BEFORE THE INDEPENDENT COMMISSIONERS

Under the Resource Management Act (1991)

In the matter of Plan Change 5 to the Canterbury Land and Water Regional Plan: Nutrient Management and Waitaki Sub-region

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EVIDENCE IN CHIEF OF ADAM DOUGLAS CANNING ON BEHALF OF LOWER WAITAKI RIVER MANAGEMENT SOCIETY
19th of July 2016

QUALIFICATIONS AND EXPERIENCE

1. My full name is Adam Douglas Canning.

2. I am a Doctoral Researcher in Freshwater Ecology in the Institute of Agriculture and Environment – Ecology at Massey University. I have a Bachelor of Science with Honours – First class (Biological Sciences and Environmental Science).

3. I am a member of the Ecological Society of America, The International Association for Ecology (INTECOL), and the New Zealand Freshwater Sciences Society, the International Society for Ecological Modelling, the Australasian Society for Fish Biology, and the Society for Ecological Restoration. I have presented research at conferences held in both New Zealand and the USA

4. My research is focussed on understanding community and ecosystem thresholds to ensure ecosystem health (life supporting capacity) of freshwater and estuarine systems in New Zealand. I am very familiar with literature relating to ecological community stability, environmental thresholds, modelling thresholds, and nutrient and environmental determinants of New Zealand freshwater ecosystem health.

5. In preparing this evidence I have reviewed:

   a. Proposed Plan Change 5 to the Canterbury Land and Water Regional Plan;
   b. National Policy Statement for Freshwater Management 2014 (NPS-FM);
   c. State of Environment monitoring water quality (both physicochemical and biota),and hydrological data for the Canterbury Region as provided by the Canterbury Regional Council;
   d. Canterbury Regional Council's technical report: Upper Waitaki catchment flows, water quality and ecology: state and trend, report number R15/57, by Duncan Gray, published May 2015.

e. Canterbury Regional Council's technical report: Lower Waitaki water quality and ecology: state and trend, report number R15/111, by Graeme Clarke and Michael Greer, published December 2015.

f. Canterbury Regional Council's technical report: The current water quality state of lakes in the Waitaki catchment, report number R15/157, by Graeme Clarke, published December 2015.

g. All of the literature cited and referenced under "References."

h. Canterbury Regional Council section 32 report for Plan Change 5 (Nutrient Management and Waitaki Sub-region) to the Canterbury Land and Water Plan.

6. I have read the Code of Conduct for Expert Witnesses in the Environment Court Practice Note. This evidence has been prepared in accordance with it and I agree to comply with it. I have not omitted to consider material facts known to me that might alter or detract from the opinions expressed.

SCOPE OF EVIDENCE

7. I have been asked by the Lower Waitaki River Management Society to prepare evidence on whether the Proposed Plan Change 5 to the Canterbury Land and Water Regional Plan will meet the ecological objectives of the National Policy Statement for Freshwater Management (2014), the Resource Management Act (1991), and the National Coastal Policy Statement (2010).

8. This includes:

a. A discussion on the causes of ecosystem degradation;

b. A review of proposed plan change 5 and in particular the proposed Freshwater outcomes/objectives, targets and limits;

c. And to recommend appropriate limits/outcomes to achieve the objectives outlined by the policy above (mainly the NPS-FM) in the area covered by the Propose Plan Change 5 to Canterbury Land and Water Plan.

9. I do not have any competing interests nor am I financially supported by any submitting party in providing this evidence.

EXECUTIVE SUMMARY

9. The principal driving factors of decreased freshwater ecosystem health include increased nutrient levels, loss of riparian habitats, altered and reduced flows, and increased suspended and deposited sediment.

10. The proposed objectives in Plan Change 5 to the Canterbury Land and Water Plan will not achieve the objectives outlined in the NPS-FM. The QMCI values listed in the freshwater outcomes are suitable, however the nutrient outcomes will not achieve the QMCI outcomes.

11. Instream habitat quality (i.e., natural character), water quantity, nutrients (nitrogen and phosphorus), suspended and deposited sediment and riparian margins all need to be managed appropriately to achieve sustainable ecosystem health. All of these factors interact together to determine ecosystem health thus all need to be managed.

12. There has been insufficient investigation done to determine the effects of the proposed freshwater outcomes on the coastal environment. Nor has the impact of climate change on ecosystem health been evaluated thoroughly and managed for. My evidence has largely focussed on safeguarding the life-supporting, natural character, indigenous species and ecosystem processes of riverine communities, however safeguarding riverine communities (by communities I refer to the ecological communities not the human ones) does not necessarily safeguard coastal communities and further analysis and management plans are required.

13. In many areas, the Proposed Plan Change 5 has many sites where the Dissolved Inorganic Nitrogen are set at levels approximately two to ten times too high to safeguard life-supporting capacity, indigenous species and ecosystem processes. Native fish also needed to managed for as they form an integral component of the ecological community.

14. Riparian buffer zones provide a range of benefits to freshwater communities including the reduction of in-stream nutrient concentrations, lowered fine suspended and deposited sediment, the exclusion of livestock, temperature control, flow variability control, maintaining natural habitat character, and providing a source of food for aquatic taxa.

THE CONCEPT OF ECOSYSTEM HEALTH

15. The concept of ‘ecosystem health’ is used widely in policy and science for the management of freshwaters (Costanza and others 1992; Reynoldson and Metcalfe-Smith 1992; Scrimgeour and Wicklum 1996; Rapport and others 1998; Boulton 1999; Norris and Thoms 1999). The European Union Water Framework Directive for example seeks to attain “good ecological status” in freshwater bodies (European Commission, 2000). In New Zealand, the National Policy Statement for Freshwater 2014 recognises the importance of values relating to “safeguarding the life-supporting capacity of water and associated ecosystems” which include the value of “healthy ecosystem processes functioning naturally.”

