BEFORE A SPECIAL TRIBUNAL UNDER SECTION 203 RESOURCE MANAGEMENT ACT 1991

UNDER	the Resource Management Act 1991
IN THE MATTER	of an application under Part 9 of the Act
AND	
IN THE MATTER	of an application for a water conservation order at Te Waikoropupu Springs and associated water bodies
BY	**NGĀTI TAMA KI TE WAIPOUNAMU TRUST AND ANDREW YUILL** **Applicant**
AND	**TASMAN DISTRICT COUNCIL** **Submitter**

BRIEF OF EVIDENCE OF GRAHAM DAVID FENWICK, 9 MARCH 2018

Introduction

1. My name is Graham David Fenwick.

2. I hold the following tertiary qualifications:

    - B.Sc. Zoology (Canterbury), 1972.
    - M.Sc. Zoology (Canterbury), 1975.
    - Ph.D. Marine Biology (Canterbury), 1984.
    - Dip. Business Administration (Canterbury), 1993.

3. I am employed by the National Institute of Water and Atmospheric Science (NIWA).

4. I am a biologist with over 40 years' experience as a practicing researcher, including eight years in business research. Since 1974, I have worked as a biodiversity scientist involved in environmental investigations for Memorial University of Newfoundland (Canada), the Australian Museum (Sydney), and the University of Canterbury. I have worked for NIWA as a scientist for 19 years (since 1998). In early December 2017, I resigned from my position of Assistant Regional Manager, Christchurch, to focus on science. My specialist areas are aquatic invertebrate biodiversity and the ecology of aquatic sediments.

5. My curriculum vitae is attached (Annex A) to show the breadth of my experience and expertise.

6. I have been asked to present aquatic biodiversity and groundwater ecosystem evidence jointly by the Tasman District Council and the applicants in relation to the Water Conservation Order Application by Ngāti Tama ki Te Waipounamu Trust and Andrew Yuill.

7. I have read the Code of Conduct for Expert Witnesses in the Environment Court's 2014 Practice Note and agree to comply with it. I confirm that the opinions I have expressed represent my true and complete professional opinions. The matters addressed by my evidence are within my field of professional expertise. I have not omitted to consider material facts known to me that might alter or detract from the opinions expressed.

My previous involvement with Te Waikoropupu Springs and associated waters

8. I note that, previously, I was engaged separately by three parties to provide biodiversity expertise on Te Waikoropupu (see 9-14 below). In undertaking all of these assignments, I used data and a substantial breadth of science information, all of which were publicly available. Because there is so little empirical research information on groundwater ecosystems, I extrapolated potential cause-effect and functional relationships from aquatic ecology to describe likely groundwater ecosystem responses to environmental changes. In doing this, I drew heavily on a multi-authored, in-depth literature review of groundwater ecology, groundwater quality and other values that commenced in 2013, and will be delivered to New Zealand regional authorities in mid-2018, once peer-reviewed (Fenwick et al. in prep.).

9. In November 2013, Mr Andrew Yuill, one of the applicants, engaged NIWA (me) to review aspects of a draft application for a Water Conservation Order for Te Waikoropupu from a groundwater ecosystem and biodiversity perspective. These aspects were:

- statements about the groundwater ecology and its effects on water quality and clarity,
- the biodiversity associated with the springs,
- potential threats to the groundwater ecosystem, and
- three principles for sustainable groundwater ecosystem management for potential inclusion in the proposed WCO application.

10. The resulting letter report (Fenwick 2013; dated 3 Dec 2013), peer-reviewed by Dr Clive Howard-Williams (Chief Scientist-Freshwater, NIWA), covered these points, including the three principles for sustainably managing groundwater ecosystems: aquifers are living ecosystems, water quality and availability determine the sustainability of groundwater ecosystems and their ecosystem services, groundwater and connected surface waters should be managed as a single resource.

11. In March 2015, Mr Yuill requested a further report, based on a desk-top evaluation, of likely water quality limits within Te Waikoropupu's aquifers for key variables that would sustain the spring's diverse values. In response, I prepared

a report entitled “Sustainability of Te Waikoropupu Springs’ aquifer ecosystems” (dated March 2015; peer-reviewed by Dr Scott Larned (Freshwater Research Manager, NIWA) and Dr Clive Howard-Williams (Chief Scientist Freshwater & Estuaries); updated July 2016). This report (Fenwick 2015) discussed potential limits for organic carbon, dissolved oxygen, nitrate and ammonia for sustaining biodiversity and ecosystem functioning at or near its present state. The report was attached to the WCO application as Appendix 9.

12. Dr Roger Young (Cawthron Institute) kindly pointed out that 1976 spring water concentrations of nitrate quoted within this report were one tenth of the correct values at that time. This typographical error did not alter any limits, guidelines or other recommendations for sustaining Te Waikoropupu’s values. NIWA re-issued the corrected report to Mr Yuill in July 2016 (Fenwick 2016a). I understand that the original, uncorrected report was the version that accompanied the WCO application. Annex B to my evidence contains the corrected report.

13. In March 2016, I was one of eight experts engaged by Cawthron Institute, funded in part by Dairy New Zealand, to jointly prepare a report, entitled “Ecosystem health of Te Waikoropupu, for Tasman District Council’s Takaka Freshwater and Land Advisory Group” (Young et al. 2017). I attended a workshop for this project in March, and contributed substantially to that report over December 2016-March 2017.

14. During June-August 2016, I worked with Mr Brian Smith, a freshwater biodiversity expert within NIWA, to prepare a comprehensive desk-top review of Te Waikoropupu Springs’ (and associated aquatic ecosystems) biodiversity. The review (Fenwick & Smith 2016) was sought by Ngati Tama ki te Waipounamu Trust as part of its case for a judicial review of TDC’s process for renewing a water take consent from a bore adjacent to the springs.

15. On 27 April 2017, I delivered a public oral presentation on the ecology and sustainability of groundwater ecosystems at the invitation of Friends of Golden Bay (Dr Don Mead).

16. I offered a similar presentation to Tasman District Council. Mr Joseph Thomas arranged for me to deliver this presentation at Cawthron Institute, Nelson, on 24 May 2017.

17. I have located and incorporated some new information into my evidence, but this additional information does not change the conclusions of any of these reports in any substantive way.

Scope of Evidence

18. My evidence provides an overview of Te Waikoropupu, the springs and contributing aquifers[1] as a series of ecosystems. It draws on a substantial literature review of a small body of research literature on groundwater ecosystems integrated with a wealth of science on the functioning of surface water ecosystems.

19. My evidence addresses:

- Groundwater-dependent ecosystems within the catchment.
- Water quality.
- Te Waikoropupü springs basin biodiversity.
- Subsurface groundwater-dependent ecosystem biodiversity.
- Biodiversity of contributing aquifers.
- Biodiversity of associated GDEs.
- SGDE functioning and ecosystem services.
- Ecological stressors and threats to biodiversity values and ecosystem services.
    - Nitrate
    - Dissolved oxygen
    - Hydrodynamics
    - Organic carbon
- Key interactions within the aquifer ecosystem.
- Conclusions

20. In presenting this evidence, I aim to give you a different perspective; one that demonstrates that Te Waikoropupü is more than the springs, that this remarkably

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[1] Aquifer: underground permeable rock or sedimentary deposit that contains water.

clear water arrives after passing through what is likely to be a very substantial groundwater ecosystem, that this ecosystem is remarkable in its own right and because it underlies many of the springs' special characteristics and values, and that **sustaining the values of the springs requires attention to everything that could affect the groundwater before the water reaches the springs**.

21. Thus, my primary focus is on the ecology of the aquifers because of their role in delivering high quality water to the springs. The springs are a secondary focus.

22. I further note that, as an ecologist and biodiversity scientist, my evidence does not cover the geohydrology and various origins of water emanating from the springs. Rather, it focusses on my understanding of the biodiversity and ecosystems living within these aquifers, the contributions that these ecosystems make to the values associated with the groundwater and the springs, and the threats to these ecosystems and values.

23. This evidence was reviewed by Dr Clive Howard-Williams, Chief Science Advisor, NIWA.

Groundwater-dependent ecosystems[2] within the catchment

24. From hydrological and ecological perspectives, groundwater and surface freshwaters are best regarded as dynamically inter-connected parts of a single water resource, with interactions between the two components varying widely in timing, rate, volume and location, even within small sub-catchments (Hatton & Evans 1998, Winter et al. 1998).

25. The dependence of many freshwater ecosystems on groundwater, either directly or indirectly, and these interactions led to the concept of groundwater-dependent ecosystems (Hatton & Evans 1998; Rohde et al. 2018): "Groundwater-dependent ecosystems (GDEs) are defined as ecosystems that require access to groundwater to meet all or some of their water requirements so as to maintain the communities of plants and animals, ecological processes they support, and ecosystem services they provide" (Richardson et al. 2011: 1).

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[2] An ecosystem is a system of living organisms (including bacteria, other microbes, invertebrate and vertebrate animals) that interact with each other and their non-living environment (e.g., air, water, nutrients, etc.), usually within a defined physical space, to receive and utilise energy and cycle nutrients.

26. This framework and terminology recognises that most of the aquatic habitats and ecosystems within a catchment depend on the catchment's groundwater, often via surface water-groundwater exchanges. Thus, several different GDEs can be recognised within most catchments. Examples of GDEs within the Takaka River catchment include seeps, streams, lakes, rivers, aquifers, springs, and wetlands.

27. Because groundwater flows through and variously connects GDEs within a catchment, the ecological health of an upstream GDE is likely to affect the health of downstream GDEs. For example, the ecological health of the Main Spring GDE will be strongly influenced by the ecosystem health of contributing aquifers.

28. Groundwater ecosystems are recognised as distinct GDEs, termed Subsurface GDEs (SGDE)(DLWC 2002, Tomlinson & Boulton 2010, Serov et al. 2012) and comprise ecological communities that interact with their non-living environment, performing natural ecological processes in the absence of light. SGDEs include all of the life present in their physical space, from microbes to primitive and advanced invertebrates (vertebrates also, but none known from New Zealand aquifers).

29. The contributing aquifers, the springs basin and the Waikoropupü River are three quite distinct GDEs, hydrologically and ecologically linked, but functionally quite different from each other. The "spring basin" (e.g., the large pool or basin surrounding Main Spring vents or boil, Figure 1, from Stewart & Williams 1981) is a transitional GDE within which the aquatic ecology changes from a functionally allochthonous[3], oxygen-consuming aquifer ecosystem to a largely autochthonous (photosynthesis-dominated), oxygen-neutral to producing, riverine ecosystem.

