sea. These life history movements will not be affected by any changes to the chemical signature of the Waikanae River as a result of the RRwGW, as this will occur outside the migration periods of these fish.
- The fact that the Waikanae has been identified as ***important trout habitat*** will not be affected by the RRwGW as this will not affect habitat conditions in the river as it is unlikely to affect substrate characteristics.
- The RRwGW option is unlikely to affect ***fish spawning***. For example, brown trout spawn in May and June (McDowall 1990), which is Outside the operating time for RRwGW. Inanga spawn in Autumn and early winter (March, April, May)

in vegetation inundated by high tide, and in the lower reaches just near the salt wedge. RRwGW will operate when necessary mainly in summer - early autumn, and is unlikely to go beyond April. As such, it is not expected to affect inanga spawning behaviour. Redfin bullies spawn in spring, and the larvae then go to sea where they return after several months into the rivers. We detected many small redfin bullies in the experimental channel in April, strongly suggesting that any groundwater discharge is associated with RRwGW will not affect these fish.

6 Comparison of RRwGW to Maungakotukutuku Dam

Although the RRwGW is the preferred option, a dam on the lower Maungakotukutuku river is regarded as the second preferred option. Indeed Council is considering its water supply approach over the very long term (over 100 years), and if RRwGW is confirmed as feasible, the intention is to protect the dam site to be considered as part of the water supply in the long term future.

While this report focuses on the effects of the RRwGW scheme, this is not to say that the dam option would not have its own range of adverse environmental effects. Adverse effects arising from a dam could occur during the construction phase, and following dam completion and filling, and a number of issues would need to be address (ie. avoided, remedied or mitigated). The largest construction effect is likely to be caused by sedimentation, although this can be minimised using standard, best practice construction techniques. Moreover, construction related effects are usually only short term.

The effects of dams following construction on streams are more severe and long-lasting. An important aspect to consider is what to do about the vegetation which would be flooded above the dam. Any drowned vegetation will inevitably decompose, with potential effects on water quality in terms of oxygenation, and release of methane and hydrogen sulphide. If vegetation is removed prior to dam filling, then steps are needed to ensure that excess sediment runoff does not occur from bare areas, particularly given the steep nature of the gorge that this river flows along.

The nature of the dam outlet needs to be considered, in terms of a surface or deep outlet. If a surface outlet structure is proposed, then the lake could undergo stratification, particularly within the deep gorge area. Although a deep-water off-take may minimise the likelihood of stratification, it could potentially affect water chemistry below the outfall. Given the large amount of vegetation in this catchment, and around the river, there is a risk that the water could go anoxic as vegetation decays. Release of this anoxic water below the dam may have significant detrimental effects to stream organisms, as initially occurred with the Opuha dam in South Canterbury.

Dams replace shallow, running-water environments with deep, slow-flowing (i.e., lentic) environments above them. Most of the invertebrates currently present in the Maungakotukutuku stream (for example filter feeding mayflies, stoneflies and caddisflies) will disappear as they can not tolerate conditions typical of deep, standing water. The current river community will be replaced by a new community typical of standing water. This fauna is likely to be dominated by snails, midges, and zooplankton. Fish such as redfin bullies, torrent fish and koaro (all of which are found in the Maungakotukutuku river) require relatively fast flowing and shallow water, so habitat for these species will be lost. However, trout, giant kokopu and eels (which are also found in the river) can tolerate lentic conditions.

Dams also disrupt the upstream and downstream movement of organisms in rivers, which is particularly relevant for many of New Zealand's migratory fish. For instance, longfin eels, giant kokopu and koaro need access to the sea for the completion of their life cycles. Dams thus need some form of fish passage to allow for both upstream and downstream movement. It is also important to have a minimum residual flow below a dam to ensure that excessive algal blooms, instream plant growth, or organic matter does not accumulate, and that sufficient beneficial habitat is left for invertebrates and fish dwelling below the dam.

