National Objectives Framework - Temperature, Dissolved Oxygen & pH

Proposed thresholds for discussion

Prepared for Ministry for the Environment

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Executive summary

In early June 2013, as part of the Water Reform process, the Ministry for the Environment requested that NIWA outline a National Objectives Framework for rivers to protect the value "Ecosystem Health" (and indigenous species) for the attributes (referred to here as variables): temperature, dissolved oxygen and pH. We were asked to recommend thresholds (limits) for discussion and review by the scientific and end-user community. If waters fall below the national 'bottom lines' (D-graded waters) this is intended to trigger management response to improve the environmental condition.

Background information is provided about each variable, as well as regulatory approaches used internationally, and existing data on native species. These form the basis of the draft thresholds we have proposed.

Temperature, dissolved oxygen and pH interact and therefore we have produced a combined report that includes recommendations about how to take into account the effect of these potentially multiple stressors on aquatic communities.

In addition to the draft thresholds our key recommendations are:

- In the absence of explicit interaction criteria, we suggest that if temperature and at least one other stressor (say DO) *both* indicate a "C" grading, that should be interpreted as a "D" (unacceptable) 'overall' grading for the water body.
- Temperature, dissolved oxygen, and pH vary on a diel cycle, and at times (e.g., summer) are ideally monitored continuously in order to understand and manage the full range of stressors to which organisms are exposed. Continuous monitoring of temperature is fairly straightforward, although 'snap-shot' calibration data, is still essential. Continuous monitoring of DO (and pH) is significantly more challenging, but New Zealand guidance documents (NEMS standards) have recently been produced.
- Some might expect different NOF limits for temperature to recognise generally warmer conditions in lowland *versus* upland waters and also a general weak latitudinal gradient in stream temperature. However, to avoid having to draw a somewhat arbitrary distinction between lowland and upland waters (or northern and southern waters – which are on a continuum), and in the interests of simplicity, we propose a single set of thresholds for *all* NZ rivers. In doing so we recognise that lowland waters, with generally higher temperatures associated with some degree of seasonal thermal stress on sensitive fauna, may be lower-graded. We have however, recognised that some areas of the country ('Eastern Dry" climates) are hotter and drier (therefore streams are shallower and heat more rapidly) than other ('Maritime' climates) with a +1^{o}C shift in temperature limits.
- A better way to account for 'regionality' of temperature, in principle, is to refer stressful mid-summer conditions to (near-pristine) reference streams. Accordingly, in addition to *absolute* temperature limits we also propose *relative* temperature limits as *increments* for stressful high temperatures above those (modelled or measured) in nearby reference streams.

The following tables provide our recommended NOF thresholds for temperature, dissolved oxygen and pH expressed in both narrative and numeric form to protect ecosystem health and indigenous species.

Proposed NOF limits for temperature regime in rivers and streams in 'Maritime' regions of New Zealand. We used the term temperature regime as a reminder that account must be taken of the diel fluctuation of temperature around the daily mean – especially in summer when animals are most likely to be exposed each day for a few hours in the afternoons to particularly high temperatures.

Value (use)		**Ecological Health**
Attribute		**Temperature regime**
Environment (river, lake, GW, estuary, wetland)		Rivers (Maritime climates)
Measurement unit		Degrees Celsius (^{o}C)
Summary statistic		Summer period measurement of the Cox-Rutherford Index (CRI), averaged over the five (5) hottest days (from inspection of a continuous temperature record).
Band descriptors (narrative – what will people notice as the impact on the value)	A	No thermal stress on any aquatic organisms that are present at matched reference (near-pristine) sites.
	B	Minor thermal stress on occasion (clear days in summer) on particularly sensitive organisms such as certain insects and fish.
	C	Some thermal stress on occasion, with elimination of certain sensitive insects and absence of certain sensitive fish.
	D(unacceptable/does not provide for value)	Significant thermal stress on a range of aquatic organisms. Risk of local elimination of keystone species with loss of ecological integrity.
Band boundaries (numeric)	A/B	≤18^{o}C
	B/C	≤20^{o}C
	C/D	≤24^{o}C
	D(unacceptable/does not provide for value)	>24^{o}C
Are there circumstances where a water body could naturally fall into the D band?		Geothermal waters are excluded. Small, naturally unshaded, lowland streams may have large diel temperature fluctuations superimposed on seasonal maxima in mid-late summer, and the summer maximum temperature may then exceed 24^{o}C. Braided rivers may be similarly affected in side braids with slow flushing rates, when consideration should be given to whether the flow regime is appropriate.
Limitations/gaps/risks		Chronic data are available for only limited numbers of native fish and invertebrate species. However, that data is complemented by international species "surrogates", such as chinook salmon and rainbow trout (important recreational fish species in NZ). Together these provide a robust basis for establishing limits. We note that the limits were not derived using a rigorous species tolerance approach for resident species. Limits would be improved by derivation of suitable sub-lethal chronic endpoints (e.g., T_{opt}) and evaluation of reference sites for native species, particularly for macroinvertebrates. The effects of diel variability in temperature have only been quantified for a limited number of macroinvertebrate species and diel temperature ranges (the CRI). Research is required to test and extend this work to other species.
Notes:		1. Summer period is from 1 December to 30 March. 2. The CRI is the average of the daily mean and maximum temperature. 3. Maximum temperature measurements may need to be used in small streams with large diel temperature variations or at sites with minimal monitoring data. 4. Applies to the Maritime Zone of New Zealand *except* if thermal conditions in local *reference* (near-pristine) streams put them into C (slightly degraded) or D (significantly degraded) categories, in which case the site-specific approach may be applied (see other temperature tables below). 5. Applies to point source thermal discharges that are regulated by resource consent; any downstream effects of these point source discharge should be taken into consideration.
References, supporting documentation S1 links		Olsen et al. (2012; and references therein) Cox and Rutherford (2000a,b); Quinn et al. (1994)

Proposed NOF limits for temperature regime in rivers and streams in 'Eastern Dry' regions of New Zealand. We used the term temperature regime as a reminder that account must be taken of the diel fluctuation of temperature around the daily mean – especially in summer when animals are most likely to be exposed each day for a few hours in the afternoons to particularly high temperatures.

Value (use)		**Ecological Health**
Attribute		**Temperature regime**
Environment (river, lake, GW, estuary, wetland)		Rivers (Eastern Dry climates)
Measurement unit		Degrees Celsius (^{0}C)
Summary statistic		Summer period measurement of the Cox-Rutherford Index, averaged over the five (5) hottest days (from inspection of a continuous temperature record).
Band descriptors (narrative – what will people notice as the impact on the value)	A	No thermal stress on any aquatic organisms that are present at matched reference (near-pristine) sites.
	B	Minor thermal stress on occasion (clear days in summer) on particularly sensitive organisms such as certain insects and fish.
	C	Some thermal stress on occasion, with elimination of certain sensitive insects and absence of certain sensitive fish.
	D(unacceptable/does not provide for value)	Significant thermal stress on a range of aquatic organisms. Risk of local elimination of keystone species with loss of ecological integrity.
Band boundaries (numeric)	A/B	≤19^{0}C
	B/C	≤21^{0}C
	C/D	≤25^{0}C
	D(unacceptable/does not provide for value)	>25^{0}C
Are there circumstances where a water body could naturally fall into the D band?		Geothermal waters are excluded. Small, naturally unshaded lowland streams may have large diel temperature fluctuations superimposed on seasonal maxima in mid-late summer, and the summer maximum temperature may then exceed 25^{0}C. Braided rivers may be similarly affected in side braids with slow flushing rates, when consideration should be given to whether the flow regime is appropriate .
Limitations/gaps/risks		Chronic data are available for only limited numbers of native fish and invertebrate species. However, that data is complemented by international species "surrogates", such as chinook salmon and rainbow trout (important recreational fish species in NZ). Together these provide a robust basis for establishing limits. We note that the limits were not derived using a rigorous species tolerance approach for resident species. Limits would be improved by derivation of suitable sub-lethal chronic endpoints (e.g., T_{opt}) and evaluation of reference sites for native species, particularly for macroinvertebrates. The effects of diel variability in temperature have only been quantified for a limited number of macroinvertebrate species and diel temperature ranges (the CRI). Research is required to test and extend this work to other species.
Notes:		1. Summer period is from 1 December to 30 March. 2. The CRI is the average of the daily mean and maximum temperature. 3. Maximum temperature measurements may need to be used in small streams with large diel temperature variations or at sites with minimal monitoring data. 4. Applies to the Maritime Zone of New Zealand *except* if thermal conditions in local *reference* (near-pristine) streams put them into C (slightly degraded) or D (significantly degraded) categories, in which case the site-specific approach may be applied (see other temperature tables below). 5. Applies to point source thermal discharges that are regulated by resource consent; any downstream effects of these point source discharge should be taken into consideration.
References, supporting documentation S1 links		Olsen et al. (2012; and references therein) Cox and Rutherford (2000a,b); Quinn et al. (1994)

Proposed NOF limits for temperature increment in rivers and streams. Limits can be applied on a site-specific basis in New Zealand at council's discretion if sufficient supporting data are available.

Value (use)		Ecological Health
Attribute		**Temperature regime**
Environment (river, lake, GW, estuary, wetland)		Rivers
Measurement unit		Degrees Celsius (^{0}C)
Summary statistic		Summer period measurement of the Cox-Rutherford Index, averaged over the five (5) hottest days (from inspection of a continuous temperature record).
Band descriptors (narrative – what will people notice as the impact on the value)	A	No thermal stress on any aquatic organisms that are present at matched reference (near-pristine) sites.
	B	Minor thermal stress on occasion (clear days in summer) on particularly sensitive organisms such as certain insects and fish.
	C	Some thermal stress on occasion, with elimination of certain sensitive insects and absence of certain sensitive fish.
	D(unacceptable/does not provide for value)	Significant thermal stress on a range of aquatic organisms. Risk of local elimination of keystone species with loss of ecological integrity.
Band boundaries (numeric)	A/B	≤1^{0}C increment compared to reference site
	B/C	≤2^{0}C increment compared to reference site
	C/D	≤3^{0}C increment compared to reference site
	D(unacceptable/does not provide for value)	>3^{0}C increment compared to reference site
Are there circumstances where a water body could naturally fall into the D band?		NO, the temperature ***increment*** approach should account explicitly for natural thermal regimes.
Limitations/gaps/risks		Chronic data are available for only limited numbers of native fish and invertebrate species. However, that data is complemented by international species "surrogates", such as chinook salmon and rainbow trout (important recreational fish species in NZ). Together these provide a robust basis for establishing thresholds. We note that the limits were not derived using a rigorous species tolerance approach for resident species. Limits would be improved by derivation of suitable sub-lethal chronic endpoints (e.g., T_{opt}) and evaluation of reference sites for native species, particularly for macroinvertebrates. The effects of diel variability in temperature have only been quantified for a limited number of macroinvertebrate species and diel temperature ranges (the CRI). Research is required to test and extend this work to other species and for comparison to the proposed limits.
Notes:		1. Summer period is from 1 December to 30 March. 2. The CRI is the average of the daily mean and maximum temperature. 3. Maximum temperature measurements may need to be used in small streams with large diel temperature variations or at sites with minimal monitoring data. 4. Applies to point source thermal discharges that are regulated by resource consent; any downstream effects of these point source discharge should be taken into consideration.
References, supporting documentation S1 links		Olsen et al. (2012; and references therein) Cox and Rutherford (2000a,b); Quinn et al. (1994)