[3] Allochthonous here means largely reliant on chemical energy (organic carbon) imported from sources outside the ecosystem, whereas autochthonous ecosystems rely mostly on energy generated within the ecosystem.

FIGURE 1. Te Waikoropupu Springs.

30. Three main aquifer systems, the Arthur Mable Aquifer (AMA), the Takaka Limestone Aquifer (TLA) and the Takaka Unconfined[4] Gravel Aquifer (TUGA) contain waters differing in ages and isotopic signatures (Stewart & Williams 1981). The WCO identified the AMA as having outstanding characteristics.

31. The matrix comprising each aquifer differs also (Mueller 1987, 1991):

(a) The AMA is mostly confined, karstified[5] marble, extends to >360 m depth, and its porosity was estimated at 8-16%. Its epikarst porosity alone was estimated to be 2.4 - 6.6% (Dr Paul Williams' evidence to this hearing).

(b) The TLA includes some unconfined outcrops and freshwater caves to at least 84 m depth, is 4-62 m thick, and its porosity is unknown. It lies directly on the AMA in the upper half of the valley, and "the lithological boundary between limestone and marble has no distinguishable influence on groundwater flows" (Stewart & Thomas 2008: 4).

(c) The TUGA comprises permeable alluvial gravels and sand, presenting huge surface areas and diverse interstitial spaces. Its porosity varies, and is probably intermediate (perhaps 4-8%). This aquifer links the AMA and

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[4] Confined aquifers are overlain by strata that are largely or completely impermeable, whereas there are no such layers or strata above unconfined aquifers.
[5] Eroded by dissolution to form fissures, caves, tunnels, caverns, etc., typically in limestone or karst deposits (Williams 2008).

both disconnected components of the TLA and, in places, “the AMA and TLA are indistinguishable” (Thomas & Harvey 2013: 25).

32. Both the AMA and the TLA include extensive karstification, which involves development of extensive volumes of fine (<1 - 5 mm) fissures, crevices, cracks and pores (Mueller 1987, 1991, Williams 2008). These vary from fully and permanently saturated to partially and/or temporarily filled seepages and trickles (Williams 2004). These karst habitats are variously integrated with water bearing gravels of the TUGA. Together, these appear to create a very large habitat of complex, interconnected interstices ideal for the bacterial and invertebrate, and which may be nationally significant in geological complexity and size, if not for biodiversity that has evolved within it.

33. Although each aquifer provides slightly different physical habitats, they are variously intercalated, include substantial fine-scale porosity, and are strongly connected hydrologically in places along the catchment (Stewart & Williams 1981; Mueller 1991; Thomas & Harvey 2013). Consequently, the three aquifers are best regarded as a single SGDE within a complex aquifer system.

Water quality

34. As water moves over the land surface and through soils and rock, including through an aquifer, it dissolves minerals from rock and picks up numerous substances and particulate matter, including bacteria and other microscopic organisms. This means that the substances dissolved in groundwater usually increase in concentration with time under ground.

35. Recharge water at any point along an unconfined aquifer’s flow path may introduce further dissolved and particulate substances, including most substances formed or used on and applied to the land surface. Concentrations of these substances also tend to increase along an unconfined aquifer’s flow path.

36. Other substances dissolved in groundwater are consumed or transformed as they pass along a groundwater flow path. Two of these are organic carbon and oxygen.

37. Organic carbon is carried into aquifers from surface environments by recharge or in-flowing water. Whole twigs, leaves and other macroscopic organic matter may be carried below ground in karst systems via dolines, caves or tomos, or it

may enter as dissolved organic carbon (DOC) where physically filtered by soils, sand, gravels, etc. Ultimately, organic matter becomes dissolved in groundwater as a result of biogeochemical processes.

38. In the absence of light and photosynthetic plants, most life in aquifers, therefore, depends upon this dissolved organic carbon from surface environments for its energy. Its concentration tends to decrease along an aquifer's flow path, unless there are new inputs.

39. Organic carbon concentrations are typically low (<1.5 mg DOC/L) in both alluvial and karst aquifers, vary seasonally and usually are higher under farmed compared with forested land (Datry et al. 2005, Hancock & Boulton 2008, Pipan & Culver 2013, Larned et al. 2015). Concentrations in sinking streams may be five times higher, on average, than in percolating water (Pipan & Culver 2013).

40. Dissolved organic carbon was considered to be "the only important light-absorbing constituent of most natural waters" (Davies-Colley & Vant 1987: 416), with the light-absorbing fraction of DOC referred to as "yellow substance" or gelbstoff. This fraction, similar to humic and fulvic acids and derived from terrestrial soils and plant matter, is typically the largest component of DOC in natural freshwaters (Davies-Colley & Vant 1987).

41. It is also the substance which was "undetectably low" in Te Waikoropupu Springs' water (Davies-Colley & Smith 1995: 255), making the springs' water extremely clear and essentially optically pure (Davies-Colley & Smith 1995). That is, the extremely low organic carbon content of Te Waikoropupu Springs water is the reason for its clarity.

42. Oxygen is another dissolved substance important for sustaining healthy aquatic ecosystems in many aquifers. It is also important for sustaining the overall health of the aquifer systems and Te Waikoropupu's biodiversity and ecological values. Dissolved oxygen is carried into an aquifer by recharge water where it is gradually consumed by biogeochemical reactions. Without photosynthesis, there is little or no re-oxygenation of groundwater within a porous aquifer, unless there is direct mixing with air.

43. Consequently, dissolved oxygen concentrations within alluvial aquifers tend to decrease with groundwater's time underground and along an aquifer's flow path (Griebler 2001, Helton et al. 2012). The same appears likely for saturated

(phreatic) parts of karst aquifers. Direct contact with air within conduits, tunnels and caves above the saturated zone (or water table, i.e., the vadose zone), is likely to at least partially re-oxygenate groundwater flowing through these passages. Confined or deeper aquifers containing older groundwater tend to have little (hypoxic) or no (anoxic) dissolved oxygen.

44. The availability of DO determines groundwater’s oxidation-reduction (redox) potential, that is, the propensity for oxidising or reducing chemical reactions, usually mediated by microbes. Thus, DO availability strongly influences biogeochemical processes within an aquifer.

45. Changes from higher DO (oxic) conditions to low (hypoxic) and no DO (anoxic) along an aquifer result in changes in some naturally occurring or human-influenced substances. For example, nitrogen tends to persist as nitrate in oxic environments, but is increasingly transformed to nitrite, ammonium compounds and ammonia as conditions become more anoxic.

46. Other natural or human-derived dissolved nutrients or other substances may be important in groundwater quality, depending on their concentrations and other factors.

47. Most dissolved substances have no effect on organisms and their ecological processes at low concentrations, but many are toxic at some (unknown and/or unnatural) higher concentration. A few substances may be toxic and inhibit ecological activity even at low concentrations.

48. Some naturally occurring substances (e.g., DOC, nitrate, etc.) stimulate ecological activity at lower concentrations, but reduce ecological functioning at higher concentrations.

49. Te Waikoropupu Springs are internationally remarkable for the clarity of their water (Davies-Colley & Smith 1995), as well as being the largest cold-water spring in the Southern Hemisphere. The springs’ water quality is generally high, containing low concentrations of bacteria and other contaminants (Michaelis 1976, Stevens 2010, Young et al. 2017).

50. The expert panel report noted small changes in nitrate, dissolved oxygen, pH and water clarity within Te Waikoropupu Springs water (Young et al. 2017). Individually, such changes in one substance may be not significant. However,

their concurrent change, all towards reduced water quality, indicate potentially adverse change in overall water quality and ecosystem health. Closer management of activities potentially affecting the quality of water emerging from these springs seems necessary.

51. The expert panel report noted the focus of past monitoring was on water quality within the spring basin, not within the unconfined aquifers that supply the springs, which are potentially more directly vulnerable to contamination from land use activities.

Te Waikoropupü Springs basin biodiversity

52. The spring basin's biodiversity was considered to be amongst the most biologically diverse of New Zealand's cold-water springs (Death et al. 2004, Fenwick & Smith 2016). Its 38 species of plants and 54 species of benthic invertebrates (plus another 80 in associated stream habitats) (Fenwick & Smith 2016), make Te Waikoropupu Main Spring basin a nationally significant spring, especially when its unusual submerged flora is considered.

53. The spring basin's rich aquatic flora comprised some unusual plant associations: permanently submerged mosses and liverworts, terrestrial species growing fully submerged, unusual growth forms of two moss species (see Fenwick & Smith 2016 for sources).

54. Animal life in the springs basin included two species apparently endemic to these springs (i.e., occur nowhere else), highest reported population densities of a common freshwater snail (*Potamopyrgus antipodarum*), the only South Island population of an otherwise North Island freshwater amphipod (*Paracalliope karitane*), and northern most populations of two caddis flies (*Hydrobiosis chalcodes, H. johnsi*).

55. The springs were recognised for Main Spring basin's biodiversity as an internationally important wetland complex (based on Ramsar Convention criteria)(Cromarty & Scott 1996), and as an internationally significant water body by the International Union for the Conservation of Nature (IUCN)(Luther & Rzoska 1971). Clearly, they support nationally significant biodiversity and there is evidence of this biodiversity's international significance.

SGDE biodiversity

56. Groundwater life is rarely seen because these environments are difficult to access and because wells or bores are usually designed to exclude all but water. However, most shallow aquifers, world-wide, support significant biodiversity (Hancock et al. 2005) and complex life persists to substantial depths[6] (e.g., 3.6 km below ground; Borgonie et al. 2011, Edwards et al. 2012). Bacteria, Fungi and Archaea (microbes) and more primitive invertebrate animals (e.g., Protozoa, Nematoda, etc.) are amongst the most universal forms of life, and inhabit almost all aquatic habitats, including both oxic and anoxic aquifers.

57. Most bacteria and other microbes in groundwater are closely associated with biofilms, thin layers of bacteria and self-produced organic (gel-like) substances (Brunke & Gosner 1997, Fischer 2002) that coat essentially all surfaces (clay grains to boulders to bedrock) within an aquifer.

58. Biofilms are almost universal on surfaces in wet environments. They create the furry-feeling on human teeth. In many man-made situations, notably food processing and other industries, including medical situations, they must be managed very carefully because they interfere with the passage of liquids by increasing friction, occluding pores, pipes and hoses (e.g., Baveye et al. 1998), among other problems. Amongst the clearest illustrations of bacterial biofilms is that a naval vessel required 18% less power to maintain a speed of 25 knots after bacterial biofilm was removed from its hull (there was almost no visible fouling prior to cleaning)(Haslbeck & Bohlander 1992).