16. Ecosystem health represents the state in which an ecosystem has the “ability to maintain its structure (organization) and function (vigor) over time in the face of external stress” (stability) (Costanza and Mageau 1999). In this context, ‘vigour’ can be related to the ability of an ecosystem to sustain life. In freshwaters, this may, for example, become impaired by the presence of a toxic pollutant, or enrichment through nutrient pollution. ‘Organisation’ relates to the extent of integration between ecosystem components. In freshwaters, this may, for example, become impaired by the extirpation of native species due to a change in habitat quality. Stability refers to the maintenance of community structure and function over time in the face of disturbance and is an important feature of a ‘healthy’ ecosystem. Ecological communities typically persist in state where the needs for stability against disturbance is traded off against highly efficient, specialist-dominated structures (some term this a “the window of vitality”) (Ulanowicz 2009; Ulanowicz and others 2009). If a community is highly frequently or recently perturbed then it is likely to develop a highly redundant structure that confers stability. However, if the system is too redundant then it will lack the efficiency needed to adequately function and community growth will be stunted. On the flip side, if a community is composed of many species with narrow but highly efficient diets then although the community will grow efficiently, it will lack the adaptive capacity needed to be stable in the face of disturbance. Therefore to persist, a balance between stability and efficiency is needed (Ulanowicz 2009; Ulanowicz and others 2009).

17. Although the ecological health of rivers and streams is determined by a wide

range of potentially interacting stressors, it is clear that nutrients are one of the most pervasive and detrimental stressors for the fauna and flora of rivers globally (Carpenter and others 1998; Allan 2004; Stevenson and Sabater 2010). Communities with low nutrient availability tend to conserve nutrients by being structured in a way that yields greater levels of cycling. Whereas highly enriched communities tend to cycle nutrients inefficiently (DeAngelis and others 1989; DeAngelis 2012). Highly enriched systems also tend to grow with little adaptive capacity because resources are not limiting (Ulanowicz 1997). High levels of cycling, driven by low nutrient availability, is likely to increase the food web resilience (the rate at which a community returns to a stable state following disturbance) (DeAngelis and others 1989; DeAngelis 2012). Low levels of enrichment also increase the number of internal flows, which can increase the robustness (the ability of a food web to maintain topological structure following disturbance) of the food webs. Cycling is more prevalent in food webs with many internal flows (Christensen and Pauly 1993; Ulanowicz 1997). Both resilience and robustness are essential components of a food web's stability and are both reduced when a community is enriched. Both cycling and internal flows (as opposed to flows of energy from outside the system) also increase exergy transfer between species and consequently the total system throughput. It is this propensity for growth in exergy transfer that permits greater capacity for resilience (Saint-Béat and others 2015). Therefore, low levels of nutrient enrichment are required to safeguard ecosystem processes.

LOTIC BIOLOGICAL COMMUNITIES

18. Within the flowing water ecosystems there is Periphyton, Detritus, Terrestrial Plant and Animal matter, Aquatic Invertebrates, and Fish. Periphyton (the coating of slightly furry green or brown algae on rocks) and detritus (both in-stream and terrestrial derived plant matter, e.g., leaves) form the basis of the stream food web. Some periphyton is required as food for many aquatic invertebrates; however, too much algal growth can dramatically change the ecology and habitat conditions of a river. Aquatic invertebrates consume the periphyton and plant matter either directly (along with other organic sources) or by predating the smaller grazing invertebrates. Native and sport fish eat these invertebrates and some terrestrial inputs. All of the biological components of a

river food web require the correct habitat and water quality conditions in order to maintain healthy populations and functioning ecosystems.

MANAGING THE FRESHWATER ECOLOGICAL COMMUNITES TO SAFEGUARD LIFE-SUPPORTING CAPACITY, ECOSYSTEM PROCESSES AND INDIGENOUS SPECIES IN THE WAITAKI CATCHMENT RIVERS

19. For freshwater communities to be stable in the long term, their constituents (the various species) must exist in the right balance and have a suitable environment to allow this balance to be sustained. A change in a single constituent can alter the entire community composition as a result of trophic cascades and resource competition.

20. Various indices of community structure have been developed as biological measures of life-supporting capacity and ecosystem processes, such as the QMCI (Quantitative Macroinvertebrate Community Index). Freshwater communities are largely a product of their environment, that is, for species to persist then environmental conditions must be within their tolerance zones. As freshwater organisms are always present in the water they are sensitive to environmental disturbances that may otherwise go un-noticed if we relied simply on physicochemical spot samples (Fig. 1.). I support the proposition of having the QMCI as a biological measure of ecosystem health and consider the proposed values for QMCI in proposed Table 15(a) of proposed plan change 5, as appropriate outcomes to be achieved by 2030. Given that several sites have ammoniacal-nitrogen levels that exceed protection thresholds In addition to the QMCI, I suggest that both the percentages of EPT (Ephemeroptera, Plecoptera and Tricoptera) taxa and abundance also be used as they have been shown to be sensitive to changes in metal and ammonia concentrations that have gone undetected by the QMCI (Winterbourn and McDiffett 1996; Collier and others 1998; Hickey and Clements 1998; Hickey and Martin 1999; Hickey and Golding 2002; Clements and others 2008; Hogsden and Harding 2011). EPT indices are very easy to calculate and would not cost any more to compute once all the data is collected for QMCI analysis.