Proposed NOF limits for dissolved oxygen regime in rivers and streams

<table>
<tr><th colspan="2">Value (use)</th><th colspan="6">Ecological Health</th></tr>
<tr><td colspan="2">Attribute</td><td colspan="6">Dissolved oxygen regime</td></tr>
<tr><td colspan="2">Environment</td><td colspan="6">Rivers</td></tr>
<tr><td colspan="2">Measurement unit</td><td colspan="6">Milligrams per litre (mg L^{-1})</td></tr>
<tr><td colspan="2">Summary statistic</td><td colspan="6">Summer monitoring data for discrete specified periods. All 3 statistics must be met for each band.</td></tr>
<tr><td rowspan="4">Band descriptors (narrative – what will people notice as the impact on the value)</td><td>A</td><td colspan="6">No stress caused by low dissolved oxygen on any aquatic organisms that are present at matched reference (near-pristine) sites.</td></tr>
<tr><td>B</td><td colspan="6">Occasional minor stress on sensitive organisms caused by short periods (a few hours each day) of lower dissolved oxygen. Risk of reduced abundance of sensitive fish and macroinvertebrate species.</td></tr>
<tr><td>C</td><td colspan="6">Moderate stress on a number of aquatic organisms caused by dissolved oxygen levels exceeding preference levels for periods of several hours each day. Risk of sensitive fish and macroinvertebrate species being lost.</td></tr>
<tr><td>D (unacceptable/doesn’t provide for value)</td><td colspan="6">Significant, persistent stress on a range of aquatic organisms caused by dissolved oxygen exceeding tolerance levels. Likelihood of local extinctions of keystone species and loss of ecological integrity.</td></tr>
<tr><td rowspan="4">Band boundaries (numeric)</td><td>A/B</td><td>7-day mean[a]</td><td>≥9.0</td><td>7-day mean minimum</td><td>≥8.0</td><td>1-day minimum</td><td>≥7.5[b]</td></tr>
<tr><td>B/C[c]</td><td>7-day mean[a]</td><td>≥8.0</td><td>7-day mean minimum</td><td>≥7.0</td><td>1-day minimum</td><td>≥5.0</td></tr>
<tr><td>C/D</td><td>7-day mean[a]</td><td>≥6.5</td><td>7-day mean minimum</td><td>≥5.0</td><td>1-day minimum</td><td>≥4.0</td></tr>
<tr><td>D (unacceptable/doesn’t provide for value)</td><td>7-day mean[a]</td><td><6.5</td><td>7-day mean minimum</td><td><5.0</td><td>1-day minimum</td><td><4.0</td></tr>
<tr><td colspan="2">Are there circumstances where a water body could naturally fall into the D band?</td><td colspan="6">▪ Geothermally-influenced waters.
▪ Groundwater dominated streams with a large input of low dissolved oxygen groundwater; this can also occur at baseflow conditions in rivers not normally dominated by groundwater.
▪ Sites with a naturally high abundance of macrophytes or periphyton where large diel variations fall below the 4.0 mg L^{-1} 1 day minimum threshold.</td></tr>
<tr><td colspan="2">Limitations/gaps/risks</td><td colspan="6">There are limited data available on the dissolved oxygen tolerances of native NZ species, particularly macroinvertebrates. However, those data are complemented by more detailed information on international species “surrogates” including chinook salmon and rainbow trout, which are important recreational fish species in New Zealand. Together these provide a robust basis for establishing thresholds. We note that (as for international guidelines) these were not derived using a rigorous species tolerance approach for resident species.
The effects of diel variability in dissolved oxygen are poorly understood and therefore may not be accounted for effectively.</td></tr>
<tr><td colspan="2">Notes</td><td colspan="6">[a] 7-day duration alone is insufficient to avoid chronic impacts. It is intended that in any continuous 7-day period throughout the year, this threshold will be met (i.e., this is the annual minimum 7-day mean).
[b] This corresponds to the 80% dissolved oxygen saturation value at the thermal threshold for achieving Class A (18°C).
[c] If the 95th percentile temperature is >24°C, this becomes the threshold for Class D due to the significant interactive effects of high temperature and dissolved oxygen.</td></tr>
<tr><td colspan="2">References, supporting documentation
S1 links</td><td colspan="6">(Dean & Richardson 1999, Franklin 2013, Landman et al. 2005, USEPA 1986a)</td></tr>
</table>

Proposed NOF limits for pH regime in rivers and streams

Value (use)		**Ecological Health**
Attribute		**pH regime**
Environment river, lake, groundwater, estuary, wetland		Rivers
Measurement unit		pH units are dimensionless
Summary statistic		Summer monitoring data upper 95^{th} percentile
Band descriptors	A	No stress caused by acidic or alkaline ambient conditions on any aquatic organisms that are present at matched reference (near-pristine) sites.
	B	Occasional minor stress caused by pH on particularly sensitive freshwater organisms (viz. fish and insects).
	C	Stress caused on occasion by pH exceeding preference levels for certain sensitive insects and fish for periods of several hours each day.
	D (unacceptable/doesn't provide for value)	Significant, persistent stress caused by intolerable pH on a range of aquatic organisms. Likelihood of local extinctions of keystone species and destabilisation of river ecosystems.
Band boundaries (numeric)	A/B	6.5 < pH < 8.0
	B/C	6.5 < pH <8.5
	C/D	6.0 < pH < 9.0
	D (unacceptable/doesn't provide for value)	pH < 6 or pH >9
Are there circumstances where a water body could naturally fall into the D band?		Geothermally-influenced waters Naturally humic-stained streams may also have pH values <5.
Limitations/gaps/risks		These limits may not apply to humic-stained streams. There is a wide range of sensitivities of freshwater fish and invertebrates to pH that is hard to capture with single criteria for each class. Special consideration is needed for (i) naturally acid waters (e.g., humic-stained streams), and (ii) where there are substances for which toxicity is affected by pH (viz. ammonia, toxic metals and sulphide and certain organic contaminants).
Notes:		Summer pH maxima data may need to be used if continuous data available for a site is limited. Continuous monitoring of pH in summer is required to provide reliable data on the diel pH variability.
Key references:		Alabaster and Lloyd (1982); West et al. (1997)

1 Introduction

1.1 Background

A National Objectives Framework is to be established to support regions in setting freshwater objectives and limits so that freshwaters are managed in a nationally-consistent way.

The National Objectives Framework allows management for a standard list of values (Table 1-1), two of which (Ecosystem Health and general protection for indigenous species), and Human Health for Secondary Contact) are to be applied to all water bodies. For each value and attribute minimum states will be described. For each attribute (e.g., temperature) there are four bands – A, B, C and D which represent a range of environmental states. A region may choose to manage to band A, B or C (i.e., to maintain or improve). The boundary between C and D describes the minimum acceptable state (i.e., a “bottom line”) to provide for that value. Choosing to manage for D would not be acceptable.

Table 1-1: Examples of values to be managed under the National Objectives Framework and examples of the attributes to be applied to each value. The first two values are to be applied to all water bodies. Table adapted from information on the MfE website in June 2013 [http://www.mfe.govt.nz/publications/water/freshwater-reform-2013/html/page6.html].

Value	Attributes	Notes
Ecosystem health and general protection for indigenous species	Temperature, dissolved oxygen, pH, periphyton (slime), sediment, flows, connectivity, nitrate (toxicity), ammonia (toxicity), fish, invertebrates, riparian margin	To be applied to all water bodies
Human health for secondary contact	*E. coli*, cyanobacteria	To be applied to all water bodies
Electricity generation	Sediment, flows	
Irrigation	Sediment, flows, *E. coli*	
Stock watering	Sediment, flows, *E. coli*	
Fisheries – for specific species (e.g., trout or inanga)	Flows, sediment, periphyton (slime), temperature, dissolved oxygen, nitrate (toxicity), ammonia (toxicity), invertebrates	
Fish spawning – protection for specific species (e.g., inanga or trout)		
Boating and navigation	Sediment, flows, periphyton (slime).	
Natural form and character	Temperature, periphyton (slime), sediment, flows, connectivity	
Indigenous species –protection for specific species	To be developed	
Swimming	*E. coli*, periphyton, cyanobacteria, water clarity, flows	
Drinking	*E. coli*, cyanobacteria, water clarity	
Food gathering / Mahinga kai	*E. coli*, cyanobacteria, water clarity, riparian margin	
Food production / freshwater aquaculture	To be developed	
Ceremonial uses	*E. coli*, clarity	

1.2 Project Brief

In early June 2013, the Ministry for the Environment (J. Phillips) requested that NIWA address the following issues for temperature, dissolved oxygen (DO) and pH (in order of priority):

1. General background on why these attributes might be important for inclusion in the National Objectives Framework to manage for the value of Ecosystem Health (in rivers – all waters) and/or conversely, what aspect of Ecosystem Health would not be maintained if they were *not* included.

2. Statement of how these attributes are applied elsewhere internationally in a comparable context to the NOF being considered here.

3. What *narrative* could describe a minimum level of Ecosystem Health for these attributes? ('bottom line' descriptor that could be applied to the C/D threshold in the NOF).

4. What information is available to support a *numeric* bottom line?

5. Is there a conceivable numeric bottom line for temperature/DO that could be derived from the information in 4?

6. Is there reasonable monitoring and/or modelled nationwide data that could be used to determine to what extent (and where) the temperature/DO bottom line would pass or fail? Or at least some regional datasets in the more impacted regions? In the absence of any of this, do you have a professional view on to what extent it might be a problem (fail a bottom line) throughout the country?

7. Is there a *narrative* approach to setting an A/B threshold for temperature/DO comparable to that for other attributes (i.e., effectively a reference condition)? Can a *numeric* A/B threshold be derived from this? Is there enough data to test how/where this threshold applies throughout the country?

8. Are there narrative approaches to setting a B/C threshold for temperature/DO? Can numeric thresholds be derived from this? Can it be tested against current state across NZ waters?

In this report we have attempted to account for natural thermal regimes in streams (in stressful dry, mid to late summer conditions) varying with latitude, altitude, and local climate. Such 'regionality' in temperature would ideally be addressed by reference to natural thermal regimes at reference sites. Consequently, in addition to *absolute* temperature limits, we also propose *relative* limits on temperature change compared with a reference site.

1.3 Defining the NOF bands

In April 2012 the second report of the Land and Water Forum (2012) proposed adopting a national instrument with tables describing three levels of protection – excellent, good, and fair. This task was delegated to the National Objectives Framework Reference Group in mid-2012, and the bands are now described as A, B, C and D, with management to D (poor condition) being unacceptable. Labelling the bands as A to D avoids confusion with the classification of water bodies under the international European Union Water Framework Directive and the requirement that all water bodies are managed to a "good" classification (i.e., one that allows “only a slight departure from the biological community which would be expected in conditions of minimal anthropogenic impact”).

In aquatic ecosystems temperature, DO and pH are closely interrelated. Therefore we have provided a combined report with separate derivations of NOF bands for each attribute addressing the project brief above. We follow that with a brief discussion of how the attributes temperature, DO and pH may interact, and how to conduct continuous monitoring so that limits for these variables can be applied.