59. The composition of these aquifer microbial communities appears determined primarily by aquifer dissolved oxygen concentrations, groundwater age and human impacts (Griebler 2001; Sirisena et al. 2014). These different microbial communities profoundly affect groundwater quality by transforming dissolved substances into different chemicals, depending on oxygen availability (Chapelle 2000, Griebler 2001).

60. Aquifers throughout New Zealand, including within the Nelson-Marlborough regions, contain significant microbial biodiversity. Research shows a high

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[6] Aquatic organisms are unaffected by hydrostatic pressure (as in an artesian aquifer) because their bodies are >95% water (which is incompressible) and contain no air or gas spaces. Humans are the same, except that pressures in their sinuses and ear canals must be manually equalised when they dive below 2-3 m depth.

microbial biodiversity (>250 likely species) in New Zealand aquifers (Van Bekkum et al. 2006, Sirisena 2014), including in the Takaka Valley (Sirisena et al. 2013).

61. Animal life (unicellular Protozoa and multicellular metazoan invertebrates) also inhabits aquifers world-wide (Griebler & Lueders 2009). Groundwater metazoans, referred to as stygofauna (Humphreys 2000), are invertebrates adapted to life underground (i.e., no body pigments, no or very small eyes, elongated bodies, elongated antennae) (Gibert et al. 1994, Coineau 2000, Gibert 2001). Small body size is another adaptation to subsurface, interstitial life, but some New Zealand stygofaunal macroinvertebrates grow to 20 mm long (Wilson & Fenwick 1999).

62. New Zealand's stygofauna is widespread and diverse. It includes families, genera and species that occur only in New Zealand (Scarsbrook et al. 2003). Exploratory collecting revealed stygofauna in aquifers throughout the country, from Southland to Northland (Fenwick 2000). The stygofauna includes some remarkable, ancient lineages (e.g., Barnard & Barnard 1983) moulded by New Zealand's unique geological history.

63. The few known non-native species comprise small copepods, apparently opportunists that inhabit both surface and groundwaters and translocated to New Zealand in the freshwater supply tanks of ships (Karanovic 2005).

64. More than 50 stygofaunal species are known from one intensively investigated shallow alluvial aquifer (the Selwyn) in Canterbury (Fenwick 2016b). Several species are known from the Waimea and Motueka aquifers near Nelson.

Biodiversity of contributing aquifers

65. The very small amount of equivalent sampling from bores and wells within Takaka Valley revealed both bacteria and stygofauna within the SGDE.

66. Takaka Valley aquifers are known to contain significant bacterial biodiversity, based on a single sampling (Sirisena 2014, Sirisena et al. 2013). Using molecular techniques, microbial diversity within the groundwater[7] was assessed as complex (11-12 species, 42-55% of maximum species/location in New Zealand),

[7] Bacteria are usually much (>10-100 times) more abundant and more diverse in groundwater biofilms than suspended (planktonic) in groundwater (Brunke & Gosner 1997, Griebler & Lueders 2009).

and similarly diverse to groundwater bacterial communities at most of the 100 locations sampled across New Zealand (Sirisena 2014).

67. Two of the three bores sampled for stygofauna rom confined aquifers in the valley (Kotinga Road & old hotel corner, no details available; apparently into the TUGA (Joseph Thomas, pers. comm. 23 March 2018)) contained stygofauna: species of an undescribed (new) amphipod crustacean, *Paraleptamphopus* sp. These collections confirm the presence of stygofauna within the valley's aquifer system. I expect the total stygofaunal biodiversity of the three Takaka Valley aquifers to include many more stygofaunal species, some of which are likely to occur nowhere else.

Biodiversity of associated GDEs

68. Eight collections (held by NIWA) from the valley's springs and caves produced two undescribed species of stygofaunal amphipods. Yet other collections contained specimens of a new genus and two recently named species of isopods (*Bilistra millari, B. cavernicola*), known only from the region's karst springs and caves (Sket & Bruce 2004).

69. A relatively intensive survey of seep, spring and cave stream habitats across New Zealand for one group of very small snails discovered six species in caves and small springs in Takaka valley (Haase 2008). Three of these (*Opacuincola caeca, O. lentesferens, O. geometrica*) are known from single cave streams in the valley. A fourth (*O. takakaensis*) was reported from several caves and cave streams, but only within the valley. The other two (*O. ignorata, Catapyrgus fraterculus*) are each known from a cave stream at one location within the valley, plus another location beyond the valley (Haase 2008).

70. The few available collections and species confirm the presence of stygofauna in these aquifers. The diversity of taxa from groundwater linked habitats (seeps, springs, cave streams) in the valley indicates that at least some of the groundwater species are likely to be short-range endemic species[8] (because of barriers to their dispersal), with inherent high biodiversity values. Based on this biodiversity and the diversity of physical habitats within the aquifers, I expect that

[8] Short-range endemic species have naturally small geographic distributions, making them more vulnerable to habitat loss, habitat degradation and climate change than more widely distributed species (see Harvey et al. 2011).

the Takaka Valley stygofaunal biodiversity is substantial and likely to be regionally, if not nationally, significant in terms of diversity and endemicity.

71. This bacterial and stygofaunal biodiversity also confirms the presence of groups required for a functional ecosystem within the aquifers linked to Te Waikoropupu Springs.

SGDE functioning and ecosystem services

72. Because groundwater biodiversity is largely hidden, notoriously heterogeneous (Danielopol et al. 2000) and difficult to access, there is limited understanding of the extent of groundwater biodiversity and its contribution to the ecology of SGDEs and surface GDEs (Gibert et al. 1994).

73. Despite this very incomplete knowledge, it is now well-established that natural, functioning ecosystems occur in most aquifers (Ward et al. 2000, Hancock et al. 2005). These ecosystems are communities of microbes and stygofauna that interact with each other, and with their non-living environment, performing natural ecological processes in the absence of light and photosynthetic plants.

74. As part of their natural functioning, SGDEs modify their environment, providing ecosystem services that benefit the wider environment and humans (e.g., Tomlinson & Boulton 2010, Fenwick 2016). Biofilms within SGDEs concentrate and transform dissolved and fine particulate matter (including bacteria), a vital part of natural bioremediation or cleansing that occurs in aquifers (Chapelle 2000, Handley et al. 2013, 2015, Wrighton et al. 2014). These biofilms utilise DOC and other substances, resulting in net losses of carbon from the ecosystem via aerobic respiration (Williamson et al. 2012, Di Lorenzo & Galassi 2013, Wrighton et al. 2014).

75. Similar processes occur within karst and epikarst habitats (e.g., Culver et al. 1992, Pipan & Culver 2013, 2018), and seem inevitable within the epikarst associated with AMA and TLA.

76. Biofilm bacteria also transform several other substances that would otherwise degrade water quality (e.g., polyaromatic hydrocarbons, such as naphthalene, from coal, tar and incomplete combustion of organic matter (Madsen et al. 1991, Fischer 2003)). In particular, they also facilitate denitrification, the transformation of nitrate into nitrogen. Bacterial denitrification appears to occur principally at

hypoxic to anoxic microsites within aerobic aquifers (e.g., Koba et al. 1997, Gold et al. 1998, Rivett et al. 2008), and can result in significant (mean 50%, range: 29-75%) nitrate attenuation within some aquifers (e.g., Stenger et al. 2013, Elwan et al. 2015).

77. The stygofauna delivers additional ecosystem services. Stygofauna ingest and digest bacteria (Sinton 1984, Fenwick et al. 2004,), keeping finer aquifer pore spaces open and water flowing through these pore spaces (Nogaro et al. 2006, Boulton et al. 2008, Tomlinson & Boulton 2010).

78. While grazing biofilms and moving within an aquifer, the stygofauna mechanically tills or disturbs the aquifer particles, turning them, abrading adhering biofilm, reworking and repositioning finer particles, and probably altering sediment matrices (Fenwick et al. 2004). This process, termed bioturbation and widely known in aquatic ecosystems (e.g., Mermillod-Blondin 2004, Wilkinson et al. 2009, Kristensen et al. 2012), is akin to the role of earthworms in healthy soils. In groundwater, bioturbation both stimulates microbial activity, leading to biogeochemical transformation of contaminants, and reduces any clogging to facilitate water flows (bioirrigation) that replenish dissolved oxygen (bioaeration) (Nogaro et al. 2006, Boulton et al. 2008) and maintain aerobic, oxidising conditions with improved water quality.

79. Similar stygofauna perform similar bioturbation and bioaeration processes within the extensive fine (<1 – 10 mm) crevices and pores comprising much of the epikarst (e.g., Culver et al. 1992, Simon et al. 2003, Pipan & Culver 2013).

80. The overall effects of these SGDE processes, termed ecosystem services,include improving groundwater quality and its suitability for human uses, maintaining an aquifers' ability to conduct water, and maintain its yield of water for abstraction (e.g. Sinton 1984, Datry et al. 2003, Boulton et al. 2008, Tomlinson & Boulton 2008, 2010). These effects sustain many of the human values associated with groundwater, notably human health and economic values (Fenwick 2016). These effects also contribute to the natural and human values associated with many rivers and streams, which receive smaller to larger contributions from groundwater.

81. The extensive, fully and partially saturated, fine (<1-5 mm wide) fissures, crevices, cracks and interstices of the Takaka valley aquifers (Williams 2008),

especially within the vast volume of epikarst comprising the upper 10 m or more[9] of the historically eroded AMA and the upper valley TLA (Mueller 1991, see Thomas & Harvey 2013, Fig 14), have a fine-scale porosity, estimated at 2.4 to 6.6% fine scale porosity (Paul Williams' evidence) represents a huge surface area available for biofilm development and a very substantial volume of fine spaces for stygofauna habitat.

82. Thus, there is good evidence that these same ecosystem services are responsible for the very substantial natural bioremediation capacity within this aquifer system, especially when the very large volume of the AMA and TLA's epikarst and the TUGA's saturated zone (5->20 m thick) are included (Thomas & Harvey 2013).

83. I believe that these processes and ecosystem services underlie the remarkable clarity of Te Waikoropupü Springs' water. The virtually complete removal of coloured organic matter (yellow substance; one type of dissolved organic carbon) from spring water, initially attributed to "chemical adsorption on the calcite mineral surfaces of the rock" (Davies-Colley & Smith 1995: 255), almost certainly involves both chemical adsorption, uptake by biofilms (Fischer 2003) and degradation by the SGDE(e.g., Boulton et al. 2008, Tomlinson & Boulton 2010, Pipan & Culver 2018).