21. To safeguard life-supporting capacity, ecosystem process and indigenous species, then we need measures for these various aspects. Whilst the

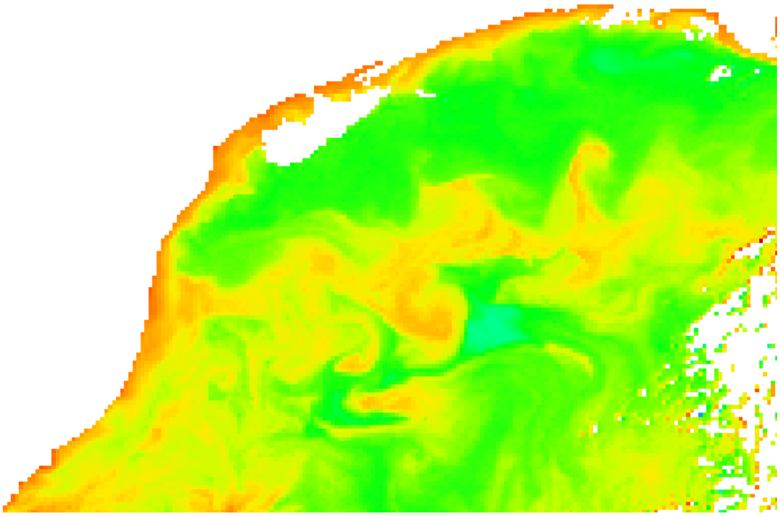
proposed plan includes measures of periphyton and macroinvertebrates, it misses the native fish – a large and integral component of the ecological community. The New Zealand Freshwater Fish Index of Biotic Integrity (IBI) provides a suitable measure of fish community health (Joy and Death 2004). The fish IBI provides an index of the fish actually present at a site relative to what should be there under ideal conditions. The IBI provides a value between 0 and 60, with 0 indicating no fish community at all (when there should be one) and 60 represents an extremely healthy fish community with all species expected being present. Fish-IBIs and similar indices based on predictive models are common place in most developed countries around the world. To safeguard fish life capacity, the local indigenous fish species, and protect the ecosystem processes that a healthy fish community support, the IBI should be incorporated into table 15(a) with a minimum value of approximately 35 which represents the boundary between fair and good quality (Joy 2009a; Joy 2009b; Joy 2010). Fish require greater interstitial spaces than invertebrates and have different sensitivity thresholds than macroinvertebrates, therefore simply relying on the QMCI may not give an accurate depiction of the health of other trophic levels (Joy and Death 2004; Leathwick and others 2005; Jowett and Davey 2007). Nationally the IBI has been in steep decline, primarily driven by land use change, with 74% of native fish listed as threatened. If these trends continue then it is expected that by 2050 or potentially before then all native fish in New Zealand will be extinct. The Waitaki catchment also contains many native fish, including the local indigenous taxa such as Canterbury Galaxias (*Galaxias vulgaris*) and the Canterbury Mudfish (*Neochanna burrowsius*). The Canterbury Mudfish is the most threatened of all the NZ mudfish and is only found in and between the Ashley River and the Waitaki River.

22. The Trophic Level Index (TLI - a multiple variable indicator of a lakes trophic status) values set for all lakes, except Lake Wainono, seem suitable. The minimum TLI proposed for Lake Wainono, is less than aspirational and has the outcome set at a level considered to be very poor water quality. There would also be no noticeable difference in appearance with currently proposed level. Whilst Lake Wainono has rapidly become very unhealthy in recent years from anthropogenic pressures, "the lake is clinging to an ecological threshold or tipping point and that successful restoration to a state with stable and healthy

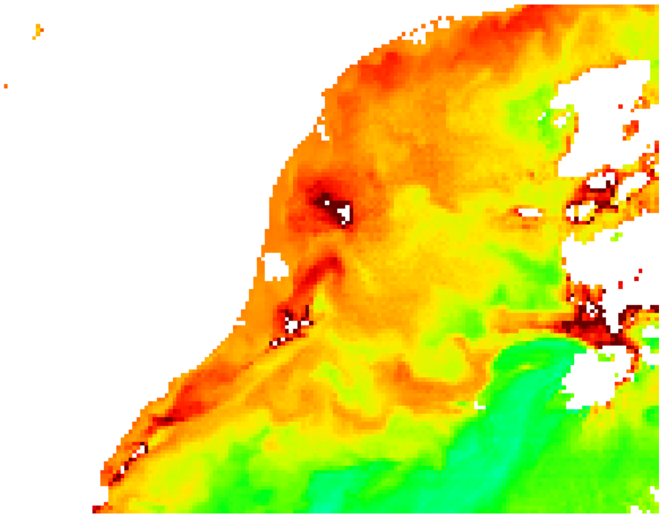
macrophyte beds, clearer waters and a diverse and productive aquatic food web could still be achievable," (Schallenberg and Saulnier-Talbot 2014). A TLI of at least 4 would be a better target for Lake Wainono.