1.4 Relationship of NOF to ANZECC water quality guidelines

Our recommendation is that the NOF is harmonized with the risk-based approach and narrative effects descriptors used for toxicant effects in the ANZECC (2000) water quality guidelines – so that where possible ANZECC is incorporated into the foundation for the NOF. For example NOF “bands” (i.e., A, B, C or D) or (in mid-2012) “management classifications” (i.e., Excellent, Good, Fair, and Poor) were proposed for nitrate, ammoniacal-N, suspended solids, metals and metalloids. Furthermore, a clear “audit trail” was provided that explained the derivation of the bands (or management classifications), and the strengths and weakness of the derivation (e.g., a lack of native species data, or a lack of invertebrate data) (Winterbourn 1969a).

This approach assists future refinement of the NOF “bands” for each attribute (e.g., nitrate), because it is explicitly stated how the bands have been derived, what data are included/excluded, and why (e.g., data rejected because of insufficient quality assurance). This process is well-advanced for the ANZECC water quality guidelines (currently under revision) and will provide a strong foundation for the NOF. We note that the earlier version of ANZECC (1992) has some additional material (e.g., for dissolved oxygen) that was excluded from ANZECC (2000) but will assist the NOF derivation process.

Note that the ANZECC (2000) guidelines also provide “trigger values” for physical and chemical stressors based on statistical analysis of regional monitoring data from reference sites in Australia and New Zealand (ANZECC, 2000; section 3.3.2.1). The default trigger values in the present ANZECC guidelines were derived from ecosystem data for unmodified or slightly-modified ecosystems supplied by state agencies. However, the choice of these reference systems was not based on any objective biological criteria. As such, the physical and chemical trigger values do not represent thresholds for adverse effects on the respective ecosystems.

2 Temperature

2.1 Background

2.1.1 Importance of temperature

Temperature of water is usually considered part of water quality even though, unlike almost all other water quality variables, temperature is unaffected by water composition. However, temperature very strongly influences water composition, both because it affects the equilibrium point (for instance, the equilibrium solubility of dissolved oxygen) and the rates of physico-chemical and biochemical reactions (notably the rate of dissolved oxygen consumption by bacterial respiration).

Temperature profoundly affects aquatic ecosystems because, besides the (indirect) effects on oxygen and other chemical constituents (e.g., ammonia), growth of most organisms (so-called ‘poikotherms’ that have little or no ability to thermo-regulate) is a strong function of temperature. The temperature of maximum growth rate is referred to as the temperature optimum, T_{opt}. Below that optimum, growth falls fairly slowly, usually to zero near $0^{o}C$ at which temperature the function of enzymes (biochemical catalysts) is strongly inhibited. Some organisms can tolerate lower temperatures than zero (without growth) so long as they avoid tissue damage from freezing by special physiological adaptations such as biological ‘antifreeze’.

Our main concern, though, is for temperatures *above* the growth optimum, because such temperatures impose thermal stress on organisms, and lethal temperatures are reached only slightly higher than (perhaps $5^{o}C$ above) the growth optimum. That is to say, the temperature growth curve for most aquatic organisms is highly asymmetric (Figure 2-1), with a fairly gentle rise in growth rate from near zero at about $0^{o}C$ to a maximum at T_{opt} (this rise being an expression of increasing biochemical reaction rates with increasing temperature); then a steep decline in growth rate above T_{opt} back to zero (no growth). Temperatures higher than the optimum become increasingly stressful because of effects on cellular function with enzymes becoming denatured.

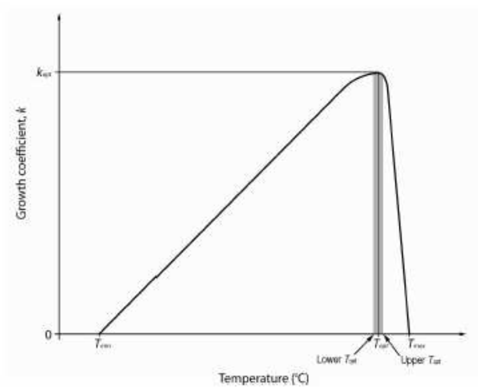

Figure 2-1: Growth coefficient, k, for an organism as a function of temperature (from Olsen et al. 2012). T_{min} = minimum temperature at which growth ceases, T_{max} = maximum temperature at which growth ceases, T_{opt} = temperatures at which growth is optimal (viz., the growth coefficient (k_{opt}) is maximal).

T_{opt}, at least for keystone organisms, provides a theoretically superior criterion for temperature management compared with measurements of lethal temperatures. For long-term (chronic) management we want to avoid temperatures going into the 'stress zone' for organisms, let alone approaching acutely lethal temperatures. Unfortunately less information is available on growth curves from which to define T_{opt} than for lethal temperatures because of the very substantial effort required to measure growth curves for just one species (Olsen et al. 2012).

2.1.2 Controls on temperature in river waters

On a seasonal basis, fastest heating, and therefore fastest rate of temperature rise, occurs near the summer solstice (22 December) when solar altitudes are highest and day-lengths longest, so that insolation (or solar irradiation) is seasonally maximal (cloudiness patterns may also influence insolation, usually delaying the maximum somewhat in, often cloudy, New Zealand). Maximum temperatures are typically achieved a little later, in late January to early February when seasonally averaged insolation is balanced by seasonally averaged heat loss processes. The seasonal cycle of temperature in rivers is well-fitted by a simple sinusoid (e.g., Mosley 1982; Figure 2-2).

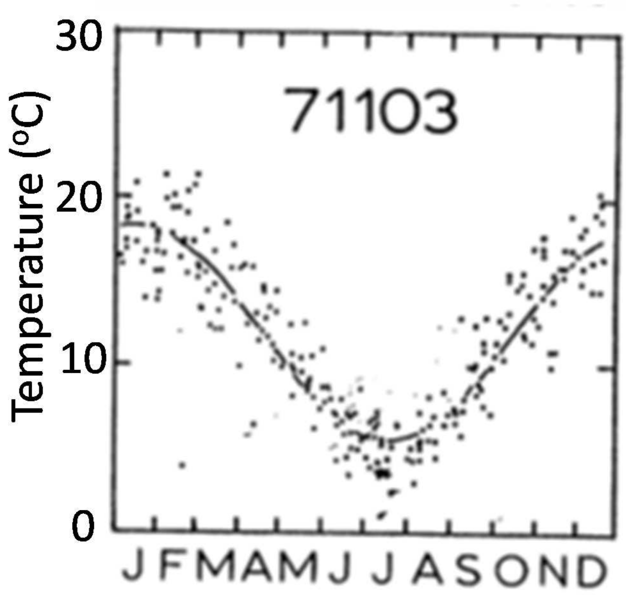

Figure 2-2: Seasonal trend in temperature of river waters. Temperature data are for the Hakataramea River, South Canterbury (Hydrometric station 71103). The plot is taken from Mosley (1982) who fitted a simple sinusoidal equation (curve shown) to account for the seasonal change in spot temperature data taken by flow gauging teams.

The diel cycle of temperature is a little more complicated with the diel pattern being appreciably asymmetric because of rapid solar heating in daylight hours (reaching a maximum in the mid to late afternoon) with fairly rapid cooling through the remainder of the day and slower cooling through the subsequent night to reach a trough (minimum temperature) near dawn (Figure 2-3).

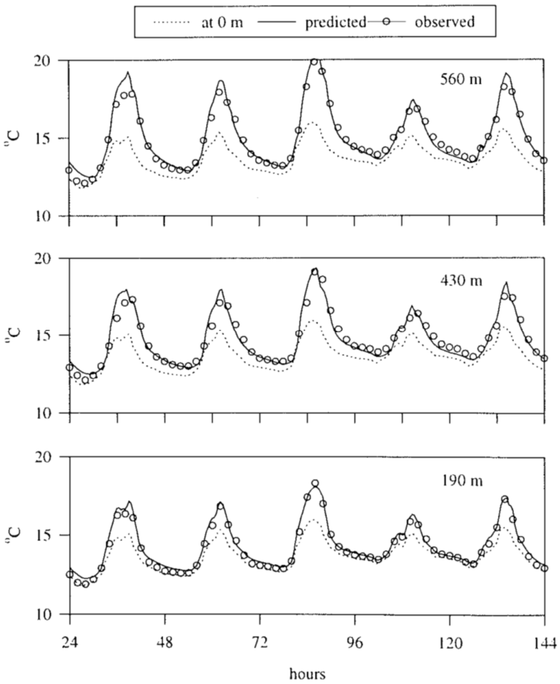

Figure 2-3: Diel trend in temperature of rivers and streams. The plot is taken from Rutherford et al. (1999) based on the modelling and experimental work of Rutherford et al. (1997). Observed temperature data (circles) for three sites at different distances downstream of a forest reserve (at 0 m) along the (2nd order) Kiripaka Stream, Whatawhata (12-16 December, 1993). Data are fitted by model predictions (continuous curve) based on the heat balance equation. The model accounts fairly well for diel temperature fluctuation driven by insolation.

Both the seasonal and diel fluctuation in temperature of rivers must be accounted for with NOF limits for temperature. We are concerned here mainly with thermal stress in mid to late summer (which we have defined for the NOF as 1 December to 31 March) near the seasonal maxima of temperatures and, sometimes, perhaps a little later because of the typical lag of seasonal patterns in river flow compared to temperatures. Note that rate of solar heating is inversely proportional to water depth (Rutherford et al. 1997), so the most 'unfavourable' thermal conditions in streams occur when there is a 'convergence' of clear days with extended dry conditions resulting in low flows – which is typically a feature of settled dry weather in mid to late summer.

2.2 Criteria for temperature for NZ native organisms

The task of assembling temperature criteria contributing to NOF limits for temperature was greatly aided (if not actually simplified) by a comprehensive review for New Zealand native organisms (Olsen et al. 2012). This review gives extensive discussion as background to thermal stress and compiles a range of temperature criteria in extensive tables. Here we give a brief overview of this valuable reference, but readers are referred to the review for details at:

http://www.aucklandcouncil.govt.nz/EN/planspoliciesprojects/reports/technicalpublications/Documents/TR2012036watertemperaturecriteria.pdf

The review by Olsen et al. (2012) focusses firstly on fish, and secondly on macroinvertebrates, although a fairly brief discussion of periphyton is given. Available data suggests that periphyton are generally more temperature-tolerant than stream animals, so periphyton are not further considered here.

A difficulty with interpreting the temperature criteria compiled by Olsen et al. (2012) is that a rather large number of thermal indexes or parameters are discussed because different experimental approaches yield different parameters. Furthermore there is some inconsistency in the literature as regards nomenclature. Comparing different experiments with different methods yielding different parameters or temperature indexes becomes quite confusing, although Olsen et al. (2012) have done an excellent job steering through the diverse literature (Figure 2-4).

For example, one of the most fundamental temperature parameters is the optimum growth rate temperature, T_{opt} (Figure 2-1). This parameter is recommended by Todd et al. (2008) for developing both acute criteria (e.g., the 2 hour daily maximum) and chronic criteria (e.g., the maximum weekly average temperature: MWAT – the seven-day mean of consecutive daily mean temperatures). Unfortunately, T_{opt} has not been measured for any NZ freshwater animal, probably because of the experimental effort required. However, Todd et al. (2008) give alternative methods for deriving temperature criteria based on other parameters. Furthermore, empirical equations published by Jobling (1981) permit approximate estimation of T_{opt} from other more easily and commonly measured parameters.