Ecological stressors and threats to biodiversity values and ecosystem services

84. Surface water quality is well-known to affect aquatic ecosystem health (AEH), with numerous dissolved and suspended substances degrading AEH when beyond critical limits (shortages and over-supplies) (e.g., Hynes 1972, Davies-Colley & Wilcock 2004). This applies equally to groundwater and to SGDE AEH (e.g., Sinton 1984, Notenboon et al. 1994, Korbel et al. 2013, Korbel & Hose 2015, Espanol et al. 2017).

85. Although there may be some physical filtration and chemical transformations en route to and within an aquifer, most dissolved substances in surface waters will enter the groundwater if there is an exchange pathway between surface and groundwater. Thus, land-use activities can markedly change the quantities and types of dissolved substances and fine particulate matter entering groundwater,

[9] Thomas & Harvey (2013) reported karstification to >100 m within the AMA upstream of Te Waikoropupu.

and these substances and fine particles may have important effects on groundwater quality.

86. As with surface water ecosystems, there is good evidence that human land-use activities frequently affect SGDE health by changing water quality and/or groundwater hydrology (e.g., Sinton 1984, Fenwick et al. 2004, Boulton et al. 2008, Stein et al. 2010, Hartland et al. 2011, Di Lorenzo & Galassi 2013, Korbel et al. 2013).

87. Harmful concentrations of dissolved substances common in freshwaters are known for some surface water organisms and habitats, and there are established maximum or guideline concentrations for sustaining the ecological health of surface water ecosystems for several common contaminants e.g., (Hickey 2016, MfE 2017).

88. No guideline concentrations have been determined for protecting SGDE AEH from any contaminant. Harmful concentrations of common pollutants (e.g., nitrates) are not known for any stygofauna world-wide or in New Zealand. One study indicated that stygofauna were more sensitive to some pollutants than their surface water equivalents (Mosslacher 2000), but robust evidence is lacking.

Nitrate

89. Nitrate is a key contaminant of aquifers throughout New Zealand and in the Takaka valley aquifers (Daughney & Randall 2009, Stevens 2010). Experience elsewhere shows that high concentrations of nitrate can occur in groundwater over large areas (Hayward & Hansen 2004) and persist for decades (Stewart et al. 2011).

90. Although there is no unequivocal evidence that nitrate is harmful to stygofauna and SGDEs, its widely known toxicity to surface freshwater invertebrates at relatively low concentrations (e.g., Hickey 2013b, MfE 2017) almost certainly means that nitrate is similarly harmful to SGDE health.

91. The physiology of crustaceans, the dominant invertebrates in most SGDEs, is impaired by nitrate (and its hypoxic states: nitrite and ammonia) (Alonso & Camargo 2003, 2006, Soucek & Dickinson 2012, Hickey 2013a). Some evidence indicates that crustaceans are more sensitive than other invertebrate groups to

nitrate, whereas other evidence suggests the opposite (e.g., Soucek & Dickinson 2012).

92. One of the new insights is that nitrate toxicity is reduced by dissolved minerals (i.e., increased water hardness) and by elevated chloride also (e.g., Hickey 2016). Although most waters within the Takaka Valley-Te Waikoropupü system are moderately hard (dissolved mineral concentrations) (e.g., Michaelis 1976, Stevens 2010) and Main Spring has elevated chloride, waters in the AMA and TUGA lack any chloride enrichment and hardness of TUGA water is considerably lower (Stevens 2010).

93. The toxicity guideline concentration (1.1 mg/L NO_3-N) for protecting high conservation value surface water ecosystems (Hickey 2016) is consistent with the National Objectives Framework concentration (≤1.0 mg/L NO_3-N) for protecting the Compulsory National Values for rivers (MfE 2017). However, there are no equivalent toxicity or objective concentrations for stygofauna or for groundwater ecosystems because no stygofauna species have been adequately tested.

94. I consider that the lower trigger concentration range recommended by Young et al. (2017) is appropriate for Te Waikoropupu Springs' basin ecosystem for these reasons. First, present day nitrate concentrations (0.42 mg/L NO_3-N on average during 2017 (Andrew Yuill, pers. comm., 1 Feb 2018) represent a 30% increase over 1970-71 concentrations (0.31 mg/L NO_3-N, Michaelis 1976). Second, Young et al's (2017) 0.4-0.5 mg/L NO_3-N concentration was recommended as a trigger, a concentration which, if met or exceeded, would trigger or initiate a process to evaluate the significance of the change and to determine what, if any action was needed to ensure that the health of the aquatic ecosystems involved (SGDE, springs basin) will not be harmed.

95. Nitrate concentrations within the springs' discharge water are essentially averages for their contributing groundwaters, so concentrations in some parts of each aquifer will be considerably higher than the average in water emerging at the springs (and, conversely, lower in some other parts of the aquifers). Two matters thus require resolution.

96. First, what aquifer nitrate concentrations should be established for sustaining the SGDE, which influences the quality of much of the water reaching Te Waikoropupu Springs? The toxicity of nitrate to stygofauna is unknown and may

differ than for river and lake invertebrates, especially when combined with exposure to reduced dissolved oxygen or temporary anoxia. A conservative approach would dictate maintaining concentrations in the high biodiversity value aquifers below the toxicity guideline and compulsory national value for rivers concentrations (1.1 and <1.0 mg NO_3-N/L, respectively, Hickey 2016, MfE 2017). Therefore, Young et al's (2017) range seems appropriate as an interim trigger concentration for managing nitrate in the SGDEs, in tandem with an objective of sustaining groundwater ecosystem health[10].

97. Second, SGDE nitrate concentrations must be monitored within the groundwater at multiple points within the aquifers in order to sustain Te Waikoropupu Springs' water quality and to sustain the SGDEs involved in remediating this water. Monitoring at multiple points in the vicinity of nitrate inputs to these aquifers (i.e., unconfined aquifer recharge areas on the valley floor) seems essential to protect all parts of this very large bioremediation system and to facilitate implementing any actions to manage inputs, where necessary. Monitoring nitrate (and other key substance) concentrations within the springs will detect any gross changes only, probably only long (eight years, Stewart & Thomas 2008) after the cause of the change. In consequence, monitoring springs discharge water would provide little guidance for management action.

Dissolved oxygen

98. Dissolved oxygen concentrations naturally differ widely between aquifers, are typically moderate to low in most unconfined aquifers, and some groundwaters, notably confined aquifers, lack dissolved oxygen (i.e., are anoxic) (Rosen 2001). Concentrations within shallower aquifers vary seasonally and spatially (e.g., Larned et al. 2015) and generally decrease along an aquifer's flow-path.

99. Concentrations of dissolved oxygen in Te Waikoropupu Springs averaged 6.6 mg/L in 1970-71 (within groundwater exiting the spring vent; Michaelis 1976) and ranged between 5.5 and 5.7 mg/L in early 2016 (close to the spring basin surface; Young et al. 2017). Some of the indicated c. 1 mg/L or 20% decrease on DO concentrations may be due to differences in measurement methods over the 46 intervening years (Young et al. 2017).

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[10] Wellington Regional Council's proposed Natural Resources Plan seeks to establish this objective for managing nitrate in its groundwaters.

100. Dissolved oxygen is essential for sustaining most stygofauna (Malard & Hervant 1999), even though truly groundwater (stygobitic) species appear to require less dissolved oxygen than their epigean (surface-dwelling) counterparts (Spicer 1998, Mosslacher 2000, Wilhelm et al. 2006), and some stygofauna show various behaviours to avoid hypoxia (reviewed by Fenwick et al. in prep.).

101. Larger stygofauna species appear restricted to oxygenated (oxic to hypoxic) aquifer habitats, but some smaller invertebrates, particularly Archaea and protozoans (e.g., amoebae, ciliates) probably live with little or no oxygen. No macroinvertebrates are known to inhabit anoxic aquifers in New Zealand. The ability of larger invertebrate stygofauna to survive and persist with very little or no dissolved oxygen is poorly understood and seems unlikely.

102. Consequently, dissolved oxygen availability is considered fundamental to sustaining aerobic SGDE health (Mosslacher et al. 1996, Malard & Hervant 1999) and sustaining SGDE ecosystem services (Tomlinson & Boulton 2008).

103. Hypoxia slightly increases the sensitivity of some epigean crustaceans to elevated nitrite or ammonia (nitrate is usually reduced to these substances in hypoxic to anoxic environments) (Broughton et al. 2018). Low dissolved oxygen concentrations in groundwater may increase toxicities of some contaminants (nitrate, ammonia) to some stygofauna slightly, although the effect is untested for stygofauna and the groups predominating in groundwater.

104. This information indicates the need for monitoring and managing groundwater within the three aquifers to maintain their near-natural dissolved oxygen concentrations in order to sustain Te Waikoropupu Springs' water quality and associated values.

105. It also shows that any potential synergistic effects due to land use effects and/or contaminants within the three aquifers must be considered and managed to sustain the values associated with both these aquifers' and Te Waikoropupu's water.

Hydrodynamics

106. Because there is very limited re-oxygenation of water within saturated parts of aquifers (Boulton et al. 2008), changes in groundwater velocity will affect ambient dissolved oxygen concentrations (Hoehn 2001).

107. Groundwater velocities are driven by water level differences or hydraulic gradients within an aquifer. Thus, reduced groundwater levels, caused by reduced recharge and/or groundwater abstraction, can result in slower replenishment and lower dissolved oxygen (and DOC) concentrations, potentially compromising SGDE health.

108. Water levels within most unconfined aquifers vary naturally with season and climate variation. Within unconfined parts of the AMA (and presumably the TUGA), water levels fluctuated by almost 5 m in some years between 2003 and 2013 (Thomas & Harvey 2013), and Paul Williams (evidence, this hearing) reported vertical fluctuations of about 30 m in mid-valley bores penetrating the marble.

109. Maintaining aquifer water level regimes similar to those during the last c. 20 years seems likely to minimise any potential reductions in DO and other velocity-related changes that might otherwise affect the SGDE.

Organic carbon

110. Most groundwater is naturally low in available food (dissolved and particulate organic carbon (DOC, POC) (e.g., Coineau 2000, Poulson & Lavoie 2000, Williamson et al. 2012, Larned et al. 2015). Increased DOC availability stimulates biofilm development and increased dissolved oxygen consumption by biofilm microbes (e.g., Simon & Buikema 2003). Beyond some undefined limits, increased DOC inevitably leads to reduced oxygen availability, which reduces the stygofauna's ability to control biofilm development (Boulton et al. 2008).

111. Uncontrolled growth of biofilms may clog progressively larger pore spaces within an aquifer, reducing water velocities and dissolved oxygen replenishment, at least at finer scales (Baveye et al. 1998, Seifert & Engesgaard 2007, Bottero et al. 2013).