While many of these adverse effects can be addressed through vaious means, it is important to note that they have not been studied in detail as part of this study. The effects are noted here simply to demonstrate that in the event that the dam is identified as a preferred option at this time, there will be further ecological studies required to document the adverse effects and identify mitigation measures required. In the event that the dam is to be seen as a long term option, the effects of construction will need to be considered at some point in the future. By this time, there will be a history of the impacts of RRwGW in place, and it will be possible to better understand the impacts of both schemes operating in tandem. Given that this is likely to be 50 or 100 years in the future, it is not considered possible to do this scientific assessment at this time.

7 Replies to external peer review comments

As part of the investigation process behind the RRwGW, CH2M Beca and KCDC arranged for an independent review of this report, undertaken by Associate Prof Russell Death at Massey University (see Appendix 1). This report was received by NIWA on 19 July, and a videoconference to discuss the comments raised was held one month later (18 August 2011). In attendance were:

- Alastair Suren, NIWA;
- Andrew Watson and Greg Pollock, CH2M Beca;
- Phillip Stroud, Kapiti Coast District Council;
- Associate Prof Russell Death, Massey University.

The review by Dr. Death concluded by saying:

The only conclusion from Suren et al. (2011) that I have concerns with is that increased nutrient concentrations from the recharge will not affect periphyton biomass and as a result the rest of the riverine ecological community. Their BACI design shows a clear effect of the recharge on periphyton growth but they dismiss this as a result of practical difficulties of replication (a BACI designed experiment should eliminate this). Similarly the nutrient diffusing substrates they deployed did not produce results to equivocally rule out nutrient limitation on periphyton in the Waikanae River.

The argument they make for their experimental observations seems plausible. However, it also seems equally plausible to me that the increased nutrient levels from the groundwater recharge are leading to increased periphyton biomass (which was observed) which in turn can alter riverine ecological communities. It is a shame that after so much research one is still left with considerable uncertainty around the potential effects of the recharge.

The research has not convinced me that the recharge will not affect periphyton growth in the river and consequently changes in the other biological components of the river. However, similarly there seems a reasonable level of uncertainty around the observed increases in periphyton being a result of the recharge. As a pragmatic way forward I would therefore conclude that there is probably no ecological reason for the recharge not to be used as a potential solution for the Waikanae River water abstraction problems. However, I would suggest that periphyton biomass and cover, and invertebrate communities above and below the recharge location are monitored at least once a year.

The following responses are made to address the concerns raised by Dr. Death.

1. The major uncertainty around the results from this study concern any potential effects on algal communities from the high nutrients associated with the RRwGW. Dr. Death commented that, from a theoretical point of view, the discharge will increase nutrients in the river, increasing algal cover. In this revised report, we present clear evidence to support our argument that this is unlikely to happen under RRwGW because:

    - most of the algal cover reflects cyanobacterial mat growth, and there is evidence that riverine cyanobacteria are not necessarily nutrient limited;

- we observed similar or higher cyanobacterial biomass immediately outside the experimental channel that was not exposed to groundwater during the study as part of natural summer processes of low flow and warm temperatures;
- cyanobacterial cover at the State Highway 1 site increased in a similar manner as that in the experimental channel, and was also very high;
- monitoring results of GWRC and ECan has shown high cyanobacteria cover in rivers with low nutrient status, suggesting that cyanobacteria growth is not always linked to nutrient enrichment;
- recent results from the Heath et al. (2011) paper highlight the fact that cyanobacterial growth is related more to periods of low and stable flow, than to nutrients.

2. Furthermore, it must be stressed that the ecological assessment presented here was based on a scenario of a 1 in 50 year low flow and projected water demand of 32,000 m^3/day in the year 2060. Under such a scenario, approximately 70% of the river flow below the treatment plant would be groundwater, and this would last for a 2 month period during summer. However, the reality is that in some years there may be no need to implement RRwGW, whilst in other years groundwater may be discharged at a much lower volume and for only short periods of time. For example, during the 59 days of this investigation, 8 floods were observed, with the highest peaking at 44.1 m^3/second on 5 April. If RRwGW was operating during this time, it would have stopped during all occasions when the flow above the Water Treatment Plant was greater than the current demand, minus the residual minimum flow. Given the fact that we observed such little ecological change even at 70% groundwater for 59 days, it seems highly unlikely that the implementation of RRwGW at a lower intensity and duration will have any adverse effects. Furthermore, KCDC is planning to implement water metering to reduce consumptive water use in the region. The successful implementation and uptake of such a strategy will go further to reduce the demand and therefore reduce the likelihood of the above scenario occurring.