What has been measured on NZ native animals is: (1) the critical thermal maximum (CTM) from experiments in which temperature of the test organism is raised gradually (at a constant rate) until test organisms die; and (2) the incipient lethal temperature (ILT) from experiments in which test organisms are held at different constant temperatures for different exposure times (e.g., 96 hr) and the 50 percentile survival is interpolated (LT_{50}). The ILT increases with acclimatisation temperature and the ultimate, upper ILT (UUILT) is the ‘plateaux’ ILT where it becomes independent of acclimatisation temperature. Most experiments with NZ native fish have used the CTM method (e.g., Simons 1986). Conversely most experiments with NZ native invertebrates have used the LT_{50} method (e.g., (USEPA 1986b)).

A noteworthy set of experiments that deal with diel temperature fluctuation in rivers was conducted by Cox and Rutherford (2000a) with subsequent modelling by Cox and Rutherford (2000b). In experiments with both a (sensitive) mayfly, *Deleatidium* sp. and a (tolerant) snail, *Potomapyrgus antipodarum*, they showed that the LT_{50} for a temperature regime varying on a diel cycle was mid-way between the daily maximum and the daily mean temperatures (Figure 2-5). We refer henceforth to the Cox-Rutherford index (CRI = $(T_{max} + T_{mean})/2$) which permits application of (constant) temperature criteria to temperature regimes varying over a diel cycle in rivers. The Cox-Rutherford Index will generally be greatest (i.e., the likelihood of thermal stress is greatest) on clear (cloud-free) days when solar insolation is maximal and the amplitude of diel fluctuation is greatest. In Figure 2-5 we have selected a clear day to illustrate the calculation of this index.

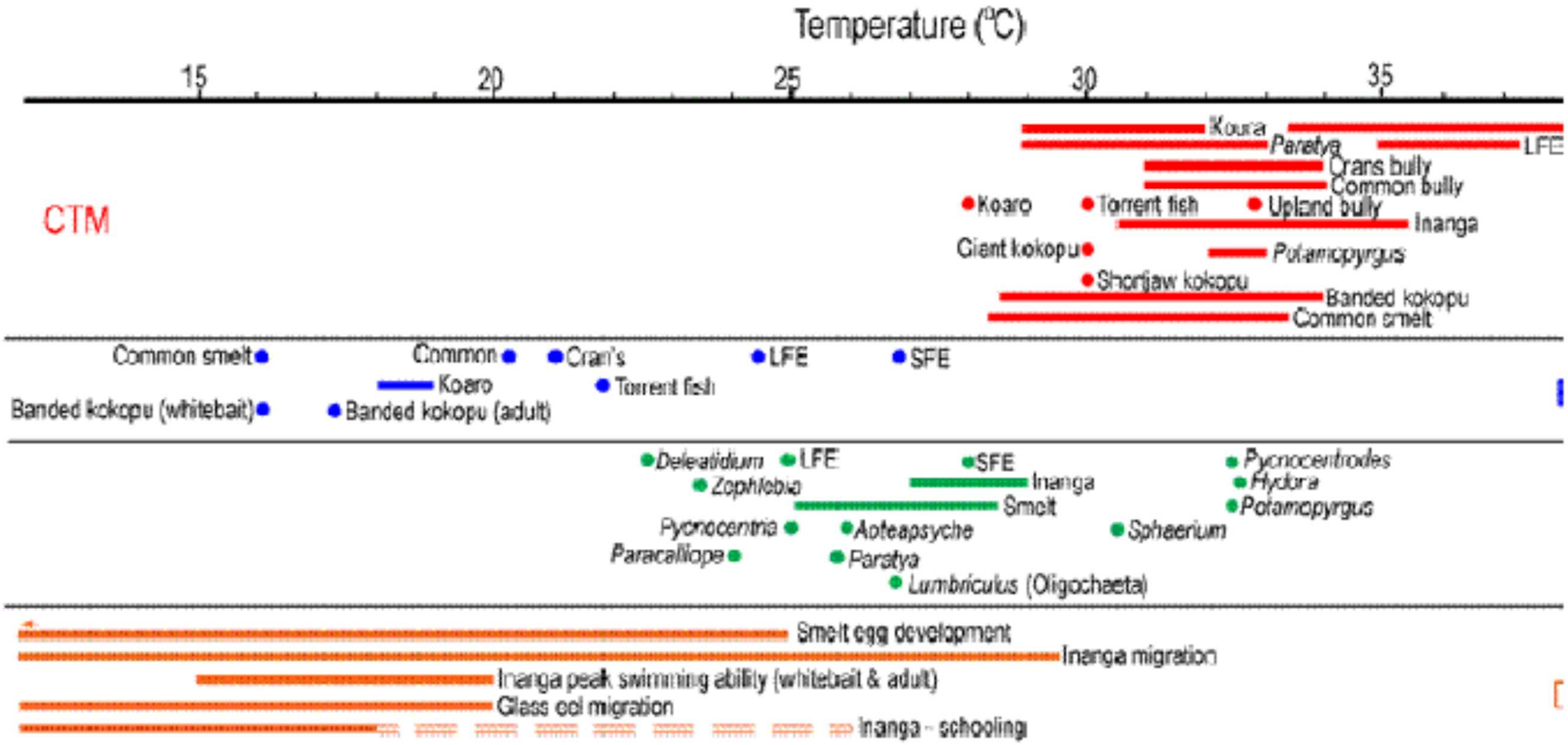

Figure 2-4: Different thermal tolerances for native New Zealand biota as summarized by Olsen et al. (2012). Critical Thermal Maxima (CTM-red), thermal preferences (blue), upper incipient lethal temperature (UILT – green), and behavioural and developmental effects (orange). Olsen et al. (2012) indicated where CTM or UILT have been determined for multiple acclimation temperatures by using a bar to show the range of results. Similarly they indicated with a bar the range of temperatures across which normal behavioural and developmental effects were apparent. A dashed line indicates where inanga schooling is dependent on acclimation temperature.

Experiments with exposure of test animals to different temperatures can, in principle, be augmented by field studies of the occurrence of organisms. This is obviously more useful for (common, abundant) macroinvertebrates than for large mobile, relatively scarce, animals like fish. Most usefully, Quinn & Hickey (1990) reported that stoneflies (Plecoptera) were absent from rivers with annual maximum temperatures[1] over 19°C, and mayflies (Ephemeroptera) were absent from rivers with annual maximum temperatures over 21.5°C. Similarly stoneflies were absent from rivers with annual mean daytime temperatures over 13°C.

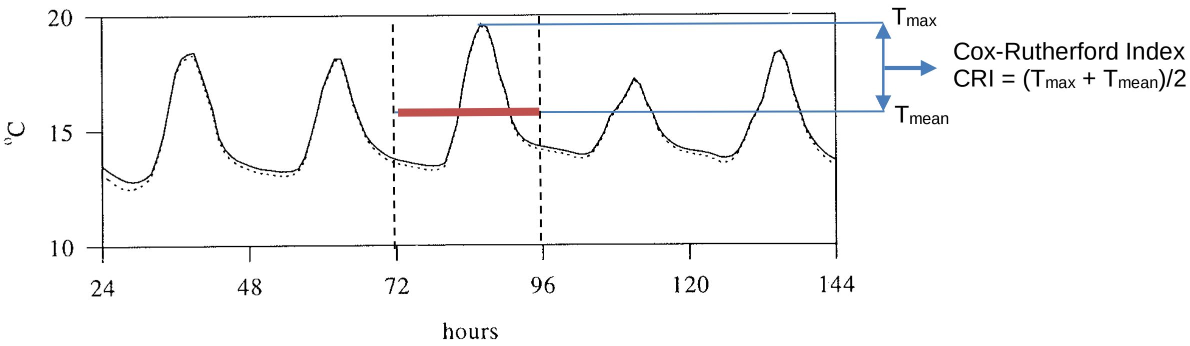

Figure 2-5: Accounting for diel temperature fluctuations in streams and rivers. The plot is taken from Rutherford et al. (1999, Figure 25) based on modelling by Rutherford et al. (1997) of temperatures in the Kiripaka Stream, Whatawhata. An index accounting for diel temperature fluctuations based on the work of Cox and Rutherford (2000a, b) is illustrated for the temperature profile fluctuation of a clear day. The Cox-Rutherford index, in this case, is (19.5+16)/2 = 17.75 °C.

2.3 Water temperature management in other countries

2.3.1 USEPA

The USEPA (1986b) Gold Book defines a protective short-term temperature exposure by subtracting 2°C from the upper incipient lethal temperature (the temperature at which fifty percent of the sample dies), and a protective weekly average temperature.

The protective weekly average temperature is defined as follows:

$$T_{lt} = T_{og} + \frac{(T_i - T_{og})}{3}$$

where:

T_{lt} = maximum permissible temperature for long-term exposure

T_{og} = temperature for optimum growth

T_i = incipient lethal temperature.

[1] Quinn and Hickey (1990) define annual maximum temperature or "MAXTEMP" as mean annual daytime temperatures + half mean winter-summer range (°C)

At least nine species (including three fish, three invertebrates and three plants) are used to determine T_{lt} with the value determined for the most sensitive species being adopted.

In 2003 the USEPA published temperature guidelines for the protection of salmonids in the Pacific Northwest that are based on an extensive literature review and are markedly more conservative than those in the Gold Book (USEPA 2003b). Two of the salmonid species, chinook salmon and rainbow trout (known as steelhead trout), are found in New Zealand. The guideline recommends summer temperature maxima (defined as Maximum 7 Day Average of the Daily Maximums) for certain values (e.g., juvenile rearing and trout migration) of 16 to 18^{o}C, depending on the "use" to be protected – even lower values (Maximum 7 Day Averages of 13 to 14^{o}C) are proposed for spawning and smoltification. Temperature limits are also proposed to protect salmonids from thermal plumes (e.g., temperature limits within the plume and extent of riverbed occupied by the plume).

Furthermore the USEPA recommended that Northwest States and Tribes create standards to protect waters with summer maximum temperatures currently lower than the guidelines – as (for example) these waters are often the last stronghold for endangered salmonid species and are essential for maintaining temperatures lower in the watershed. Finally the USEPA recommends a wide range of collaborative actions (e.g., replanting native riparian margins, reconnecting portions of the river channel with its floodplain, restoring more natural flow regimes to allow alluvial river reaches to function), to restore temperatures in degraded reaches.

2.3.2 Canada

In Canada water temperatures are regulated at a national level by a narrative standard (CCREM 1987). The guidelines specify that the addition of heated water (e.g., cooling water from manufacturing) to natural water bodies should not disrupt natural thermal stratification. In addition in receiving waters Maximum Weekly Average Temperatures (MWAT) are not exceeded, as follows. In warmer months, the MWAT is derived in the same manner as that for USEPA (1986b) (i.e., is equivalent to T_{lt}), but CCREM (1987) specify it should be "for the most appropriate life stage of the sensitive important species that normally is found at that location and time". In cooler months an MWAT definition is specified (regarding thermal additions to natural water) that important species would survive if water temperatures suddenly returned to normal ambient temperatures – with a limit of the acclimation temperature minus 2^{o}C when the lower lethal threshold temperature equals the ambient water temperature. During reproductive seasons the MWAT must meet specific site requirements for reproductive functions such as migration, spawning, egg incubation and fry rearing. Finally MWAT also preserves normal species diversity or prevents undesirable growths of nuisance organisms. Guidelines are provided for acute exposures such that MWAT is not exceeded or that there are any adverse effects on important species by the length and frequency of exposure to elevated temperatures. Short-term temperature limits for growth are "the 24 hour median tolerance limit, minus 2^{o}C, at an acclimation temperature approximating the MWAT for that month". In addition the short term maximum temperature for the season of reproduction should not exceed the maximum incubation temperature for successful embryo survival, or the maximum temperature for spawning (CCREM 1987).