112. The resultant shift towards hypoxic and anoxic conditions will change microbial communities (e.g., Cheung et al. 2014), favouring bacteria that use different metabolic pathways and produce different respiratory end-products (i.e., from CO_2 to H_2S) (Chapelle 2000). Such changes may significantly degrade water quality, initially at smaller (<10-100 mm) scales. Conceivably, this process, unchecked, may compromise the health of larger parts of an SGDE, degrade

water quality further and reduce groundwater yield from the aquifer (Boulton et al. 2008, Fenwick 2016).

113. These changes in chemistry and microbial communities can occur within saturated aquifers, whether alluvial, epikastic, karst or other type (e.g., Holsinger 1966, Sinton 1984, Simon et al. 2003). I believe that such changes could occur, at least at smaller scales, within parts of all three Takaka Valley aquifers, if their natural bioremediation processes are disrupted. Water quality in Te Waikoropupu Springs could be compromised as a consequence of such changes in the aquifers.

Key interactions within the aquifer ecosystem

114. Interactions between hydrology, water quality and the biological components of aquifer ecosystems are summarised in a simplified model for explanatory purposes (Figure 1).

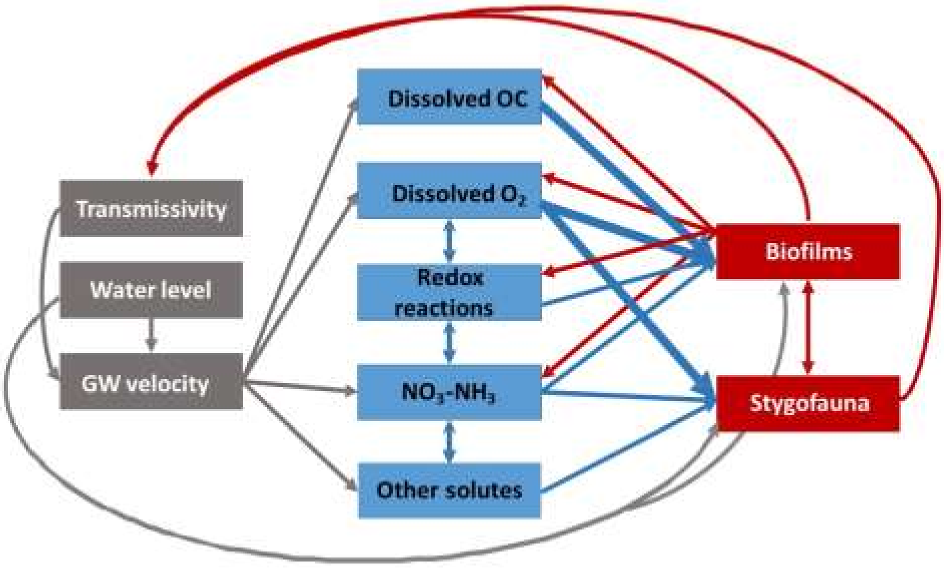

Figure 1. Critical interactions between groundwater water quality, biofilms and stygofauna, and aquifer geohydrology that are considered to underlie Te Waikoropupū Springs' optically near-pure water. Fenwick et al. (in prep.).

115. Water level (left side, grey box) drives groundwater velocity, determining the delivery rate of dissolved substances within the aquifer. Higher velocities ensure that much of the aquifer is oxic and the biofilm bacteria utilise aerobic respiration to oxidise organic carbon, also bound within biofilms on aquifer matrix surfaces. Stygofauna browse this biofilm reducing its thickness, digesting bacteria, mechanically dislodging biofilm and finer sediment particles, opening finer pore

spaces to maintain the aquifer's ability to transmit water and maintain groundwater velocity.

116. The balance of this ecosystem can be disrupted by insufficient replacement of dissolved oxygen (stygofauna cease bioturbation; anaerobic bacterial respiration creates a chemically-reducing environment, nitrate converted to ammonia (toxic to stygofauna), other metabolic by-products degrade water quality). These changes mean that, if conditions permit, biofilms may develop unchecked, reduce aquifer transmissivity, reduce water velocity and a cascade of change results as increasingly reducing biogeochemical reactions occur.

117. Similar disruption may occur if nitrate (or other dissolved substance) concentrations become toxic to stygofauna or if excessive organic carbon enters the ecosystem.

Conclusions

118. Te Waikoropupü comprises three inter-connected ecosystems: the contributing aquifers, the spring basins and the Springs River. The Main Springs' basin's biodiversity is well-documented and considered nationally and internationally significant.

119. Many of the values associated with Te Waikoropupu Springs are attributable to the quality of their water, especially its clarity. This water comes from a hydrologically and ecologically linked system of three aquifers. These aquifers include high surface-area, epikarstic formations and permeable gravels, which together appear to comprise very significant habitat, both in volume and physical habitat diversity, for a natural aquifer ecosystem.

120. The biodiversity of the aquifer system is very poorly known, but is probably rich in microbial and stygofaunal diversity, based on the diversity of physical habitats within the three aquifers and their long geological history.

121. This aquifer microbial and stygofaunal biodiversity delivers the very significant ecosystem services, which, when in balance, remove essentially all organic matter (especially coloured DOC) from water arriving at the springs. Thus, the aquifer biodiversity and its balanced ecosystem functioning underlie the very substantial and diverse human values associated with both the springs and the aquifer, especially the spring's remarkably clear water. Thus, sustaining the

springs' water clarity and other values requires careful management to sustain the aquifers' biodiversity and ecosystem services.

122. The main threats facing the aquifer and spring basin ecosystems are changes in aquifer water quality. However, there are no directly relevant water quality guidelines for ensuring the health of groundwater ecosystems, nor any useful measures of groundwater ecosystem health.

123. Thus, the conservative trigger concentrations recommended for the springs water by Young et al. (2017) for nitrate, dissolved oxygen and dissolved phosphorus, plus maintaining water levels to sustain flows and delivery of dissolved oxygen, are essential to minimise further declines in water quality, both within the springs and within the aquifers. These same trigger concentrations for managing groundwater quality appear appropriate to the aquifers also. Trigger concentrations for dissolved organic carbon are essential also, but further investigation is required to determine appropriate concentrations for this substance in different parts of the aquifer system.

124. Management to sustain these important ecosystems must include monitoring water quality and groundwater ecosystem health[11] within unconfined aquifers beneath and immediately downstream of areas of intensive land use, especially where porous media (alluvial deposits, epikarst) provide pathways for contaminants to enter the aquifers. This is essential to detect any adverse effects early and to identify the sources of the effects, so that management action can be timely and appropriately targeted to protect all biodiversity and ecological values in the overall system, especially the balanced ecological services that deliver Te Waikoropupu's remarkable water.

Graham Fenwick

28 March 2018

[11] Wellington Regional Council's proposed Natural Resources Plan identified aquatic ecosystem health as its primary objective for managing nitrate concentrations within its groundwaters (http://www.gw.govt.nz/assets/Plans--Publications/Regional-Plan-Review/Proposed-Plan/Chapter-3-Objectives.pdf : 41-44).

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Annex A Curriculum vitae: Graham Fenwick

Qualifications

1993	Dip. Business Administration, University of Canterbury.
1984	Ph.D., Marine Biology, University of Canterbury.
1975	M.Sc., Zoology, University of Canterbury.
1972	B.Sc., Botany & Zoology, University of Canterbury.

Experience

2006-present	Assistant Regional Manager, NIWA, Christchurch.
Feb 2002-Oct 2006	Scientist/Principal Scientist and Group Manager, NIWA Christchurch.
July 1998-2002	Contract scientist to NIWA.
Jan-Dec 1997	Research and Teaching Associate, Department of Management, University of Canterbury.
Jan 1990-Dec 1996	Lecturer in Marketing (fixed term), Department of Management, University of Canterbury.
Feb 1989-Oct 1989	Marketing consultant to local businesses.
Nov 1986-Jan 1989	Marketing Consultant, Target Services Group Ltd and the Consulting Group of Horwath & Horwath (N.Z.) Ltd.
Nov 1985-Oct 1986	Research Associate, Department of Zoology, University of Canterbury.
Jan 1984-Oct I985	Post-doctoral fellow, Chemistry Department, University of Canterbury, funded by Sea Pharm Inc., Princeton, U.S.A.
Nov 1982-Oct 1983	Marine Biologist, University of Canterbury.
Jan 1980-Feb 1983	Ph.D. thesis research.
Sept 1977-Dec 1979	Marine Biologist, Department of Zoology, University of Canterbury.
Apr-Aug 1977	Assistant Curator (fixed term), Department of Coelenterates and Crustacea, The Australian Museum, Sydney, Australia.
Nov 1976-Mar 1977	Marine Scientist and Deputy Leader, University of Canterbury Snares Islands Expedition.
Jan-Nov 1976	Marine Benthic Ecologist, Estuarine Research Unit, Zoology Department, University of Canterbury.
Nov 1974-Oct 1975	Marine Biologist, Department of Biology, Memorial University of Newfoundland, St Johns, Canada.
Mar 1973-Oct 1974	M.Sc. thesis studies.
Nov 1971-Jan 1972	Research Assistant, University of Canterbury Antarctic

Research and consulting contributions

Refereed publications:

1. Fenwick, G.D. 1973. Breeding biology and population dynamics of the Weddell seal, *Leptonychotes weddelli*: a review. *Mauri Ora 1*: 29-36.
2. Fenwick, G.D. 1975. Decapoda collected by the Auckland Islands Expedition 1972-73. Pp. 126-135. *In:* Yaldwyn, J.C. (editor). *Preliminary results of the Auckland Islands Expedition 1972-73*. Department of Lands and Survey, Wellington, N.Z. 440 pp.
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15. Fenwick, G.D. & D.H. Steele. 1983. Amphipods of Placentia Bay, Newfoundland. *Memorial University of Newfoundland Occasional Papers in Biology 7*: 1-22.
16. Lowry, J.K. & G.D. Fenwick. 1983. The shallow water gammaridean amphipods of the subantarctic islands of New Zealand and Australia. Part 1. Gammaridae. *Journal of the Royal Society of New Zealand 13*: 201-260.
17. Fenwick, G.D. 1984. Life history and population biology of the giant ostracod *Leuroleberis zealandica* (Baird, 1850) (Myodocopida). *Journal of Experimental Marine Biology & Ecology 77*: 255-289.