3. Finally, we acknowledge the difficulty of any research, either field or laboratory-based in being able to provide definitive answers to ecological questions. There will always be a recurrent tension between scientists and non-scientists in that the latter are likely to want absolute certainty on an issue (e.g., that RRwGW will have no effect on the ecological values of the Waikanae River) whereas our results can, at best, provide only experimentally-based answers at a certain degree of statistical precision (e.g., the ubiquitous alpha = 0.05 test) to answer this question. Based on a rigourous examination of the effects of groundwater on three independent trophic levels (algae, invertebrates, and fish) and based on observations of other temporal patterns in the river unaffected by the experimental discharge of groundwater, we maintain that the ecological effects of our RRwGW are minor. Nevertheless, discussions during the videoconference conceded the importance of minimising any risk to the environment that may arise because of RRwGW, and agreed that a number of adaptive management responses may be required to minimise potential adverse effects.

The first response would be to develop a hierarchy of bore preference to use during times when RRwGW is needed, to discharge water from the bores with the lowest phosphorus concentrations first.

The second response would be to implement a monitoring regime in the river. This should be done following the standard MfE 2009 protocols for monitoring cyanobacteria, and be established to determine whether the river recharge is indeed causing any cyanobacterial growth over and above that occurring naturally in the river. Although it is ultimately up to the consent authority (in this case GWRC) to set out an appropriate monitoring strategy, the following recommendations are made:

- monitoring be done using consistent methods as highlighted in section 4.4 of the MfE 2009 cyanobacterial monitoring guidelines;
- select at least two sites above the RRwGW, and at least two sites below, but above the State Highway. A further two sites could be selected below the State Highway, and at Jim Cooke Park as per this study;
- sites should be selected to have as similar substrate composition, water velocity and shade as possible;
- monitoring be done by as few people as possible to minimise inter-operator variability;
- commence monitoring as soon as possible even before RRwGW commences, in order to better document the natural summer succession of cyanobacterial growth in the absence of RRwGW;
- monitoring to last until the first significant flood occurs in the autumn that removes excess cyanobacterial material.

Note that some of this monitoring can be incorporated into existing monitoring schemes currently undertaken by the GWRC as part of their SOE work.

The third response would be to undertake a range of active management techniques such as removal of cyanobacterial blooms from rocks downstream of the discharge should monitoring detect an increase in cyanobacteria cover in any discharge that may occur under RRwGW.

8 Overall conclusions

This study examined the responses of three discrete biological components at three different trophic levels in Waikanae River to the discharge of groundwater. Although algal biomass increased in the experimental channel during the study, this increase was observed in other locations in the river, and could not be attributed to the discharge of groundwater. Instead, these changes appear to be part of a natural succession of algal communities in the river occurring during periods of summer low flow, and warm temperatures. Algal communities in the Waikanae River are known to become dominated by cyanobacterial mats, and the Regional Council is currently monitoring these every summer. Groundwater discharge into the river appears very unlikely to further exacerbate the proliferations of cyanobacterial mats that already occur.

No consistent changes were observed to invertebrate communities in Waikanae River that could be attributed to groundwater discharge. Instead, strong relationships were evident between periphyton biomass (expressed as chlorophyll biomass) and invertebrate communities, and it is this natural periphyton succession during summer low flows that controls and regulates invertebrate community composition. Changes to water chemistry will not result in any chemical differences which could result in loss of invertebrate taxa. Moreover, groundwater discharge had no adverse impacts on fish. As such, the discharge of groundwater into the Waikanae River is very unlikely to have any adverse effects on invertebrate communities. Overall, the results of the studies conducted appear to show that no consistent ecological effect could be detected as a result of discharge of groundwater into the Waikanae River.

9 Acknowledgements

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Appendix A Review comments

The following comments were received from Associate Professor Russell Death. Overall comments to these are included in Section 8 of this report. Specific comments to some of the smaller points he raised are dealt with here, *in italics.*

Review of “The effects of groundwater discharge on the algal, invertebrate and fish communities of the Waikanae River” by Suren, Wech and Gadd, NIWA, contracted to Beca.