2.3.3 European Inland Fisheries Advisory Commission (EIFAC)

The EIFAC (Alabaster & Lloyd 1982) recommended guidelines divided into three seasons to protect European Inland fisheries. These guidelines are described by CCREM (1987) and are applied to protect those species considered important at the time and location under consideration (e.g., salmonids). During autumn and winter an increase of only 2°C or 6°C above normal was considered damaging to the reproduction of various resident species – with the upper limit applying to early life stages of some salmonids, and causing early spawning in a number of other species. During spring an optimum temperature *range* of only 8°C was considered suitable for spawning and embryonic development. In summer upper permissible temperatures for salmon and trout waters were 20-21°C, although they noted natural waters may rise above this temperatures. Other important food fish (not native to New Zealand), the Coregonids (or whitefish, a type of salmonid) were thought to be able to withstand temperature increases of 5-6°C (presumably above ambient seasonal temperatures) but nonetheless the regulations noted "the sustained maximum should not exceed 22–23°C. Importantly they noted that increases "of 5°C to a maximum no greater than 23°C would destroy salmonid populations, except for some species of *Coregonus*, and an increase of 8°C to a maximum no greater than 30°C would favour a preponderance of some cyprinids" (i.e., carp species).

Water Framework Directive (WFD)

The Water Framework Directive (2000/60/EC) came into force on 22 December 2000. It sets out objectives for the water environment including the following default objectives:

- To prevent deterioration of the status of all surface water and groundwater bodies.
- To protect, enhance and restore all bodies of surface water and groundwater with the aim of achieving Good Ecological Status for surface water and groundwater by 2015.

Under the WFD, individual countries are responsible for setting environmental standards and conditions to underpin domestic implementation of the Directive. Similar to the outlined principles of the New Zealand National Objectives Framework (MfE 2013), the WFD is based on status bands (High, Good, Moderate, Poor, Bad), with the objective of achieving at least Good Ecological Status in all water bodies.

In the UK, the UK Technical Advisory Group on the Water Framework Directive (UKTAG) is responsible for setting environmental standards required to meet the WFD. The UKTAG has taken the approach of where possible deriving standards based on statistical associations between water quality variables and ecological communities (UKTAG 2008a, UKTAG 2008b). Broadly, this involved taking thousands of sites of "good" biological quality and looking at a selected summary statistic of water quality for all sites. The value achieved by 90 per cent of the sites was selected as the standard (UKTAG 2008a).

In the case of temperature however, it was not possible to complete this type of analysis because they note "the adverse effects of temperatures on biology are rare, and tend to be mixed with the effects of other kinds of pressures" - thus it was not possible to isolate paired relationships of "good" ecological status aquatic communities with water temperature variables. Instead water quality standards were developed by literature review and expert consultation including analysis of the temperature requirements of aquatic species with a focus on fish. Standards for macroinvertebrates are still under investigation, with a view to an inclusion at a later date. The UKTAG has focussed on three criteria for fish: lethal temperatures; a preferred range of temperature (which includes feeding, growth, swimming and disease resistance); and the requirements for spawning.

In order to regulate temperature, UK rivers were classified into cool water (i.e., those that support salmonid populations) or warm water rivers (i.e., those that support cyprinid (carp) species), and standards for the WFD are expressed as boundaries between high, good, moderate, poor and bad (ecological status)(Table 2-1)(UKTAG 2008b). The boundary between moderate and poor status is the lower limit of the range of estimates for lethal temperature effects on fish. The standards are expressed as values at the edge of mixing zones of thermal discharges that must be achieved for 98 per cent of the time. The UKTAG (2008b) noted that in some locations more specific locally derived background reference conditions may be required for regulation of thermal discharges.

Table 2-1: UK water temperature standards for rivers. The standards are values at the edge of mixing zones that must be achieved for 98 percent of the time (UKTAG 2008b).

	Temperature (ºC) (Annual 98-percentiles)			
	High	Good	Moderate	Poor
Cold water	20	23	28	30
Warm water	25	28	30	32

The Freshwater Fish Directive also provides imperative guidelines that *must* be met. These include that the edge of mixing zone of a thermal discharge must not exceed the unaffected temperature by more than 1.5ºC for a salmonid river and 3ºC for a cyprinid river. A caveat is included that sudden variations in temperature should be avoided. Absolute standards are provided that at the edge of the mixing zone the water temperatures of 21.5ºC should not be exceeded for more than 2% of the time for "salmonid rivers"; in cyprinid rivers the standard is 28ºC. The caveat is included that species that require cold water for reproduction are protected by an upper limit of 10ºC during the breeding season.

Regarding the issue of the release of cold water (e.g., from the depths of a hydropower lake) the UKTAG (2008b) proposed that these releases not decrease ambient water temperatures by more than 3ºC in all cases, except for waters of high ecological status in which case a maximum decrease of 2ºC was proposed.

2.4 ANZECC guideline for temperature

ANZECC (2000) state that "the maximum permissible increase in the natural temperature of any inland waters should not exceed the 80th percentile of ecosystem reference data, or that temperature set by the formula relating maximum permissible temperature for long-term exposure (T_{lt}) to the temperature for optimum growth and the incipient lethal temperature, whichever is the least." Ecotoxicology testing protocols (see Section 8.3.2. ANZECC & ARMCANZ 2000) are recommended to derive the appropriate temperature guidelines for Australian and New Zealand waters. "The method used will depend on the ecosystem type, the desired level of protection, and the availability of suitable reference systems and adequate data for these systems". The ecotoxicology protocols include detailed guidance such as converting acute endpoints to chronic endpoints, and recommend combining information from field studies with laboratory derived endpoints.

2.5 Interpretation of temperature criteria – NOF limits for temperature

Tables 2-1 to 2-3 set out a proposed national objective framework (NOF) for temperature in running (lotic) waters. Narrative band descriptors are proposed in terms of thermal stress on aquatic fauna. NOF limits for temperature (band boundaries) are (tentatively) proposed for two broad climatic regions in New Zealand, followed by a site-specific option.

In these tables we recognise that thermal regimes of streams, including potentially stressful mid-summer low flow conditions, will vary in a complex way with latitude, altitude and local climate. There is a suggestion from consideration of available continuous temperature data (Section 2-6) that climate is actually a stronger driver of potentially stressful high temperatures than latitude (as such) – probably because rate of solar heating is inversely proportional to water depth and so is unusually rapid during summer low flows in dry climates. Therefore we propose temperature limits 1 degree Celcius higher for 'Easter Dry' climates compared to 'Maritime' regions of New Zealand.

We recognise that reference to natural thermal regimes for streams would be a better approach than somewhat arbitrary reference to climatic conditions. Therefore temperature limits are also proposed in terms of increments above reference (near pristine) conditions.

2.5.1 Narrative band descriptions

Consistent with other NOF frameworks, we propose that A grade waters are those where no aquatic organisms that would be present in reference or near-pristine New Zealand rivers are subjected to thermal stress (Table 1). Conversely, D grade waters (below the 'bottom line') are those in which sensitive aquatic organisms suffer "considerable" thermal stress, with mobile animals (fish) moving to cooler refugia (if they can), while some macroinvertebrates such as certain insects will be locally extinguished. A grading of D also implies some ecosystem level degradation (loss of ecological integrity), which is therefore to be avoided.

Between these extremes, moving from A to B to C-graded waters, there is a gradient of increasing thermal stress on certain (sensitive) aquatic fauna under 'unfavourable' conditions. These occasions will usually be clear days in mid to late summer dry weather when flows are low, water depths are shallow and so solar heating rates are greatest – giving diel temperature fluctuations of greatest amplitude (and highest values for the Cox-Rutherford index of temperature regime).

2.5.2 Compliance statistics for temperature monitoring

Compliance will be evaluated from instruments and datasets that have been assessed and edited for quality control following principles such as those described in the National Environment Monitoring Standards (NEMS 2013).

We propose that the temperature thresholds should be assessed using the Cox-Rutherford Index (CRI). This is defined as the average of the mean daily and daily maximum temperatures, and is a valuable metric because it permits direct application of constant temperature criteria from laboratory experiments (Cox and Rutherford 2000a, b). That is to say, animals respond to diurnally fluctuating temperatures in much the same way as if exposed to a constant temperature equal to the CRI. We consider that if (from inspection of a continuous temperature record) the CRI averaged over the five hottest days in summer exceeded relevant thresholds this would indicate exposure of resident organisms to thermal regimes consistent with the narrative thresholds. In fact, it is most likely thermally stressful conditions will occur during a period of long, clear, dry summer days, and therefore high CRI values will occur in a series of consecutive days. The summer period has been defined as 1 December to 31 March to encompass the maximum temperatures typically occurring in New Zealand streams and rivers.

The CRI was initially found to estimate the LT_{50} for two very different species of invertebrate (i.e., a sensitive mayfly *Deleatidium* sp. and a tolerant snail *Potamopyrgus antipodarum*) exposed to a temperature regime varying on a diel cycle (Cox & Rutherford 2000 a, b). The CRI is a robust metric for the NOF thresholds because it accounts implicitly for thermal stress of a diurnally fluctuating temperature regime, and so permits direct comparison with constant temperature criteria from laboratory experiments. We have proposed absolute temperature thresholds for CRI that incorporate a safety factor of 1^{o}C below the conservative expression of annual maximum temperatures used in Quinn & Hickey 1990 (who used an estimate of the average summer temperature = mean annual daytime temperature + half mean winter-summer range).

If the CRI does not gain acceptance, then we recommend the 95th percentile of summer monitoring data as an alternative statistic to be applied to the same absolute or incremental thresholds proposed in Tables 2-2 to 2-4. We prefer CRI over 95th percentile for the following reasons:

- The relationship between stress or mortality and the CRI has been quantified (albeit for only two macroinvertebrates) but this has not been done for the 95% percentile temperature.
- Such conditions are most likely, for a few hours on the afternoon of very clear days (therefore with maximal solar heating) in late summer dry (low flow) conditions when water depths are very shallow such that heating rates (inversely proportional to water depth – see Rutherford et al. 1997; 1999) are very rapid.
- Therefore continuous temperature records through summer are best used, NOT to estimate percentiles, but to identify and interpret the ‘few’ stressful very hot days (usually a small cluster of days) under adverse hot, dry conditions. Percentiles can only be estimated with high precision with a large amount of data (ideally for several summers covering all of the range between hot and dry versus cool and wet). And all of this continuous data has to be edited for ‘spikes’ and missing data and other errors in order to provide a good estimate of the ‘population’ 95th percentile characterising a stream.

Both statistics (CRI and 95th percentile) should ideally be evaluated against temperature regimes and ecosystem health parameters at appropriate reference and impacted sites in a national validation project. Such work should be undertaken as soon as possible to support the NOF temperature thresholds.

We recognise that NOF thresholds for temperature for two (crudely delineated) broad climate zones will not adequately represent the diversity of microclimates and therefore temperature regimes naturally present in New Zealand streams as a result of factors such as latitude, altitude, hydrology and geology. If the NOF thresholds expressed as absolute temperatures are considered to *not* be representative of local 'reference' conditions we propose that site-specific classification into NOF temperature bands is determined by evaluation against allowable incremental increases or decreases of temperature compared to an appropriate reference site supporting good Ecological Health. Ideally the site will be compared against several years of data from a reference site in order to take account of inter-annual variability in summer maxima (e.g., El Niňo effects).