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1. Fenwick, G.D. 1987. Organic carbon pathways in the Canterbury groundwater ecosystem and the role of phreatic crustaceans. Unpubl. report to National Water & Soil Conservation Organization, 84 pp.
2. Fenwick, G.D. 1998. Groundwater invertebrates as potential indicators. Appendix 1. In: Bright, J; Bidwell, V.; Robb, C.; Ward, J. Environmental performance indicators for groundwater. Ministry for the Environment Report No. 4306/1 (Environmental perormance indicators Technical Paper No. 38 Freshwater). 21 pp. (http://www.mfe.govt.nz/publications/ser/tech-report-38-freshwater-may98.pdf)(http://www.mfe.govt.nz/publications/ser/tech-report-38-freshwater-appendix-1.pdf
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9. Fenwick, G.D.; John, A. 2007. Quick guide to New Zealand freshwater Isopoda. http://www.niwa.co.nz/__data/assets/pdf_file/0003/59277/Isopoda.pdf
10. Fenwick, G.D.; John, A. 2007. Quick guide to free-living orders of New Zealand freshwater Copepoda. http://www.niwa.co.nz/__data/assets/pdf_file/0010/59275/Quick_guide_to_the_NZ_freshwater_Copepoda.pdf
11. Fenwick, G.D.; Wilson, G.D.F. 2007. Field guide to New Zealand phreatoicid isopods. *NIWA Biodiversity Report*. 24 pp.
12. Fenwick, G.D. (one of several contributors) 2009. Indigenous biodiversity and water management in Canterbury. Paper prepared for Steering Group, Canterbury Water Management Strategy.

Popular articles

1. Scarsbrook, M., Fenwick, G., Radford, J. 2000. Living groundwater: studying the fauna beneath our feet. *Water & Atmosphere 8(3)*: 15-16.
2. Fenwick, G.; Scarsbrook, M. 2002. The rich biodiversity in our groundwater. *Aquatic Biodiversity & Biosecurity No. 2*.
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1. Invited participant: Environmental Futures Network Research Working Group: Evolution of short-range endemic taxa associated with troglobitic and groundwater ecosystems. Meeting 1, University of Adelaide, 20-21 Jan 2007.
2. Host: Environmental Futures Network Research Working Group: Evolution of short-range endemic taxa associated with troglobitic and groundwater ecosystems. Meeting 2, NIWA, Christchurch, 15-16 June 2007.
3. Invited participant: Environmental Futures Network Research Working Group: Evolution of short-range endemic taxa associated with troglobitic and groundwater ecosystems. Meeting 3, Fremantle, Western Australia, 20-21 Sept 2008.
4. Stakeholder workshop, Lyttelton Port of Christchurch Ltd's capital dredging programme. Lead workshop to identify natural values, agree

monitoring priorities and locations for priority monitoring stations. 25 Sept 2014.
5. Marine ecology expert, Port Otago Ltd Maintenance dredging consent renewal. Public open days, 14-15 Oct 2015.

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35. Fenwick, G.D. 2017. Managing groundwater for our region's future. Oral presentation to a public meeting of Friends of Golden Bay, 7-30 pm, 27 April 2017. Sacred Heart Catholic Church Hall, c. 80 present. http://www.gbweekly.co.nz/wp-content/uploads/2017/05/2017_may12GBWeekly.pdf

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Evidence and appearances as expert witness

1. Fenwick, G.D. 2003. Pigeon Bay Aquaculture Ltd application for a resource consent to extend an existing marine farm in Pigeon Bay, Banks Peninsula. Environment Canterbury. Evidence: marine ecology, especially subtidal and intertidal benthos. 6 pp. 1 August 2002. Successful.
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4. Fenwick, G.D. 2004. Marlborough Mussel Co. Ltd applications for resource consents for seven marine farms in northern Banks Peninsula. Environment Canterbury. Evidence: effects on sediments, benthic fauna, intertidal rocky shore biota, introductions of non-endemic pest

species. Recommendations for monitoring. 49 pp. 12 March 2004. Partially successful.

5. Fenwick, G.D. 2005. Pegasus Bay Aquaculture Joint Venture application for resource consent for a 2500 ha farm in central Pegasus Bay. Environment Canterbury. Evidence: benthic ecology and effects of the proposed farm; recommendations for monitoring and recommendations for monitoring. 31 pp. 17 March 2005. Successful.
6. Fenwick, G.D. 2005. Southern Seas Aquaculture Ltd appeal before the Environment Court of New Zealand decision against its application for marine farming consent in Port Levy, Banks Peninsula. Evidence: marine environment, benthic ecology, intertidal ecology, 25 pp. Rebuttal, 9 pp. 24 May 2005. Successful.
7. Fenwick, G.D. 2005. Christchurch City Council application for resource consents for installation of an ocean outfall and for discharge of wastewater into Pegasus Bay. Environment Canterbury. Evidence: benthic ecology of the Avon-Heathcote Estuary and of Pegasus Bay, including surf clams. 30 pp. 10 June 2005. Successful.
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15. Fenwick, G.D. 2018. Water quality & groundwater ecosystem health. Oral presentation to commissioners to support technical evidence on behalf of Wellington Regional Council on water quality in regard to objectives and submissions for aquatic ecosystem health and mahinga kai (Objective O25) of the proposed Natural Resources Management Plan. 12 February 2018.

Annex B Revised version of Fenwick 2015: cited herein as Fenwick 2016.

Sustainability of Te Waikoropupu Springs' aquifer ecosystems

Prepared for Andrew Yuill

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Prepared by:
Graham Fenwick

For any information regarding this report please contact:

Graham Fenwick
Principal Scientist
Biodiversity
+64 3 343 8051
graham.fenwick@niwa.co.nz

National Institute of Water & Atmospheric Research Ltd
PO Box 8602
Riccarton
Christchurch 8011

Phone +64 3 348 8987

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Contents

Introduction

Background

Te Waikoropupu Springs emerge from a complex of aquifers[12] (for convenience here called the Te Waikoropupu Springs aquifer complex (WaiSAC) and, because of the extremely high natural, ecological, biodiversity, spiritual, cultural and economic values associated with this remarkable feature, work towards ensuring that their values are sustained has commenced. This initiative seeks a Water Conservation Order to sustainably manage the springs themselves, plus the surface and ground waters that supply and sustain them.

NIWA was requested to recommend numerical water quality limits for water in these aquifers, based on a desk-top evaluation of available information on groundwater ecosystem responses to key water quality variables.

Scope and limitations

This report provides recommendations based on a preliminary analysis of scant data and available information, and its recommendations must be regarded as tentative. A more rigorous water quality guideline and limit setting process, ideally backed by a more substantial body of research information, is essential to review and revise these recommendations as soon as practical.

The approach taken here was to review the limited available information on stygofauna tolerances to a few key water quality variables and compare this toxicity information with the relevant concentrations in New Zealand's surface water quality guidelines (i.e., the ANZECC guidelines ((ANZECC & ARMCANZ 2000)). Effects of water quality on aquifer microbes, notably those comprising biofilms, are not considered here, despite their importance in aquifer ecological functioning and the established relationships between water quality and both suspended and attached (i.e., non-biofilm and biofilm, respectively) bacterial community composition (Flynn et al. 2013; Sirisena et al. 2014).

Two key variables, organic carbon and dissolved oxygen, are not covered by the ANZECC guidelines, but are essential for most groundwater ecosystems. Their concentrations vary naturally in groundwaters, as well as being influenced indirectly via human activities.

Groundwater quality

Water quality generally is a measure of the extent to which water and the substances that it contains is fit for purpose, either for human purposes and/or for natural ecosystem functioning. Several categories of substances may be involved in water quality, such as toxicants (e.g., metals and other chemicals that are toxic in low concentrations), others that are resources at low concentrations but toxic at high concentrations (e.g., nitrate and other nutrients), and other resources which are essential for life and may interact with each other (e.g., dissolved oxygen and dissolved organic carbon).

The quality of New Zealand's surface freshwaters is managed in large part using the ANZECC guidelines (ANZECC & ARMCANZ 2000). These guidelines were intended "to achieve the sustainable

[12] Te Waikoropupu Springs water is considered to originate from at least three aquifers, the Arthur Marble, Takaka Limestone and Takaka Valley Unconfined Gravel aquifers (Stewart & Williams 1983). This has significant implications for managing the springs' water, but these complexities are beyond the scope of this report.

use of Australia's and New Zealand's water resources by protecting and enhancing their quality while maintaining economic and social development" (ANZECC & ARMCANZ 2000: xii), indicating that any guideline concentrations or limits for protecting a high conservation value surface water body should be more stringent than the ANZECC guideline values. The guidelines are, however, the only available and comprehensive set of research-based and ecologically meaningful concentrations of several important potential toxicants for surface freshwaters.

The water quality requirements for sustaining groundwater ecosystems and their biodiversity components are poorly researched and poorly understood internationally and in New Zealand. This applies even for the small set of key variables discussed here: organic carbon, dissolved oxygen, nitrate and ammonia. Water quality within the WaiSAC is poorly defined, and few relevant data are available. Thus, any limits for water quality variables to sustain the WaiSAC and its ecosystem must be very tentative, and regarded as very preliminary until: (a) more comprehensive monitoring data on of all relevant water quality parameters are available, (b) a rigorous limits setting exercise can be completed, and (c) there is a much better understanding of the water quality requirements for sustaining groundwater ecosystems generally and for the WaiSAC in particular.

In the absence of the underpinning science outlined above, water quality guidelines for the proposed Water Conservation Order must be linked to the WCO's objective of sustaining the diverse values of Te Waikoropupu Springs. This means that any water quality guidelines or limits for the springs and associated aquifers should be based on historical and present water quality of the springs and of the contributing aquifers, tempered by any additional relevant scientific information. The 99% protection level concentrations provided within the ANZECC guidelines (see ANZECC & ARMCANZ 2000; Table 3.4.1) could provide interim default guidelines concentrations for groundwater in the absence of any other relevant information, but the numerous limitations of these guidelines identified within that report, and their uncertain applicability to groundwater ecosystems, must be considered.

The following discusses preliminary guidelines concentrations of four key attributes for the WaiSAC water, including some not included in the ANZEEC guidelines.

Organic carbon (no ANZEEC guideline concentration)

Organic carbon, as the primary food source for most groundwater organisms, varies seasonally and generally determines groundwater community composition and abundance (e.g., Baker 2000, Sinton 1984, Fenwick et al. 2004, Datry et al. 2005, Hancock & Boulton 2008). In dissolved or very fine particle forms (including in bacteria cells), it may be carried into the aquifer with inflowing water in the upper catchment or at any point along a catchment (Baker 2000; Jones 1995; Scarsbrook 2003). Most importantly, organic carbon also enters groundwater from overlying land use activities where it is incorporated into biofilms (Fenwick et al. 2004, Boulton et al. 2008, Hartland et al. 2011).