23. The ecological impacts of the freshwater management plans on coastal and estuarine ecosystems do not appeared to have been thoroughly assessed. Coastal and estuarine ecosystems are often impacted by rivers in several ways: 1) As river flow changes, so do the outflow salinity gradients. Salinity gradients provide unique, highly diverse communities over a relatively short stretch. Changes in salinity gradients can affect the total habitat area of brackish communities and affect the productivity of the inhabiting species. 2) As rivers spread and merge with coastal waters they deposit sediment, which can bury benthic communities. The remaining suspended sediment can limit light in coastal systems and prevent photosynthesis by phytoplankton – the base of coastal communities. And 3) nutrients carried by rivers ultimately end up in coastal ecosystems, these can cause algae blooms that, like in rivers, can cause large fluctuations in oxygen that reduce life supporting capacity. There is currently little understanding about the effects of the Waitaki catchment outflow on the coastal community. However, using SeaDAS (Fu and others 1998) and MODID Aqua (Mu and others 2013) images of chlorophyll a concentration (from phytoplankton) we can get an insight into whether runoff is causing coastal blooms. Figures 1 (a-c) show high algae density off the coast of the Waitaki outflow which is then, eventually, carried towards the South Chatham Rise. In February 2014, there was an algae bloom seen by chlorophyll a as high as 52mg/m^3 of the coast of the the Waitaki region, this is in contrast to typical levels of around 1-2mg/m^3. In February 2016 there was also a widespread bloom around the Waitaki and Lower Canterbury region. The cause of these blooms are not yet known, however it is plausible, if not very likely, that these blooms have been exacerbated by high nitrate levels from the Waitaki catchment. Whilst the Waitaki River itself has low nitrate levels towards the outflow, the groundwater that runs parallel to the river that flows towards the coast has much higher than natural nitrate levels. Further investigation is urgently needed to understand the impacts of Waitaki catchment management on the coastal environment so that they can be managed as required by the NPS-FM.

a)



b)



c)

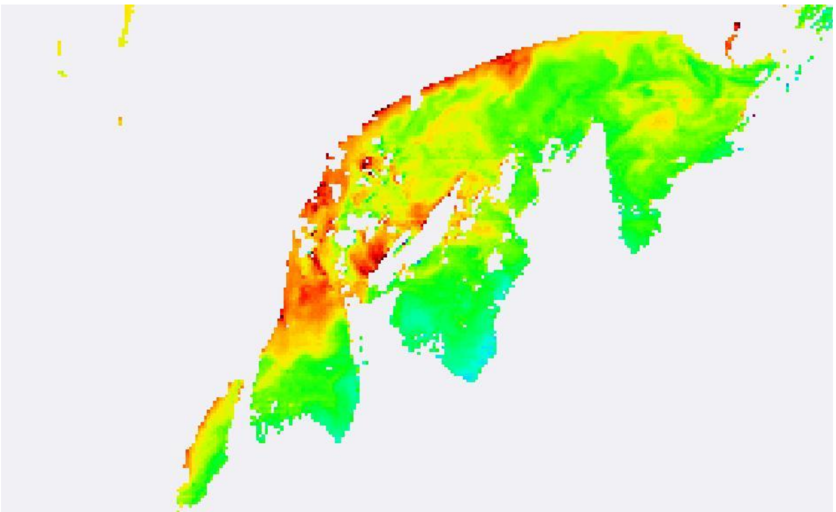


Figure 1. MODIS Aqua images of chlorophyll a concentrations off the coast of the Waitaki catchment in February 2013 (a), 2014 (b) and 2016 (c).

24. High turbidity and deposited sediment can drastically alter community composition. High turbidity can smother fish and invertebrates as well as make it difficult for them to find food (Lenat and others 1981; Ryan 1991; Rowe and Dean 1998). Furthermore, native freshwater fish have been shown to exhibit preference for waterways with low turbidity and avoid those with high suspended sediment (Boubée and others 1997). Fine deposited sediment reduces the space available for freshwater organisms to inhabit. The majority of New Zealand freshwater fish and organisms are benthic species and live in the interstitial spaces between substrate rocks (Jowett and Boustead 2001; Suren and Jowett 2001; Richardson and Jowett 2002; McEwan and Joy 2013; McEwan and Joy 2014b; 2014a). When high sediment loading occurs, the fine sediment becomes deposited and fills the spaces between the rocks. Thus leaving little room for fish and invertebrates to live and be protected, consequently reducing the life supporting capacity of the stream and driving out taxa from these areas and many to local extinction (Lenat and others 1981; Ryan 1991; Wood and Armitage 1997; Jowett and Boustead 2001; Suren and Jowett 2001; Richardson and Jowett 2002; Harrison and others 2007; Burdon and others 2013; Ramezani and others 2014).The proposed values for fine deposited sediment cover in table 15(a) are suitable outcomes and I support their retention (Burdon and others 2013).

25. Water temperature can also affect community composition as the freshwater organisms are ectotherms and their productivity rates will change with temperature. It has been found that 50% of Deleatidum mayflies (one of the most important dietary taxa for many native fish) will die after 4 days in water at 22.6°C (Quinn and others 1994). Furthermore, Quinn and Hickey (1990) found that as temperatures surpassed 19°C that distributions of Ephemeroptera and Plecoptera taxa were restricted, thus drastically altering the community composition. Whilst the native fish species each have different preferred thermal ranges (Richardson and others 1994), the Ephemeroptera and Plecoptera are important components of native fish diet and their absence could significantly reduce fish productivity and increase competition between native fish (Main and Winterbourn 1987; Jellyman 1989; Jellyman 1996; McDowall and others 1996; Hollows and others 2002; Rowe and others 2002; West and others 2005; Montori and others 2006). To safeguard life supporting capacity,

ecosystem process and indigenous species, I recommend that for all waterbodies that the maximum daily temperature in Table 15(a) be reduced from 20°C to 19°C. This reduction is very achievable by maintaining sufficient flows and having sufficient cover from riparian vegetation.