2.5.3 Numeric boundaries (tentative)

Numeric band boundaries are proposed in Tables 2-2 to 2-4, subject to peer review and discussion among New Zealand ecologists and experts on fish and invertebrate responses to thermal stress. Two sets of numeric bands have been proposed to take into account two broad climate zones for 'Maritime' regions versus 'Eastern Dry'. This split on climatic zones is based on general familiarity with stream thermal regimes and an inspection of available continuous temperature data (Section 2-6) – both of which suggest that climate over-rides latitude in terms of potentially stressful summer temperatures. 'Eastern Dry' regions include mostly lowland areas of: Bay of Plenty, Gisborne, Hawkes Bay, Wairarapa, Nelson, Marlborough, Canterbury and (perhaps) Otago, in which mid to late summer conditions are typically hot and dry with successive clear days producing rapid solar heating of shallow streams at low flow. A Council may assign catchments within these areas to either the Maritime or Eastern Dry zone at their discretion. We were not able to map these zones within the timeframe of this project.

Finally a site-specific approach to temperature thresholds may be applied (Table 2-4) if thermal conditions in local *reference* (near-pristine) streams with 'Good Ecological Health' put them into C (slightly degraded) or D (significantly degraded) categories. This approach carries a greater 'burden-of-proof' because reference data must be obtained, ideally for multiple years (and at least a full summer), in order to take account of inter-annual variability in summer maxima, and to demonstrate 'Good Ecological Health' as expressed in the narrative thresholds.

Table 2-2: Proposed NOF for temperature regime in rivers and streams in 'Maritime' regions of New Zealand. We used the term temperature regime as a reminder that account must be taken of the diel fluctuation of temperature around the daily mean – especially in summer when animals are most likely to be exposed each day for a few hours in the afternoons to particularly high temperatures.

Value (use)		**Ecological Health**
Attribute		**Temperature regime**
Environment (river, lake, GW, estuary, wetland)		Rivers (Maritime climates)
Measurement unit		Degrees Celsius (^{0}C)
Summary statistic		Summer period measurement of the Cox-Rutherford Index (CRI), averaged over the five (5) hottest days (from inspection of a continuous temperature record).
Band descriptors (narrative – what will people notice as the impact on the value)	A	No thermal stress on any aquatic organisms that are present at matched reference (near-pristine) sites.
	B	Minor thermal stress on occasion (clear days in summer) on particularly sensitive organisms such as certain insects and fish.
	C	Some thermal stress on occasion, with elimination of certain sensitive insects and absence of certain sensitive fish.
	D(unacceptable/does not provide for value)	Significant thermal stress on a range of aquatic organisms. Risk of local elimination of keystone species with loss of ecological integrity.
Band boundaries (numeric)	A/B	≤18^{0}C
	B/C	≤20^{0}C
	C/D	≤24^{0}C
	D(unacceptable/does not provide for value)	>24^{0}C
Are there circumstances where a water body could naturally fall into the D band?		Geothermal waters are excluded. Small, naturally unshaded, lowland streams may have large diel temperature fluctuations superimposed on seasonal maxima in mid-late summer, and the summer maximum temperature may then exceed 24^{0}C. Braided rivers may be similarly affected in side braids with slow flushing rates, when consideration should be given to whether the flow regime is appropriate.
Limitations/gaps/risks		Chronic data are available for only limited numbers of native fish and invertebrate species. However, that data is complemented by international species "surrogates", such as chinook salmon and rainbow trout (important recreational fish species in NZ). Together these provide a robust basis for establishing limits. We note that the limits were not derived using a rigorous species tolerance approach for resident species. Limits would be improved by derivation of suitable sub-lethal chronic endpoints (e.g., T_{opt}) and evaluation of reference sites for native species, particularly for macroinvertebrates. The effects of diel variability in temperature have only been quantified for a limited number of macroinvertebrate species and diel temperature ranges (the CRI). Research is required to test and extend this work to other species.
Notes:		1. Summer period is from 1 December to 30 March. 2. The CRI is the average of the daily mean and maximum temperature. 3. Maximum temperature measurements may need to be used in small streams with large diel temperature variations or at sites with minimal monitoring data. 4. Applies to the Maritime Zone of New Zealand *except* if thermal conditions in local *reference* (near-pristine) streams put them into C (slightly degraded) or D (significantly degraded) categories, in which case the site-specific approach may be applied (see other temperature tables below). 5. Applies to point source thermal discharges that are regulated by resource consent; any downstream effects of these point source discharge should be taken into consideration.
References, supporting documentation S1 links		Olsen et al. (2012; and references therein) Cox and Rutherford (2000a,b); Quinn et al. (1994)

Table 2-3: Proposed NOF for temperature regime in rivers and streams in 'Eastern Dry' regions of New Zealand. We used the term temperature regime as a reminder that account must be taken of the diel fluctuation of temperature around the daily mean – especially in summer when animals are most likely to be exposed each day for a few hours in the afternoons to particularly high temperatures.

Value (use)		**Ecological Health**
Attribute		**Temperature regime**
Environment (river, lake, GW, estuary, wetland)		Rivers (Eastern Dry climates)
Measurement unit		Degrees Celsius (^{0}C)
Summary statistic		Summer period measurement of the Cox-Rutherford Index, averaged over the five (5) hottest days (from inspection of a continuous temperature record).
Band descriptors (narrative – what will people notice as the impact on the value)	A	No thermal stress on any aquatic organisms that are present at matched reference (near-pristine) sites.
	B	Minor thermal stress on occasion (clear days in summer) on particularly sensitive organisms such as certain insects and fish.
	C	Some thermal stress on occasion, with elimination of certain sensitive insects and absence of certain sensitive fish.
	D(unacceptable/does not provide for value)	Significant thermal stress on a range of aquatic organisms. Risk of local elimination of keystone species with loss of ecological integrity.
Band boundaries (numeric)	A/B	≤19^{0}C
	B/C	≤21^{0}C
	C/D	≤25^{0}C
	D(unacceptable/does not provide for value)	>25^{0}C
Are there circumstances where a water body could naturally fall into the D band?		Geothermal waters are excluded. Small, naturally unshaded lowland streams may have large diel temperature fluctuations superimposed on seasonal maxima in mid-late summer, and the summer maximum temperature may then exceed 25^{0}C. Braided rivers may be similarly affected in side braids with slow flushing rates, when consideration should be given to whether the flow regime is appropriate.
Limitations/gaps/risks		Chronic data are available for only limited numbers of native fish and invertebrate species. However, that data is complemented by international species "surrogates", such as chinook salmon and rainbow trout (important recreational fish species in NZ). Together these provide a robust basis for establishing limits. We note that the limits were not derived using a rigorous species tolerance approach for resident species. Limits would be improved by derivation of suitable sub-lethal chronic endpoints (e.g., T_{opt}) and evaluation of reference sites for native species, particularly for macroinvertebrates. The effects of diel variability in temperature have only been quantified for a limited number of macroinvertebrate species and diel temperature ranges (the CRI). Research is required to test and extend this work to other species.
Notes:		1. Summer period is from 1 December to 30 March. 2. The CRI is the average of the daily mean and maximum temperature. 3. Maximum temperature measurements may need to be used in small streams with large diel temperature variations or at sites with minimal monitoring data. 4. Applies to the Maritime Zone of New Zealand *except* if thermal conditions in local *reference* (near-pristine) streams put them into C (slightly degraded) or D (significantly degraded) categories, in which case the site-specific approach may be applied (see other temperature tables below). 5. Applies to point source thermal discharges that are regulated by resource consent; any downstream effects of these point source discharge should be taken into consideration.
References, supporting documentation S1 links		Olsen et al. (2012; and references therein) Cox and Rutherford (2000a,b); Quinn et al. (1994)

Table 2-4: Proposed NOF limits for temperature increment in rivers and streams. Limits can be applied on a site-specific basis in New Zealand at council's discretion if sufficient supporting data are available.

<table>
<tr><th colspan="2">Value (use)</th><th>Ecological Health</th></tr>
<tr><td colspan="2">Attribute</td><td>Temperature regime</td></tr>
<tr><td colspan="2">Environment (river, lake, GW, estuary, wetland)</td><td>Rivers</td></tr>
<tr><td colspan="2">Measurement unit</td><td>Degrees Celsius (^{0}C)</td></tr>
<tr><td colspan="2">Summary statistic</td><td>Summer period measurement of the Cox-Rutherford Index, averaged over the five (5) hottest days (from inspection of a continuous temperature record).</td></tr>
<tr><td rowspan="4">Band descriptors (narrative – what will people notice as the impact on the value)</td><td>A</td><td>No thermal stress on any aquatic organisms that are present at matched reference (near-pristine) sites.</td></tr>
<tr><td>B</td><td>Minor thermal stress on occasion (clear days in summer) on particularly sensitive organisms such as certain insects and fish.</td></tr>
<tr><td>C</td><td>Some thermal stress on occasion, with elimination of certain sensitive insects and absence of certain sensitive fish.</td></tr>
<tr><td>D(unacceptable/does not provide for value)</td><td>Significant thermal stress on a range of aquatic organisms. Risk of local elimination of keystone species with loss of ecological integrity.</td></tr>
<tr><td rowspan="4">Band boundaries (numeric)</td><td>A/B</td><td>≤1^{0}C increment compared to reference site</td></tr>
<tr><td>B/C</td><td>≤2^{0}C increment compared to reference site</td></tr>
<tr><td>C/D</td><td>≤3^{0}C increment compared to reference site</td></tr>
<tr><td>D(unacceptable/does not provide for value)</td><td>>3^{0}C increment compared to reference site</td></tr>
<tr><td colspan="2">Are there circumstances where a water body could naturally fall into the D band?</td><td>NO, the temperature increment approach should account explicitly for natural thermal regimes.</td></tr>
<tr><td colspan="2">Limitations/gaps/risks</td><td>Chronic data are available for only limited numbers of native fish and invertebrate species. However, that data is complemented by international species “surrogates”, such as chinook salmon and rainbow trout (important recreational fish species in NZ). Together these provide a robust basis for establishing thresholds. We note that the limits were not derived using a rigorous species tolerance approach for resident species.
Limits would be improved by derivation of suitable sub-lethal chronic endpoints (e.g., T_{opt}) and evaluation of reference sites for native species, particularly for macroinvertebrates.
The effects of diel variability in temperature have only been quantified for a limited number of macroinvertebrate species and diel temperature ranges (the CRI). Research is required to test and extend this work to other species and for comparison to the proposed limits.</td></tr>
<tr><td colspan="2">Notes:</td><td>1. Summer period is from 1 December to 30 March.
2. The CRI is the average of the daily mean and maximum temperature.
3. Maximum temperature measurements may need to be used in small streams with large diel temperature variations or at sites with minimal monitoring data.
4. Applies to point source thermal discharges that are regulated by resource consent; any downstream effects of these point source discharge should be taken into consideration.</td></tr>
<tr><td colspan="2">References, supporting documentation S1 links</td><td>Olsen et al. (2012; and references therein)
Cox and Rutherford (2000a,b); Quinn et al. (1994)</td></tr>
</table>

Otherwise modelling will have to be employed to estimate the clear day, low flow (mid- to late-summer) thermal response of a reference stream. Such models would be tuned to continuous data from the poorly shaded stream under consideration (measurement of bank shading would be required), before modelling the temperature of the *same* stream under full shade (approximately 95% shading of all solar radiation averaged over near-infrared and visible wavelengths – Rutherford et al. 1997). The temperature increment limits would be applied to the shaded *versus* unshaded CRI. That is, the difference in CRI between the poorly shaded and shaded temperature of the stream on a hot, dry day in mid to late summer would be compared to temperature increment limits (Table 2-4).