Biological activity in groundwater ecosystems is frequently limited by organic carbon availability (Baker 2000; Jones 1995). Many stygobitic[13] taxa are adapted to living in aquifers where food is scarce, with their metabolic (and reproductive) rates and oxygen requirements generally appreciably lower than equivalent epigean[14] or stygophilic[15] species (e.g., Spicer, 1998; Wilhelm et al. 2006).

[13] Stygobite or stygobitic species: obligate or strictly subterranean aquatic inhabitants for the entire lives. Taxa: generally used here to mean a species, at times means some other taxonomic unit or grouping of organisms.
[14] Epigean: inhabiting surface waters.
[15] Stygophilic species or stygophile: inhabit both surface and subterranean aquatic environments, not constrained to either.

Increased organic carbon and food availability potentially cancels the competitive advantages of this stygobitic physiological adaptation, enabling stygophilic species with higher metabolic rates (and faster generation times) to displace the natural stygobitic community (assuming dissolved oxygen is not limiting)(Wilhelm et al. 2006). Thus, while increased organic carbon supply increases abundances of some species, it may lead to other strictly stygobitic species and communities being displaced or even eliminated through competition by non-obligate stygophilic species, especially if other environmental factors (e.g., dissolved oxygen) change to suit the stygophiles (Datry et al. 2005). Such a shift in community composition occurred within a large coastal aquifer contaminated by treated wastewater (increased nitrate, biochemical oxidation demand, dissolved organic carbon) over 45 years, with one omnivorous species becoming the dominant, displacing others (including apparent extinction of one endemic stygobitic species)(Marsciopinto et al. 2006).

Community density increased with organic carbon enrichment in a New Zealand alluvial aquifer some 5 km from the nearest surface waters (Sinton 1984), but taxonomic resolution was insufficient to observe any associated changes in species richness. That study did report repeated significant kills of stygofauna at the most contaminated wells, apparently due to excessive organic carbon from effluent leading to anoxic conditions (Sinton 1984).

Stygofauna within karst cave systems appears similarly affected by organic carbon enrichment. For example, massive organic enrichment resulting from dumping sawdust into a cave exterminated the previously abundant and diverse stygofauna, biofilms >1 cm thick coated the gravel substrate, and huge populations of opportunistic species (tubificid worms and chironomid flies) developed (Culver 1992). Similar shifts in community composition in response to organic carbon enrichment are noted for several other SGDEs (e.g., Illife 1984).

Organic carbon concentrations tend to be higher closer to upper catchment recharge areas, than lower in the catchment (see Table 1) and some decrease in concentration with increasing depth in the aquifer seems likely. Some organic carbon hotspots associated with buried ancient wood or other organic material seem likely at any depth within many alluvial aquifers. Organic carbon concentrations also may vary over quite small distances and quite short time spans within alluvial aquifers. For example, dissolved organic carbon (DOC) varied from 1.5 to 24 mg/L seasonally over an eight-month period in an Austrian aquifer (2-6 m depth) (Gunatilaka 1994).

Some of the variability reported in these studies may be natural, and some does result from human activities. To date, there is no clear understanding of organic carbon concentrations or its natural variation in groundwaters completely unaffected by human activities. Organic carbon was optically undetectable in water emerging from the springs when measured twice (February 1993 and March 1995) (Davies-Colley & Smith 1995). There are no data on organic carbon concentrations elsewhere within the catchment and aquifer, but it must be generally very low for the organic carbon to be entirely consumed during the water's transit to the springs.

Table 1 lists reported organic carbon concentrations available from New Zealand research on aquifers. These values come from diverse measurements, some gathered over several years, others a single measurements. Perhaps the most relevant values are those from groundwater adjacent to the upper reaches of the Selwyn River, at a point where the river leaves less intensively farmed foothills to disappear into the aquifer that flows seaward under more intensively farmed plains. In our experience, organic carbon concentrations of up to 3-4 mg/L, in combination with moderate concentrations of dissolved oxygen (e.g., >4 mg/L) are associated with apparently healthy, functional alluvial groundwater ecosystems.

Clearly, organic carbon concentrations within the WaiSAC require urgent measurement to provide a meaningful background for guiding management of water quality at the springs. On-going monitoring, especially to determine any seasonal variations and changes to these, are essential. This measurement and monitoring is likely to find different organic carbon concentrations at different points within each of the contributing aquifers and at different seasons at each point. Thus, it is inappropriate to suggest any initial guideline or limit concentrations for this variable, other than for at the springs water itself, where concentrations of dissolved organic carbon must remain undetectable to maintain the water's extreme clarity.

Table 1: Concentrations of organic carbon reported for New Zealand aquifers.

Organic carbon (mg/L)	Location	Contamination	Aquifer details	Source
undetectable	Te Waikoropupu Springs	Low; probably uncontaminated	Karst & alluvial (3 aquifers contribute)	Davies-Colley & Smith 1995
1.1-3.4	Templeton, Canterbury	Moderate	Alluvial aquifer c. 18 m to water table; control well	Fenwick & Wilson 1999
1.5-5.6	Templeton, Canterbury	Highly	Alluvial aquifer c. 18 m to water table; wastewater	Fenwick & Wilson 1999
8.1	Leeston, Canterbury	Moderate	Fine-grained alluvial aquifer, contaminated	Hartland et al. 2011
9.0-18.2	Leeston, Canterbury	Highly	Fine-grained alluvial aquifer, wastewater contaminated	Hartland et al. 2011
1.2 (n=4)	Selwyn River, Canterbury	Low (headwaters)	Alluvial aquifer riverine recharge zone	Williamson et al. 2012
0.7 (n=4)	Selwyn River, Canterbury	Moderate (lower reach)	Alluvial aquifer close to lowland river	Williamson et al. 2012
0.4 (n=4)	Lincoln, Canterbury	Moderate	Alluvial aquifer	Williamson et al. 2012
0.6-3.4 (n>20)	Selwyn River, Canterbury	Low (headwaters)	Alluvial aquifer riverine recharge zone	Larned et al. 2014
0.4-2.1 (n>20)	Selwyn River, Canterbury	Moderate (lower reach)	Alluvial aquifer close to lowland river	Larned et al. 2014

Dissolved oxygen (no ANZEEC guideline concentration)

Normally, unpolluted, gravel-bed stream water is close to 100% saturated with oxygen (i.e., c. 10 mg/L, depending on temperature (Davies-Colley & Wilcock 2004)), although natural processes and human impacts can deplete oxygen, especially where higher temperatures and/or organic carbon enrichment increase chemical and biological demand for oxygen beyond its replenishment rate. In aquifers, water flowing through the aquifer matrix often has minimal or no oxygenation from contact with air for long periods (weeks, months, years, decades). Consequently, alluvial aquifer waters tend to contain less oxygen with increasing distance from their recharge zones and typically are 5-45% saturated (e.g., Danielopol et al. 2001; Hancock et al. 2005).

Oxygen is fundamental to aerobic organisms, including most stygofaunal invertebrates and especially crustaceans (Malard & Hervant 1999), and its availability can be the dominant, direct effect on stygofunal community composition and abundance (Mosslacher, Pospisil et al. 1996). Aerobic organisms take up and use oxygen for respiration, even at rest, although taxa differ in their oxygen consumption rates and ability to withstand reductions in dissolved oxygen availability. True stygobitic species consume less oxygen than their stygophilic and epigean counterparts (Spicer 1998; Mosslacher 2000; Wilheim, Taylor et al. 2006), frequently enabling survival at the lower (<3 mg/L) dissolved oxygen concentrations common in subterranean interstitial habitats (Malard & Hervant 1999). Under such hypoxic conditions (oxygen concentrations typically < 2-3 mg/L), some stygobites switch to anaerobic metabolism to fuel their energy needs (Hervant et al. 1996), although there is no clear evidence that any normally aerobic stygobitic species survives anoxia indefinitely. Others, such as some hyporheic amphipods, actively move towards and into higher dissolved oxygen concentrations, independent of flow direction (Henry and Danielopol 1999).

Any food or organic carbon enrichment that stimulates microbial activity may use much or all of the available dissolved oxygen (e.g., Baker et al. 2000). Stygofaunal communities increase in density in response to increased food (i.e., organic carbon) only if there is sufficient dissolved oxygen (Mösslacher & Notenboom 1999). Enrichment and bacterial stimulation without sufficient dissolved oxygen (perhaps due to reduced water flows or increased temperature) can lead to anoxia that kills much of the stygofauna (Sinton 1984; Boulton et al. 2008).

Field evidence of the effect of dissolved oxygen on community compositions and species abundances are generally confounded by other interacting environmental variables. For example, the stygofauna inhabiting wells generally closer to a river differed from that at more distant wells where the aquifer was shallower, contained less dissolved oxygen and transmissivity was lower (Dumas et al. 2001).

As with organic carbon, therefore, setting any limits for dissolved oxygen concentrations is complicated and requires a substantial body of research information, much of this specific to the WaiSAC. A cursory survey of readily available information on dissolved oxygen in New Zealand alluvial aquifers (Table 2) provides little guidance, except that Te Waikoropupu's water contains c. 6.5 mg/L of oxygen (at least in 1976). The only appropriate guideline is that the WaiSAC should be managed to ensure that water discharging from the springs contains at least 6.0 mg/L of dissolved oxygen. It is inappropriate to suggest any guideline levels for dissolved oxygen concentrations elsewhere within the WaiSAC in the absence of specific information on current dissolved oxygen concentrations within different parts of the aquifer and their relationships to spring water.

We note that dissolved oxygen is replenished primarily via recharge water and that as recharge declines, so too do water levels (depths below ground)(i.e., hydraulic head decreases) and dissolved oxygen concentrations. In particular, dissolved oxygen appears to become a critical factor at low aquifer levels when the hydraulic gradient is reduced and the rate of water replacement (containing more dissolved oxygen) is slowed. Thus, managing, water levels to ensure near natural velocities/flows through the aquifer matrix, in tandem with managing organic carbon concentrations within groundwater, seems likely to sustain higher dissolved oxygen concentrations within most aquifers.

Table 2: Dissolved oxygen concentrations for various New Zealand aquifers.