26. Excessive periphyton growths are not only aesthetically unappealing, but they can also result in dramatic changes to the biological communities in rivers and streams. They lead to a change from mayfly, stonefly and caddisfly dominated communities to ones with worms, snails and midges that do not support the same abundance, biomass or diversity of fish that the former communities do. The periphyton can also build up to such a biomass that the lower layers start to rot. This can dramatically reduce the oxygen levels and change the pH of the water leading to significant adverse effects on many invertebrates and fish. Whilst oxygen concentration may be very high during the day time from high rates of photosynthesis, at night the lack of light prevents oxygen from being released into the water and oxygen levels can plummet to lethal levels (Dean and Richardson 1999; Franklin 2013). A good example of this the Manawatu River, particularly a sampling site at Hopelands Road where continuous monitoring of dissolved oxygen revealed levels swinging between 40% and 140% over 24hrs in late summer (Clapcott and Young 2009). Thus many fish and invertebrate species are unable to survive, regardless of high oxygen concentrations that are recorded from daytime measurements, leading to differences in community composition. Surviving fish species would become stressed, susceptible to disease and develop poor condition as a result of undesirable dietary changes from alterations in macroinvertebrate community structure (Dean and Richardson 1999; Franklin 2013). It is for these reasons that I recommend dissolved oxygen sampling being conducted over at least a 24 hour period rather than spot sampling during the day time. The outcomes that are set in proposed table 15(a) should be upheld regardless of the time of day.

27. I do not deem the proposed dissolved oxygen limits in table 15(a) to be sufficient on their own, to sustain life supporting capacity. The proposed dissolved oxygen metric is measured in terms of percent saturation. The problem with this approach is that as water temperature increases, the amount of oxygen that

can be held by the water decreases. Thus at high temperatures, high levels of saturation are easily met with low levels of absolute dissolved oxygen concentration. This will be most problematic during Summer, when water temperature is high, flows are low, and periphyton growth is greatest (from typically low frequencies of scouring events). When oxygen concentration is low (even for a short period), the inhabiting species will be suffocated to death or suffer hypoxia, when oxygen saturation is high then fish and invertebrates get gas bubble disease (similar to the benz or decompression sickness experienced by scuba divers that ascend rapidly). I therefore suggest Table 15(a) be expanded to include a minimum absolute dissolved oxygen concentration (as used in the NPS-FM) and a maximum dissolved oxygen saturation no higher than 110%. These values will not only protect sensitive macroinvertebrates, they will also protect native fish species including inanga whitebait species, thereby safeguarding life-supporting capacity, ecosystem processes and indigenous species (Dean and Richardson 1999; Landman and others 2005; Franklin 2013).

28. The amplitude of oxygen concentration fluctuation is a factor of both periphyton and bacteria biomass. The greater the periphyton biomass, the more oxygen there is produced during the day and the more decomposing biomass there is causing greater oxygen reductions at night (Welch and others 1988; Welch and others 1992; Biggs 2000). I do not deem the proposed outcomes on chlorophyll a and filamentous algae (measures of periphyton biomass) to be sufficiently low enough to prevent lethal or stressful levels of dissolved oxygen concentration. High periphyton biomass can then reduce both macroinvertebrate and fish community health (as measured by the QMCI and IBI). The values suggested for the hill-fed lower rivers are in line with those associated with eutrophic streams, high filamentous algae proliferation and will not achieve the planned QMCI of 6 (Biggs and Price 1987; Welch and others 1988; Welch and others 1992). Also having no limit set for the spring-fed plains will mean the national bottom line of 200mg/m^2 is used which is also far too high to safeguard life-supporting capacity, ecosystem processes and indigenous species; furthermore, the national bottom line will not allow the planned QMCI of 5 to be achieved. Therefore, the current planned levels of periphyton is likely to result in a plan that is inconsistent and unachievable. I

instead propose the chlorophyll a outcome (in table 15(a)) for hill-fed lower values be reduced to 50mg/m2 and the spring-fed plains be set at 120mg/m2. These proposed values are in line with those required to allow the QMCI outcomes to be met will prevent large proliferations from occurring and causing stressful oxygen depletion. The proposed macrophyte outcome in table 15(a) seem sensible (Matheson and others 2012; Snelder and others 2013; Matheson and others 2016).

29. The proposed cyanobacteria levels in table 15(a) for the hill-fed lower and the spring-fed plains are extremely high. If met, or even half met, would require the public health unit to be notified, warning signs erected, and weekly sampling (Wood and others 2009). Reducing cyanobacteria at all sites to 20% cover is recommended.

30. Periphyton biomass is largely kept in check by the abundance of available resources, the amount of predation occurring, temperature, and the size and frequency of floods. Resources that limit periphyton growth are almost always shade and nutrients.

31. Maintaining the key elements of the hydrological regime of rivers and streams is also vitally important for protecting the ecological health of freshwater environments along with their geomorphology (physical form and structure). Decreased flow can mean more sediment being deposited, greater nutrient concentrations, less wetted habitat, greater temperatures, and more periphyton biomass. See the review by Dewson, James and Death (2007) for a comprehensive review of the ecological consequences of reducing flow. High flow events are important for scouring periphyton and keeping the standing stock low. Manipulations that cause lower flow variability than otherwise natural mean that periphyton may grow excessively and reduce ecosystem health. Flow variability also allows for runs, riffles and pools to all occur rather than a homogenous stretch, thus supports greater habitat for biota (Jowett and Duncan 1990; Biggs and others 2005). I support all of the points within 15.4.29 as they do not allow flow reductions that disturb increase periphyton, sedimentation or alter the natural character. I have not had time to evaluate the analysis and use of the values in Tables 15(g) to 15(i), though I recommend these values are set in line with minimum flows required to address the points made in 15.4.29(b).