The Maritime thresholds were developed first, followed by an incremental +1ºC 'adjustment' (proposed for discussion) for Eastern Dry zones. Surprisingly, it is perhaps easier to enumerate the A/B boundary at the no-thermal-effect limit, than the C/D ('bottom line') limit. Consistent with the A grade narrative (Table 2-2), the A/B boundary should be set where no aquatic organism (that would otherwise be present) is subjected to thermal stress, even on the most 'unfavourable' occasions of clear days during low flows in mid to late summer. The review by Olsen et al. (2012), emphasising laboratory experiments, suggests that a maximum annual temperature less than 20ºC should protect even the most sensitive native taxa in upland streams.

However, we note (as also noted by Olsen et al. 2012) that Quinn and Hickey (1990) reported that stoneflies were confined to rivers with annual maximum temperatures[2] less than 19ºC, which argues for the A/B threshold being set (slightly) lower (i.e., 18ºC).

We had some difficulty deciding how to interpret the criteria compiled by Olsen et al. (2012) as regards the 'bottom line' (C/D boundary). Temperatures of around 24-27ºC are indicated, at which several sensitive invertebrates (particularly insects, Figure 2-4 from Olsen et al. 2012) would be severely stressed in summer, and probably eliminated, while certain fish would be absent. The C/D thresholds for the Maritime and Eastern Dry zones (24 and 25ºC respectively) fall in the range of the UILT for the native invertebrates *Deleatidium* spp. (mayflies), *Zephlebia dentata* (mayflies), *Paracalliope fluviatilis* (amphipod), and *Pycnocentria evecta* (sand-cased caddis fly) that were either collected from field locations where the water temperature was in the range of 12-14ºC and were used without acclimation, or were acclimated to 15-16ºC prior to temperature threshold testing using 48 to 96 h exposures (summarized in Figure 2-4, see Tables 2 and 7 in Olsen et al. 2010 for details). Acclimation temperature affects the UILT. Therefore we have assumed that acclimation of stream invertebrates to mean summer temperatures provides a safety factor of 1-2ºC for these NOF thresholds expressed as CRI.

Olsen et al. (2012) favour *separate* management of 'upland' versus 'lowland' rivers and streams, when they state:

"Based on these criteria, maximum temperatures in upland streams that are less than 20°C should protect even the most sensitive native taxa. In comparison, the most sensitive native taxa in lowland streams should be protected as long as maximum temperatures are less than 25°C."

[2] Quinn and Hickey (1990) defined annual maximum temperature or "MAXTEMP" as mean annual daytime temperatures + half mean winter-summer range (°C).

We note that the present day thermal regimes of many lowland streams are elevated as a result of widespread deforestation and are not representative of reference site conditions that support good Ecological Health. In forested catchments the thermal inertia of shaded upstream waters would mitigate the effects of a decrease in altitude. Therefore, we think that, for simplicity, a *single* NOF temperature framework should be applied to *both* upland and lowland waters. As well as being simpler, this avoids the necessity for a (somewhat arbitrary) distinction of upland and lowland waters – which, after all, are end members on a continuum. It does, however, mean that lowland waters will generally grade lower than upland waters – consistent with local absence of some highly sensitive fauna such as stoneflies and some mayfly species in lowland systems, and seasonal absence of sensitive fish. Accordingly, we (tentatively) propose a C/D boundary based on a summer period 5 day maximum CRI of 24ºC.

The B/C boundary is proposed to match the narrative criteria for the B band of minor thermal stress on particularly sensitive organisms to be 20ºC - although we recognise this does not provide even temperature increments between the NOF bands.

It is worth noting that both the absolute and incremental temperature “bottom lines” (limits) that we propose here have some precedent in New Zealand.

The Water and Soil Conservation Act (1967) Section 26c set a numerical standard for temperature increment: “The natural water temperature shall not be changed by more than 3 degrees Celcius” (applying to many, but not all, classifications of waters). Graham McBride (pers. comm.) thinks this standard was probably based on the work of Colin Cowie, then of the Ministry of Works and Development, Water and Soil Division, and was probably designed to protect salmonids.

In their report proposing standards for the Resource Management Act (1991) to the Ministry for the Environment, Burns et al. (1989) recommended an absolute temperature limit of 25ºC – explicitly recognising field observations on native animals such as those of John Quinn and Chris Hickey as regards stoneflies and mayflies (later published as Quinn and Hickey 1990).

These recommendations for numerical temperature standards were adopted in the Resource Management Act (1991) in certain Standards for classifying water (Schedule 3, referring to Section 69) (Classes: AE for aquatic ecosystems, F – for fisheries, FS for fish spawning).

2.5.4 Confidence in temperature thresholds

While there are chronic data for only limited numbers of native fish and invertebrate species, that data is complemented by international species “surrogates”. These include the important recreational fish species in NZ, such as chinook salmon and rainbow trout. Together these provide a robust basis for establishing thresholds. We note (that as for international guidelines) these were not derived using a rigorous species tolerance approach for resident species. We expect that the NOF thresholds will be subject to a review process approximately two years after introduction and that new information can be used to adjust the thresholds to ensure they are effective regulatory tools.

2.5.5 Knowledge gaps - temperature

Olsen et al. (2012) provide a comprehensive review that details the lack of information about native species, particularly the most useful parameters upper T_{opt} and UUILT that would inform the development of acute and chronic thermal criteria. Olsen et al. (2012) also provide alternatives to these endpoints and recommend temperature thresholds which we have applied to the NOF. We note that sub-lethal temperature thresholds derived from laboratory and field studies would increase confidence in the efficacy of the NOF thresholds to protect ecological health.

How should annual maximum temperature be defined? USEPA (2003) guidelines developed for the protection of endangered salmonid species in the Northwest States use the summer maxima defined as the Maximum 7 Day Average of the Daily Maximums. Research is required to develop a practical statistic that can be applied to the thresholds (e.g., 95th or 99th percentile, Maximum 7 Day Average of the Daily Maximums) or *vice versa* (e.g., Cox-Rutherford index) taking into account the data on temperature regime and ecological health that is currently available to councils, and that is logistically feasible to obtain in the near future.

Standard protocols can readily be developed for measurement of annual maximum temperature from existing standard practices, such as those in the recent NEMS project.

Should separate NOF criteria be developed for upland and lowland streams? Upland and lowland streams could be defined by altitude – with the classification system favouring a more conservative designation to upland rivers and streams. The River Environment Classification network may be a useful tool for this (Snelder et al. 2004, Snelder et al. 2000). We recommend the subject of separate temperature bands for upland and lowland rivers and streams as a discussion point, although we currently recommend one set of thresholds for upland and lowland after considering the thermal inertia provided by forested catchments (i.e., reference conditions).

We strongly recommend that this initial proposal for NOF temperature thresholds is followed-up by a validation project to collect data on temperature and ecosystem health from reference and impacted sites to determine the efficacy and practicality of the proposed thresholds. Recent technological advances have made it relatively easy and cost-effective to collect continuous temperature records using temperature loggers. Consultation with councils, and ecologists would produce more robust and pragmatic thresholds.

We have not addressed the impact of release of water cooler than ambient temperatures, most notably from the hypolimnion of large dams.

2.6 Modelling data for evaluation of thresholds and current compliance

At a broad scale of resolution NIWA has accessed monitoring and modelled nationwide data with the cooperation of many regional and district councils, and other organisations (e.g., Universities) to make an initial evaluation of the spatial extent of temperature gradings. However it is beyond the scope of the present project to refine this analysis. Regional datasets are available in some of the more impacted regions. Some councils however, lack the continuous monitoring data to fully define temperature regimes occurring in New Zealand rivers and streams. Only monthly ‘snap shot’ data is likely to be available for many monitoring sites.

Owing to land clearance, many of our rivers and streams are degraded with respect to natural thermal regimes (especially in lowland areas) and may fall into a C or even D band requiring ongoing rehabilitation efforts.

Continuous temperature records from 204 sites distributed around the country (Table 2-5) were analysed for the proportion that meet various temperature thresholds. All sites had at least one summer with records for 150 of the 181 days of summer (Nov-April inclusive), but had various lengths of record. CRI was calculated for each day of record at each site. The mean of the 5 highest CRI's for each year was calculated. When averaged across years, less than 5% of the sites 'failed' the proposed Maritime zone C/D threshold (i.e., bottom line) of 'a CRI averaged over the five (5) hottest days of 24ºC'. When averaged across years, less than 3% of the sites 'failed' the proposed Eastern Dry zone C/D threshold of 'a CRI averaged over the five (5) hottest days of 25ºC'. However, when only the hottest year was taken these percentages increased to 15 and 9% respectively.

This analysis is only indicative because we do not know how representative these records are of microclimates and stream thermal regimes across New Zealand. Furthermore, the records were of various lengths and did not all cover the same time period. For example the differing years of record and record lengths were not explicitly accounted for in our analysis. The analysis suggests that care will be required when comparing observed data with the proposed thresholds since temperature extrema vary somewhat from year-to-year.

Table 2-5: Number of sites with continuous temperature records in each region.

Region	Number of sites
Northland	0
Auckland	7
Waikato	2
Bay of Plenty	5
Gisborne	0
Hawkes Bay	20
Taranaki	22
Horizons	48
Wellington	25
Tasman	2
Marlborough	2
Canterbury	36
West Coast	1
Otago	12
Southland	20