Dissolved oxygen mg/L	Location	Contamination	Aquifer details	Source
6.5	Te Waikoropupu Springs	None; natural?	Discharge from Arthur Marble Aquifer of	Michaelis 1976
6.4-8.1	Templeton, Canterbury	Moderate	Alluvial aquifer c. 18 m to water table; control well	Fenwick & Wilson 1999; Scarsbrook & Fenwick 2003
3.7-8.4	Templeton, Canterbury	Highly	Alluvial aquifer c. 18 m to water table; wastewater	Fenwick & Wilson 1999; Scarsbrook & Fenwick 2003
6.3-9.6	Waimakariri R, Canterbury	Low	Alluvial aquifer riverine recharge zone	Scarsbrook & Fenwick 2003
3.3-8.5	Hawkes Bay: Ngaruroro R.	Moderate	Riverine alluvial aquifer	Scarsbrook & Fenwick 2003
6.0-7.8	Hawkes Bay: Waipaua R.	Moderate	Riverine alluvial aquifer	Scarsbrook & Fenwick 2003
2.1–8.6 (n=3)	Selwyn River, Canterbury	Low (headwaters)	Alluvial aquifer riverine recharge zone	Williamson et al. 2012
1.52–4.73 (n=3)	Selwyn River, Canterbury	Moderate (lower reach)	Alluvial aquifer close to lowland river	Williamson et al. 2012
7.4 (n=4)	Lincoln, Canterbury	Moderate	Alluvial aquifer	Williamson et al. 2012
0.3-7.1	Selwyn River, Canterbury	Low (headwaters)	Alluvial aquifer riverine recharge zone	Larned et al. 2014
0.7-7.2	Selwyn River, Canterbury	Moderate (lower reach)	Alluvial aquifer close to lowland river	Larned et al. 2014

Nitrate

The nitrate[16] ion (NO_3^-) occurs naturally in the environment along with ammonium (NH_4^+) and nitrite (NO_2^-) in ionic form as the most common inorganic forms of nitrogen. Ammonium is usually converted (oxidised) to nitrite and nitrate by common aerobic bacteria when oxygen is present, even

[16] It is the concentration of nitrate ions (NO_3^-) that determines toxicity. However, toxic concentrations frequently are reported in terms of nitrate-nitrogen (NO_3-N), which can be converted to nitrate ion equivalent by multiplying by 4.43 (and the converse by multiplying by 0.23 to derive mg NO_3-/L)(after Hickey 2013: 8). Here, we follow the common approach of reporting toxicities as mg NO_3-N /L, but the difference in reporting unit makes no difference to toxicity (Hickey 2013: 8).

at low (1 mg/L) oxygen concentrations, so that nitrate predominates in aerobic aquatic environments (e.g., Camargo et al. 2005). Nitrate is removed from aquatic environments when taken up as an essential nutrient by plants or converted to nitrogen gas (N_2) by bacteria in anaerobic situations (and at anaerobic micro-sites within more generally aerobic environments). However, substantial additional nitrate enters many surface and groundwaters from human sources (e.g., agricultural runoff, municipal and industrial wastewaters, urban runoff), frequently increasing total dissolved nitrate concentrations substantially (e.g., Tidswell et al. 2012).

The primary concern over nitrate in the environment is due to its toxicity to humans, farm and domestic stock, and to aquatic invertebrates. In all cases, nitrate binds to the oxygen-carrying blood pigments (haemoglobin in humans and mammals, haemocyanin in many invertebrates), preventing these pigments from transporting oxygen to body tissues (Camargo et al. 2005). Nitrates also are implicated as potential carcinogens for humans, adding to concern about drinking nitrate contaminated water. Thus, nitrate is a high priority for resource management, especially for managing freshwaters.

Although there are few useful data on nitrate toxicities for groundwater invertebrates, equivalent information for surface water faunas provide useful guidelines. Nitrate increases in toxicity to aquatic animals with increasing concentrations and with exposure times, and may decrease with increasing body size, water salinity, and environmental adaptation (Camargo et al. 2005). Based several experiments and other results, a maximum nitrate (as nitrogen) concentration of 2.0 mg NO_3-N/L (or 8.86 mg NO_3^-/L) was recommended to protect sensitive surface water species during longer-term exposures (Kincheloe et al. 1979; Camargo et al. 2005).

The effects of nitrate on groundwater biofilms and stygofauna in situ are less clear. Amphipod crustaceans appear to be among the more sensitive of invertebrates and are especially relevant here because they dominate many groundwater communities.

In a detailed, expert review of all available data on nitrate toxicology for freshwaters and using the ANZECC (2000) and Environment Canada's methodology, Hickey & Martin (2009) recommended specific NO_3-N concentrations for high conservation/ecological value surface water ecosystems, slightly to moderately disturbed systems and for highly disturbed systems for Canterbury's freshwater environments. They noted, however, that the "datasets are particularly lacking in species which are known to be of high sensitivity to contaminants", especially "amphipods, mayflies and some native fish species that are more sensitive to some chemical contaminants than the standard international test species" (Hickey & Martin 2009: 19). A subsequent update of that review for New Zealand lakes and rivers (not groundwaters) included several new acute and chronic data (including for a native mayfly and juveniles of an endemic fish), partially addressing the earlier information gaps (Hickey 2013). It recommended average long-term exposure concentrations of 1.0 mg NO_3-N/L to protect high conservation value ecosystems (concentrations at which no effect was observed; termed Grading) and threshold effect (termed Surveillance) concentrations of 1.5 mg NO_3-N/L for managing seasonal (up to three months) maximum concentrations (Table 3).

Table 3: Guideline concentrations for nitrate (reported as NO_3-N concentrations) to protect surface water species. Grading guidelines are based on species' no observed effect concentrations, and Surveillance guidelines based on threshold effect concentrations. From Hickey (2013): 16 (Table 5.1).

Guideline Type	Grading Nitrate concentration (mg NO_3-N /L)	Surveillance Nitrate concentration (mg NO_3-N /L)	Description of Management Class
Chronic – high conservation value systems (99% protection)	1.0	1.5	Pristine environment with high biodiversity and conservation values.
Chronic – slightly to moderately disturbed systems (95% protection)	**2.4**	**3.5**	**Environments which are subject to a range of disturbances from human activities, but with minor effects.**
Chronic – highly disturbed systems (90% protection)	3.8	5.6	Environments which have naturally seasonally elevated concentrations for significant periods of the year (1-3 months).
Chronic – highly disturbed systems (80% protection)	6.9	9.8	Environment which are measurably degraded and which have seasonally elevated concentrations for significant periods of the year (1-3 months).
Acute	20	30	Environments which are significantly degraded. Probable chronic effects on multiple species.
Method of comparison	Annual median	Annual 95^{th} percentile	

This is the best available compilation of relevant toxicity data for freshwater and groundwater organisms. However, it noted continuing significant knowledge gaps in:"(i) the adequacy of native fish and invertebrate [nitrate toxicity] data for surface waters; (ii) absence of [data on] hyporheic species; and (iii) [nitrate] toxicity modification in relation to water mineral content (measured by hardness)" (Hickey 2013: 25). Hickey (2013) also noted the need for field validation of these results and the potential ameliorating effects of water hardness and chloride ion concentrations. Further important information gaps are (i) the sensitivities of stygobitic fauna and biofilms to nitrate, (ii) how these sensitivities change with other human-induced stresses, especially dissolved oxygen, and (iii) nitrate concentrations for sublethal effects that interfere with biodiversity and ecosystem functioning are poorly understood, particularly for stygobites.

Recent reports indicate concentrations of NO_3-N[17] mostly within 0.0-2.0 mg/L closer to the springs, concentrations between 2.1 and 4.0 mg NO_3-N /L further up the catchment and values exceeding 4.1 mg NO_3-N /L at 3-4 monitoring points upstream of the springs (Stevens 2010). Water in the springs

[17] Stevens (2010) reported nitrate concentrations in units of mg/L-N. We assume that these units are mg NO_3-N/L.

was reported to contain 0.31-0.32 mg/L (0.31-0.32 g/m^3) of NO_3-N in 1976 (Michaelis 1976), with recent nitrate concentrations reported as "typically <0.4 mg/L-N" (median 0.36 mg/L; Stevens 2010: 31).

Based simply on Hickey's (2013) recommendations, his chronic-high conservation value of 1.0 mg NO_3-N/L could be regarded as an upper limit, as an interim measure. However, because present concentrations are less than half this value and historical data indicate significant increases since the 1970s, the aquifers and catchments should be managed to ensure that NO_3-N concentrations in spring water do not exceed 0.4 mg NO_3-N/L in order to protect the springs' high conservation values.

Ammonia

Under anaerobic conditions, nitrate is reduced to ammonium (NH_4^+), which persists in equilibrium with unionised ammonia (NH_3) (Close et al. 2001). Ammonia is an important and highly toxic contaminant, whereas ammonium (NH_4^+) is largely inert (Russo 1985; Prenter et al. 2004), however the two forms exist in a dynamic equilibrium influenced by temperature and pH (Emerson, Lund et al. 1975). At lowest water levels and/or with excessive organic carbon loadings when dissolved oxygen concentrations are very low (i.e., hypoxic conditions) and especially at higher pH (>9.2) and temperature, ammonia concentrations in groundwater can threaten groundwater ecosystems.

Ammonia (NH_3) is toxic to freshwater invertebrates at low concentrations. For example, 50% of individuals of three freshwater amphipod species died after exposure to 0.36, 1.16 and 1.54 mg NH_3/L, with sublethal effects (disruption of mating) occurring at concentrations as low as 0.12 and 1.23 mg/L (Prenter et al. 2004). Another investigation of amphipods reported that 50% of individuals died after 96 h exposed to 0.71 mg NH_3/L and after 21 hours for a concentration of 6 mg NH_3/L (McCahon, Poulton et al. 1991), comparable to 50% mortality after 27 h exposure to 3 mg NH_3/L from another study (Williams, Green et al. 1986).

Ammonia concentrations reported for Te Waikoropupu Springs (as NH_3-N) were 0.00026 mg/L in the 1970s (estimated from Michaelis's (1976) 0.04 mg/L NH_4-N using an on-line calculator) and more recently reported to be 0.0-0.05 mg/L, with higher concentrations in nearby groundwater (Stevens 2010). These values and available information on toxicities of ammonia indicate that WaiSAC water should be managed to maintain ammonia concentrations below 0.05 mg/L and perhaps substantially lower. Certainly, the ANZECC trigger value of 0.32 mg/L NH_3 for protecting 99% of species seems inappropriate for Te Waikoropupu springs water and the WaiSAC generally.

Conclusions

The guideline concentrations for the four substances discussed here must be regarded as tentative because they are based on a review of a very small body of empirical information. A more rigorous and comprehensive approach is highly desirable, but there is scant information on toxicities, tolerances and sublethal effects for groundwater ecosystems, including biofilms, and specifically for New Zealand or WaiSAC stygofauna. For these reasons, refining these suggested limits will require significant time and other resources.

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