Riparian Strips

32. Riparian buffer zones can range from a simple strip of vegetation from which livestock or other agricultural activities are excluded to a completely vegetated native forest riparian strip. The principal effect of the riparian buffer is to act as a barrier to nutrients, sediment, pathogens and other potential contaminants running off the land and to prevent it entering the waterway and consequently flowing downstream to lakes and estuaries. It will also stabilise stream banks and limit erosion and undercutting. The vegetation can also take up some of the nutrients. If a forested riparian zone exists this can also serve to limit light reaching the stream bed (which can also exacerbate periphyton growth) and water temperature (most aquatic animals have an upper threshold for survival which can be comparatively low, e.g., 19°C for stoneflies). In addition, riparian vegetation can also slow water movement from catchment to stream can prevent large pulses of water flows instream following high rainfall (Hupp and Osterkamp 1996; Naiman and Décamps 1997; Anderson and others 2006). Death and Collier (2010) found that streams with 40-60% upstream native riparian vegetation is likely to retain 80% of the biodiversity that would be found in pristine forest streams, and that those with 80-90% native forest or scrub yields macroinvertebrate assemblages indicative of clean water.
33. Riparian buffer zones can prevent erosion and changes in the gravels of river beds and the riffle, pool, run sequences within a river. They can also prevent rivers from changing course, thereby preserving a rivers natural meander. These geomorphological features are part of a river's natural character and form and are also important in providing habitat to the inhabiting species. Therefore, riparian buffer zones are important in safeguarding natural character and life-supporting capacity.
34. The riparian buffer zone can also provide suitable habitat for the adult stages of many aquatic invertebrates (the in water life stage of many aquatic animals is the juvenile form with winged adults emerging from the water to mate and reproduce) (Collier and Smith 1997). Riparian buffer zones, particularly those with forested vegetation, are also important for providing instream habitat for native fish and trout by enhancing habitat diversity (e.g., overhanging branches, bank undercutting), creating pools and areas of day time and flood refuge.

Grassy or forested river banks and lake shores also provide spawning habitat for Inanga and other Galaxias species, respectively. Terrestrial insects from riparian zones often form a major component of the diet for many native and sport fish at certain times of the year (Jefferies 2000; Collier and others 2002; Thompson and Townsend 2003). Furthermore, the addition of terrestrial matter into aquatic systems may stabilise the food webs by providing an alternative energy pathway should one become perturbed (Huxel and McCann 1998; Jefferies 2000; Huxel and others 2002; Takimoto and others 2002; Thompson and Townsend 2003). Thus riparian buffer zones also serve to maintain the proper ecological functioning of instream communities.

35. Vegetated riparian buffer zones can shade streams thus limiting periphyton growth and lowering water temperature. Shade from riparian vegetation can slow periphyton growth by reducing photosynthesis and limiting the amount of energy available for growth. I liken this to building a house, a house with fewer builders will take longer to build than a house with many builders. Riparian shade can also prevent water temperatures from becoming too high. Quinn and others (1997) found that on average streams without riparian buffers (those in pasture) were 2.2°C warmer, had 30-fold higher periphyton biomass, and 11-fold higher gross photosynthesis. However, riparian buffers need to be sufficiently long and wide enough with enough shading to significantly lower temperature and control periphyton (Rutherford and others 1997; Storey and Cowley 1997).

36. A primary function of riparian buffer zones is absorbing nutrients before they enter the stream or river. The roots of the vegetation absorbs nutrients from water passing through soil from the mainland into the stream (Parkyn and others 2003; Parkyn 2004). Smith (1989) found that even retiring 10-13m of pasture can remove up to 67% of nitrate and 55% of dissolve phosphorus surface flows flowing into small headwater streams. However the effectiveness of riparian buffers to remove nutrients is highly variable and depends on plant nutrient uptake rates (which can change as the riparian matures and recycling occurs), and the direction, depth of velocity of flows (Fennessy and Cronk 1997; Parkyn 2004). Therefore, riparian buffers cannot be used alone to control nutrients, rather the best way is reducing direct and indirect nutrient inputs.

37. Riparian buffer zones can also reduce instream sediment loads by preventing

the erosion of banks. The roots of the buffer zones, ground hugging vegetation, and decomposing leaf litter hold and protect soil during high rainfall and prevent the topsoil from being washed into the stream (Naiman and Décamps 1997; Quinn and Stroud 2002; Parkyn 2004). Smith (1989) found that in small headwater streams, simply retiring 10-13m of pasture reduced instream suspended sediment by up to 87%.

38. Livestock access to waterways results in the loss or destruction of the riparian buffer zone, significantly compromising its ecological function (Quinn and others 1992; Bagshaw and Policy 2002; Collins and others 2007). Cattle and dairy cows, if given access to waterways, have a preference (in one study up to 50 times greater) for urinating and defecating directly into the waterway that will contribute to elevated levels of nitrogen and microbial contaminates (Bagshaw 2002; Davies-Colley and others 2004). Livestock (principally cattle, dairy cows and deer) trampling and wallowing can result in sediment deposition into streams, rivers and lakes. This can result in increased levels of deposited fine sediment with the direct detrimental ecological effects highlighted above. Phosphorous is also bound to the sediment and this can subsequently dissolve into the water and become available for periphyton growth. Finally, livestock grazing will remove or degrade any riparian vegetation that might provide stream cover (to reduce light and temperature), stabilise banks, and provide habitat for aquatic and terrestrial invertebrates which are part of the aquatic food web (Quinn and others 1992; Williamson and others 1992; Bagshaw 2002; Collins and others 2007).