By Associate Professor Russell Death, Massey University

This is an extremely comprehensive study to investigate any potential adverse effects of a proposed groundwater recharge of the Waikanae River by Kapiti Coast District Council to supplement the district’s water supply. This involved examination, during a trial groundwater recharge, of invertebrate, periphyton and fish communities in a BACI experimental design. As Suren et al (2011) point out this is the most rigorous mechanism for testing ecological impacts. However, the design used did not end up yielding the expected level of rigour. That is, the different outcomes for “After” at Control and Impact sites were attributed to “Before” conditions in the habitat at the Control and Impact sites (not evident in the actual ecological measures in the “Before” treatment). I fully understand the practical difficulties of selecting suitable comparison sites in the real world, but it does really leave one with considerable uncertainty about the potential impacts of the recharge. They used nutrient diffusing substrates to investigate nutrient limitation of periphyton growth in the experimental channels. They also examined invertebrate and periphyton communities at a number of sites in the river downstream of the recharge. They evaluated instream and aquaria based potential effects of the altered chemical composition of the water on growth (over an extremely short interval) and survival of Longfin eels, Redfin bullies and Inanga. Finally they investigated the invertebrate fauna of three nearby wetlands and short term changes in water depth during the recharge operation.

Specific comments

I agree the differences in pH, temperature and Dissolved oxygen between the Control and Experimental channels, although significant, are not ecologically an issue.

Clearly however, the change in median conductivity from 100 µS/cm in the control to 539 µS/cm in the experimental channel is ecologically significant.

Although there is a large change in water chemistry in terms of conductivity, we failed to detect any major changes to either algal, invertebrate or fish communities.

Results

There is clearly a demonstrated increase in algal biomass in the experimental channel, consistent with the observed increase in conductivity and nutrient levels. However, Suren et al (2011) conclude this is unrelated to the increase in nutrients, but rather is a result of lowering summer flows and larger substrates in the experimental channel (substrates not present in the control channel). As supporting evidence they present one occasion when algal biomass in the experimental channel and area alongside were compared (with no difference) and where conductivity levels (measured but not presented) in the area alongside remained low.

In the revised report, I have presented this material. The fact that we only had one observation of the high algal biomass outside the experimental channel does not lower the importance of this finding – this biomass was seen to increase over the summer period reflecting the low stable flows. The decision to collect only the one sample to quantify this biomass merely reflects natural cost constraints that all AEE work is be conducted under.

It is a shame the nutrient diffusing substrates were adversely affected by flood effects as this would have provided more convincing evidence nutrient enrichment from the groundwater recharge will not have an effect on algal abundance in the river. The t-test results comparing control and experimental channel unenriched agar is reassuring, but where are the actual t-test and P values?

I agree that the loss of some of the nutrient diffusing substrates meant that we could not definitively reach a conclusion regarding nutrient limitation. However, the loss also highlights the unpredictable flood-prone nature of the river, which highlights the fact that the RRwGW option is unlikely to run continuously for 60 days. Again, this revised report includes the missing values.

Invertebrate communities

The assessment of the experimental effects on the invertebrates suggest some of the taxa could potentially be affected by the groundwater inflow. However, the largest differences appear to be between the experimental and control channels (which suggests the experimental setup was not ideal to assess any potential affects) and with time. The interaction term that could indicate an effect from the groundwater inflow was only significant in four of the examined cases. It would have been useful to get an idea of what the densities of the various taxa were, for example in figures, tables or even an appendix with a summary of the actual collected data. The explanation put forward by Suren et al. (2011) that the observed changes in invertebrate communities are a result of increases in the amount of algae seems a plausible one.

I have included figures in this revised report showing densities of the common invertebrates, as well the calculated metrics.

Fish

Unless there was some direct toxic effect of the recharge it seems unlikely that any effect on fish growth or survival over such a short interval of study would be discovered. Also not surprising most of the eels escaped.

The reference to the short interval of the study is based more in relation to the length of the study (up to 59 days) versus the total life of a fish (years). Again, this highlights the short period of time that the RRwGW will run for in comparison to normal ecological processes.