39. Whilst the proposed plan change recognises the importance of riparian buffers in protecting mahinga kai values, I do not think the proposed plan adequately recognises nor provides for the establishment of vegetated riparian buffer zones which is a crucial component to safeguarding the life-supporting of native fish and shredding invertebrates, and safeguarding ecosystem processes such as food web stability and indirect trophic effects. It is my opinion that further attention should be given to providing for and establishing vegetated riparian margins in the proposed plan. To effectively manage the freshwater ecosystem health, I recommend that 80-90% of the streams and rivers within the catchment be excluded from livestock and covered with naturalised native vegetation. I place emphasis on the “naturalised” component because the

catchment differs vastly and there are many very rocky, steep gullies where it is inappropriate nor natural to plant with native bush. In many upland areas, naturalised could simply mean low growing grasses. Therefore, the extent to which buffers need to be naturalised will be dependent on the reach in hand and many will not require manual intervention. Buffers should be at least 10m wide to ensure they are effective in reducing sediment, maintain the natural form and character of the stream and channel, have considerable nutrient uptake, have sufficient room for vegetation to grow and support aquatic life and for plants to grow large enough to provide useful levels of shading (Smith 1989; Naiman and Décamps 1997; Parkyn 2004; Collins and others 2007; Quinn and others 2009; Death and Collier 2010).

Nutrient management

40. For any organism to grow there must be sufficient nutrients available as nutrients are essential building blocks and without one an organism will not be able to be healthy or even grow at all. Whilst they may be essential, there can be too much of a good thing such that excessive nutrients can be detrimental to the life supporting capacity of freshwater ecosystems. High nutrient concentrations can impact freshwater taxa either by becoming toxic or by enabling competitive exclusion.

41. High levels of ammonia can be toxic to fish and invertebrates as a small rise can raise pH levels and damage gills and fins, thus making gas exchange and movement difficult, and if not reduced quickly it can fatal. Many experimental studies have found that New Zealand native freshwater species all differ considerably in their tolerance to ammonia and that many common taxa (such as *Deleatidium spp.*) are particularly sensitive to ammonia. Even concentrations as low as 0.1mg/L can be fatal with sufficient exposure (Hickey and Vickers 1994a; Hickey and Vickers 1994b; Richardson 1997; Hickey and others 1999; Hickey and Martin 1999). Furthermore, Richardson, Williams and Hickey (2001) experimentally found that native freshwater fish and shrimp actively avoid water flows with high ammonia concentration and seek those with lowest ammonia. Given these findings, the proposed ammonia concentration

limits in Table 15(c) are suitable at all locations and I support their retention (Hickey and Vickers 1994a; Hickey and Vickers 1994b; Richardson 1997; Hickey and others 1999; Richardson and others 2001; Quinn and Stroud 2002). Though pH does not currently appear to be an issue at any of the relevant monitored sites, I also recommend that future management practices also ensures pH levels are kept between 6.5 and 8.5 at all locations (Collier and Winterbourn 1987; Collier and Winterbourn 1990; Winterbourn and McDiffett 1996; Hogsden and Harding 2011). Furthermore, since ammonia has shown to cause issue and be present at toxic in Waikakahi Stream and Whitneys Creek then I recommend also monitoring and managing for %-EPT taxa and %EPT abundance as QMCI has been shown to be ineffective in responding to changes in ammonia (Hickey and Vickers 1994b; Richardson 1997; Hickey and others 1999).

42. Nutrients also limit periphyton growth by capping the amount of growth that can occur. Once nutrients are no longer sufficiently available, periphyton growth ceases. Furthermore, the nutrients that are almost always limiting are either Dissolved Inorganic Nitrogen (DIN) or Dissolved Reactive Phosphorus (DRP). If either nutrient becomes limiting, then growth is also limited. Back to the analogy of building a house, having a limiting nutrient is like having no more bricks or mortar to continue building. However, unlike a house periphyton does not stop growing once the plan has been built, instead periphyton will continue growing until at least one resource becomes limiting (often Nitrogen or Phosphorus) or a flood scours the periphyton away (a bomb goes off and the house is destroyed then rebuilt until resources are limiting again). When the limiting nutrient is increased then periphyton biomass will continue to increase. High nutrient concentrations can allow periphyton to grow excessively, suffocate other wildlife (as discussed above) and cause competitive exclusion to occur. This is known as the Paradox of Enrichment. Therefore, it is necessary to manage instream nutrient concentrations for both nitrogen (DIN) and phosphorus (DRP) to prevent excessive periphyton growth from occurring.

43. Thus, when setting in-stream nutrient concentration limits, both Dissolved Inorganic Nitrogen (DIN) and Dissolved Reactive Phosphorus (DRP) need to have limits. Given that the management of only one nutrient is fraught with risk as flow, temperature, pH and nutrient fluxes can easily switch a DRP limited

stream to a DIN limited stream, and vice versa (Briand 1983; Wilcock and others 2007); that different algae species thrive in and are composed of different N:P ratios (Milner 1953; Biggs and Price 1987; Biggs 1990); and that two recent reviews of an extensive array of studies (237 and 382 studies, respectively) have found Redfield ratios (the molar N:P ratio) are inaccurate for determining nutrient limitation (Francoeur 2001; Keck and Lepori 2012), I support the use and retention of the proposed limits for both dissolved inorganic nitrogen (DIN) and dissolved reactive phosphorus (DRP).

44. In determining appropriate limits for both DIN and DRP, I recommend that a multiple lines of evidence approach is used.

45. Matheson and others (2016) used quantile regression on data from several regions, including Canterbury, and concludes that to achieve a chlorophyll a (chla) concentration of 50 mg/m2 then DIN needs to be below 0.1mg/L, for chla of 120mg/m2 then DIN needs to be below 0.63mg/L and DRP below 0.011mg/L, and for a chla of 200mg/m2 (the NPS-FM bottom line) then DIN needs to be below 1.1mg/L and DRP below 0.018mg/L. Several of the proposed DIN values in Table 15(c) massively exceed the 1.1mg/L guideline to achieve the NPS-FM bottom line for chlorophyll a.