Wetlands

There were no invertebrates of conservation interest found in the three wetlands, but what about some of the fish, at Nga Manu for example? It also seems very unlikely that any change in wetland water level would be observed over such a short interval.

We considered invertebrates to be the best indicators to monitor. Agree with the comment that it is very unlikely to see changes in wetland water level over such a short interval, but againm this is the length of time that RRwGW would occur for in a low-flow, high demand scenario of 2060.

Conclusion

I applaud Suren et al. (2011) for such a comprehensive study and for Kapiti Coast District Council for commissioning the research. The research has been well thought out and appropriately analysed and I concur with most of the conclusions drawn from the findings by Suren et al. The critical conclusions from the study, with which I concur, appear to be:

1. Water physicochemistry of the Waikanae River will be altered during the recharge interval. However, of the changes, increased nutrient concentrations appear to be the only avenue for potential adverse effects on the river's ecology.
   This issue has been covered in more detail in this revised report.

2. Fish are unlikely to be directly affected by the change in water chemistry. Although I think it will be important not to have the groundwater recharge during whitebait runs. Baker & Hicks (2003) have shown chemical signatures from upstream can potentially affect upstream whitebait migration. This was not examined by Suren et al. (2011).
   The RRwGW will not occur during the white bait runs, which occur from September to November. RRwGW is unlikely to commence until February, or March, and run until April.

3. The invertebrate communities of nearby wetlands are unlikely to be affected by the recharge. Although I do not believe the presented data indicates water levels and/or native fish populations in those wetlands will be unaffected. However, similarly there is no evidence they will be affected.

4. The only potential adverse effect on the riverine invertebrate communities will be through potential changes in the periphyton communities.
   I agree with this latter statement, but contend that changes to the algal community (or cyanobacterial community) are driven more by periods of low, stable flow and warm temperatures, than by the increased nutrients. Causes of cyanobacterial proliferations are described more in this revised report.

The only conclusion from Suren et al. (2011) that I have concerns with is that increased nutrient concentrations from the recharge will not affect periphyton biomass and as a result the rest of the riverine ecological community. Their BACI design shows a clear effect of the recharge on periphyton growth but they dismiss this as a result of practical difficulties of replication (a BACI designed experiment should eliminate this). Similarly the nutrient diffusing substrates they deployed did not produce results to equivocally rule out nutrient limitation on periphyton in the Waikanae River.

The argument they make for their experimental observations seems plausible. However, it also seems equally plausible to me that the increased nutrient levels from the groundwater recharge are leading to increased periphyton biomass (which was observed) which in turn can alter riverine ecological communities. It is a shame that after so much research one is still left with considerable uncertainty around the potential effects of the recharge.

As discussed in section 7, there is always an element of uncertainty in science, and in ecological assessments. Key issues to consider are quantifying the degree of uncertainty, the degree of risk associated with the uncertainty, and the potential environmental effects of other alternatives. I acknowledge that the inherent site conditions meant that our choice of channels for the experiment did leave a degree of uncertainty in the results, particularly for the effect of groundwater on algal growth. However, I suggest that this uncertainty is greatly lessened by other observations of high algal cover and biomass outside the experimental channel, as well as high algal cover and biomass at the lower sites not exposed to the groundwater discharge. Moreover, our contention that high algal biomass (in terms of cyanobacterial mats) reflects more below stable summer flows, and warm temperatures, is supported by monitoring work in other regions, and by the paper by Heath et al (2011).

The research has not convinced me that the recharge will not affect periphyton growth in the river and consequently changes in the other biological components of the river. However, similarly there seems a reasonable level of uncertainty around the observed increases in periphyton being a result of the recharge. As a pragmatic way forward I would therefore conclude that there is probably no ecological reason for the recharge not to be used as a potential solution for the Waikanae River water abstraction problems. However, I would suggest that periphyton biomass and cover, and invertebrate communities above and below the recharge location are monitored at least once a year.

I agree with the need to monitor, and have made a number of recommendations that should feed into a proposed monitoring programme.

References

Baker, C.F. and Hicks, B.J. (2003) Attraction of migratory inanga (Galaxias maculatus) and koaro (Galaxias brevipinnis) juveniles to adult galaxiid odours. New Zealand Journal of Marine and Freshwater Research 37: 291-299.