46. The Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC 2000) suggest a DIN limit of 0.444mg/L and DRP of 0.01 for lower rivers, and a DIN limit of 0.167mg/L and DRP of 0.09mg/L for upper rivers.

47. Joy (2009c) calculated the fish-IBI from 22,500 sites all over the country based on New Zealand Freshwater Fish Database data. When this data is regressed against nutrient data as predicted by Unwin and Larned (2013), then to achieve an IBI of 32 then DIN would need to be at least 0.21 mg/L (r^2=0.09, df=3775, $p<0.00001$) and DRP would need to be 0.007mg/L (r^2=0.04, df=15770, $p<0.00001$) (Death and others Submitted).

48. If we look across all Environment Canterbury monitoring data collected between 2008 and 2014 (inclusive) then for sites with a QMCI of 5 (+-0.3) then the geometric mean (of the mean concentration collected for the three months prior) DIN concentration is 0.38mg/L and DRP is 0.009mg/L, whereas those with a QMCI of 6 (+-0.3) then DIN is 0.16mg/L and DRP is 0.005mg/L. These values are likely to be the most reliable numbers to use as they are derived from empirical measurements within the region and will most closely resemble

the geography in the PC5 area. They are also broadly in line with the values derived using the alternative methods.

49. Clapcott and others (2013) used over data collected between 2007 and 2011 from over 1000 reaches around the country to predict (using random forests) MCI at all locations around the country (r=0.83). The predicted MCI values were converted to approximate QMCI values and then regressed (ln(y)=x) against modelled nutrient data from Unwin and Larned (2013). DIN concentration is approximately 0.45mg/L for QMCI =5 and 0.02mg/L for QMCI = 6 (r^2=0.54, p<0.00001). DRP concentration is 0.012mg/L for QMCI=5 and 0.003mg/L for QMCI=6 (r^2=0.39, p<0.0001) (Death and others Submitted).

50. For the nutrient concentrations to be aligned with (and approximately achieve) the proposed QMCI outcomes in Table 15(a):
    - The hill-fed upper sites DIN concentration should range between 0.02mg/L and 0.167mg/L, and DRP between 0.005mg/L and 0.09mg/L.
    - The hill-fed lower sites DIN concentration should range between 0.02mg/L and 0.444mg/L, and DRP between 0.005mg/L and 0.09mg/L.
    - The plains sites DIN concentration should range between 0.21mg/L and 0.63mg/L, and DRP between 0.009 and 0.012.

51. Overall to All nitrogen loads in the plan will need to be recalculated for sub-catchments where in-stream concentrations are changed.

52. Most of the proposed Total Nitrogen (TN) and Total Phosphorus (TP) for lakes are a bit too high to meet the desired TLI. Using Burns and Bryers (2000) TLI criteria: The large high country lakes need a TN of 61mg/m^3 and TP of 3.41mg/m^3; Lake Alexandrina needs a TN of 157 mg/m^3 and TP of 9 mg/m^3. Lake McGregor needs a TN of 193 mg/m^3 and TP of 11.2 mg/m^3. Lake Middleton needs a TN of 265 mg/m^3 and TP of 15.6 mg/m^3. Lake Benmore Ahuriri Arm needs a TN of 148.6 mg/m^3 and TP of 8.51 mg/m^3. Lake Benmore at Dam and Haldon Arm need a TN of 131.8 mg/m^3 and TP of 7.53 mg/m^3. Lake Aviemore needs a TN of 73 mg/m^3 and TP of 4.1 mg/m^3. The levels proposed for Kellands Pond are suitable.

CONCLUSIONS

53. Ecosystem health is poor in many areas in the Waitaki sub-catchment. Both the current and proposed median nitrate levels are approximately two to ten times too high to safeguard life-supporting capacity or achieve the proposed ecosystem health outcomes at numerous locations.

54. I have recommended a suite of environmental limits and outcomes. These include limits on sediment, nutrient concentrations, temperature, oxygen, periphyton biomass, and freshwater community health indices. The outcomes recommended are necessary to safeguard life-supporting capacity, ecosystem processes and indigenous species in for rivers.

55. Management of both DIN and DRP is recommended and limits should be set to maintain ecosystem health, not nitrogen toxicity or/and to suit current land use. The values suggested in paragraph 50 are necessary to safeguard life-supporting capacity, ecosystem processes and indigenous species in for rivers.

56. Riparian management and planting is recommended to safeguard natural river character (such as pool, riffle, run sequences), temperature, flow, sedimentation and provide food for aquatic life to ensure ecosystem health is maintained.

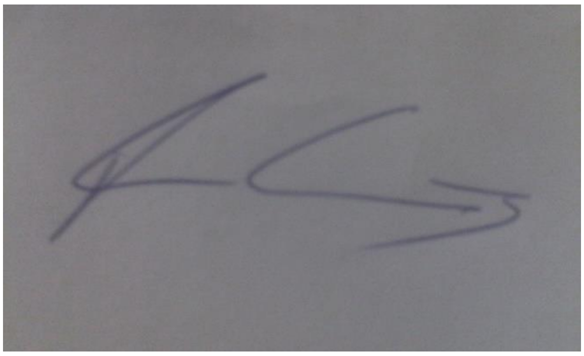
57. Further analysis needs to be undertaken to ensure that coastal ecosystems are not negatively impacted by the riverine limit setting process. I.e., river water quality also needs to take into account downstream/coastal affects.

58. The nutrient outcomes for lakes also need to be adjusted so they are in line with the proposed TLI outcomes.

Adam Douglas Canning

Aquatic Ecologist

19th of July 2016



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