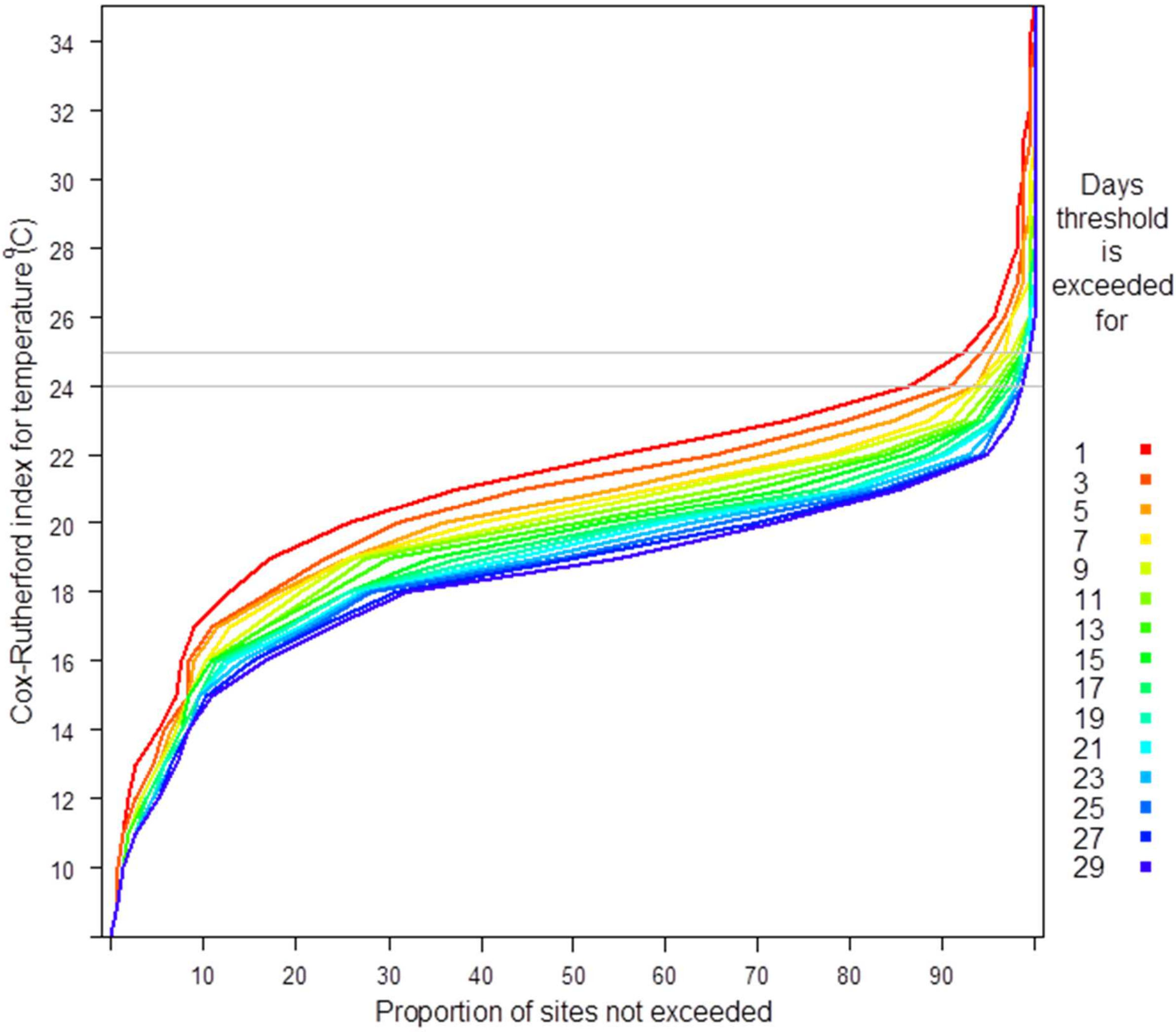

Figure 2-6: Comparison of proportion of sites <u>not</u> exceeding various CRI thresholds. The proposed NOF temperature thresholds at the C/D boundary are for 5 days exceedance of a CRI of 24°C in a maritime zone and 25°C in an Eastern Dry zone.

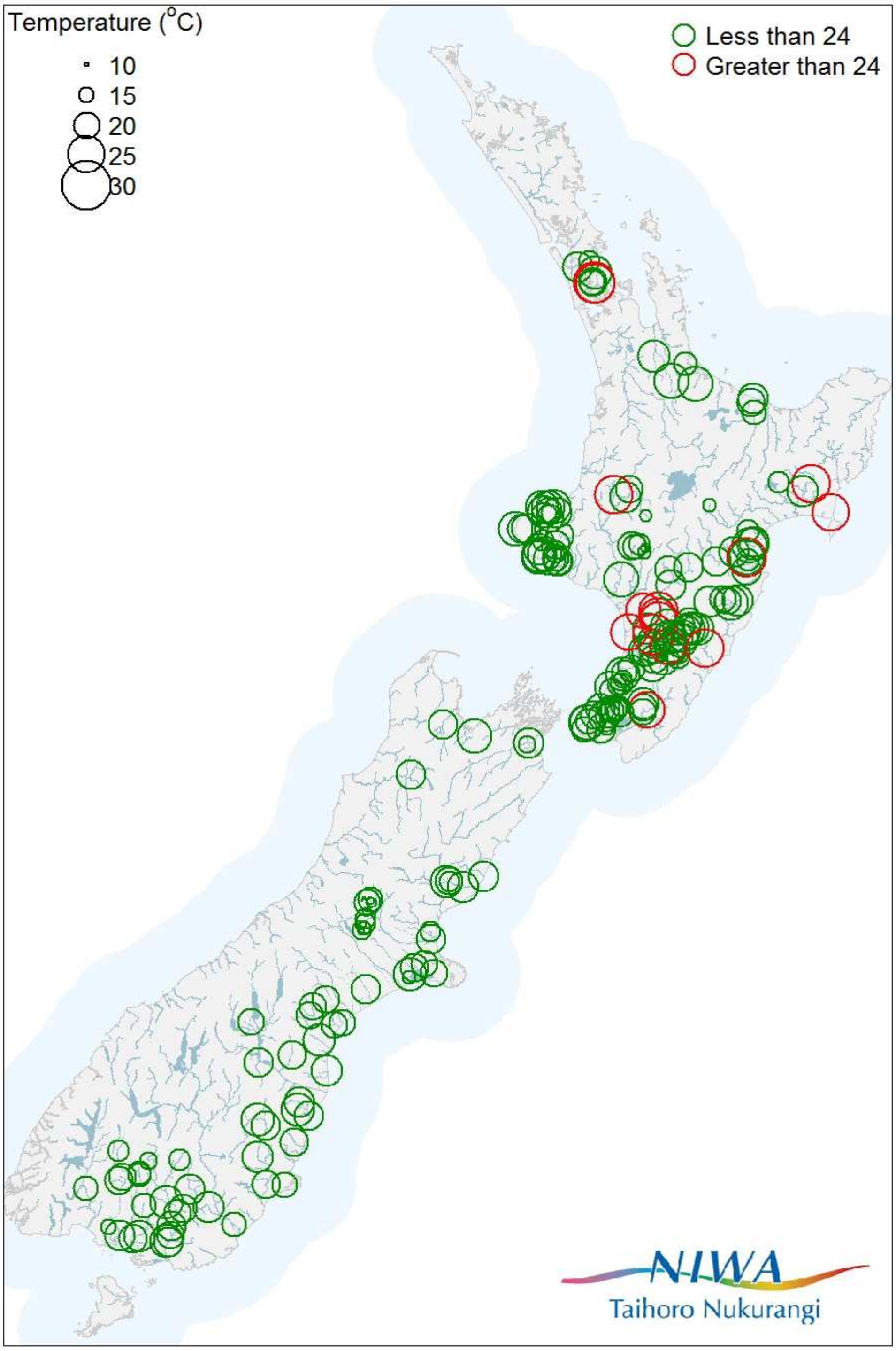

Figure 2-7: Bubble plot on the map of New Zealand indicates where different CRI values are exceeded for >5 days in temperature records. Red circles indicate those that exceed a CRI of ≥24°C for >5days. The map of CRI ≥25°C (Eastern Dry threshold) is very similar, since many sites that exceeded 24°C, also exceeded 25°C.

2.7 Mitigation of thermal stress

Much of NZ was originally heavily shaded with most of our land below the alpine tree line originally forested. Now stream shade has been substantially reduced (MfE 2007). It is a reasonable assumption that NZ stream ecology developed in the presence of heavy shade, such that removal of riparian trees during land clearance has exposed many of our running waters to considerable stress (Rutherford et al. 1999).

Obviously where thermal stress in streams arises primarily from discharge of heated wastewater, mitigation options that reduce the waste heat flux will usually be most appropriate. Approaches might include more complete wastewater treatment (cooling before discharge) or better diffuser design to achieve greater initial dilution and less of a thermal plume.

The most effective way of mitigating undesirably high temperature regimes in streams and (smaller) rivers, particularly where temperature issues arise from land clearance, is restoring shade by riparian plantings. Rutherford et al. (1999) scoped this issue in some depth as regards measurement of shade, effects of land use change, temperature effects on stream animals, and modelling of stream thermal response – including with restored shade. Small streams heat up much more quickly than large rivers when they emerge from shade, but also cool down more quickly when they flow back into shade. This can be thought of as expressing the 'thermal inertia' of larger rivers.

A modelling study by Davies-Colley et al. (2011) explored the recovery trajectories of shade and temperature, as well as other attributes, in streams of different size using models of riparian forest growth as drivers. Although small (shallow) streams suffer most from lack of shade, they are also much more easily and quickly shaded by growing riparian plantings as demonstrated in New Zealand long-term monitoring studies (Quinn et al. 2009, Quinn & Wright-Stow 2008).

The issue of thermal stress in poorly shaded streams is being considered overseas to counter the threat of global warming. For example, a campaign called "Keeping Rivers Cool" has been launched by the Environment Agency of England and Wales (www.environment-agency.govt.uk) to promote planting of riparian shade trees, and a similar campaign may be beneficial in New Zealand. Similarly a suite of actions including, lowering reservoirs, re-contouring streams to natural meandering patterns and re-establishing more natural in-stream flows so that river temperatures exhibit more natural diurnal and seasonal temperature regimes is being promoted in the Pacific Northwest (USEPA 2003a).

2.8 Conclusions – Temperature

Temperature is a fundamental state variable that strongly affects physico-chemical equilibria, chemical and biochemical reaction rates, and aquatic ecology (Franklin 2013). Protection of thermal regimes in running waters is clearly required to protect ecological integrity.

A national objective framework for temperature is proposed Table 2-2 to 2-4 built on a (narrative) gradient of increasing thermal stress from the 'no effect' A grade to significant loss of ecological integrity at D grade.

A comprehensive review of temperature criteria for New Zealand native fauna by Olsen et al. (2012) has provided the basis for proposing tentative temperature limits. An A/B ('no effect') threshold of 18ºC is proposed. A C/D ('bottom line') limit of 24ºC is proposed. We have allowed that the natural thermal regime of streams in eastern dry regions may be hotter than in more maritime climates, with a (nominal) 1ºC increase in absolute temperature limits in the former. Further refinement of (local) limits would require normalising to reference sites. Accordingly, we propose limits for temperature *increments* above reference stream high temperatures that may be used instead of absolute limits where appropriate data is available.

Mitigation of thermal effects is usually best done by riparian plantings to restore the heavy vegetation shade that characterised most New Zealand streams originally. This may, in any case, be necessary to adapt to the threat of global warming, and NZ could learn from the "Keeping Rivers Cool" campaign of the UK Environment Agency (www.environment-agency.govt.uk).

Management of temperature, DO and pH effects will usually mean continuous monitoring of these variables – which is fairly straight-forward for temperature, but more challenging for DO and pH.

3 Dissolved oxygen regime

3.1 Background

3.1.1 Why is dissolved oxygen important?

Oxygen is essential for almost all forms of life for respiration. Reduced dissolved oxygen levels (hypoxia) can impair the growth and/or reproduction of aquatic organisms and very low or zero dissolved oxygen levels (anoxia) will kill organisms. Consequently, the dissolved oxygen concentration of water is critical to stream ecosystem health.

3.1.2 What controls dissolved oxygen in water?

The main processes controlling dissolved oxygen levels in rivers (Figure 3-1) are well understood and widely described in the scientific literature (e.g., Chapra 1997, Cox 2003, Wilcock et al. 1998). The main controls are:

- Re-aeration: the transfer of atmospheric oxygen to water.
- Photosynthesis: plants and algae release oxygen into the water during photosynthesis.
- Respiration: plants and algae consume oxygen from the water during respiration.
- Biochemical oxygen demand (BOD): the amount of oxygen required by micro-organisms as they consume organic matter in the water.
- Sediment oxygen demand (SOD): the amount of oxygen required by micro-organisms as they consume organic matter in the sediments.

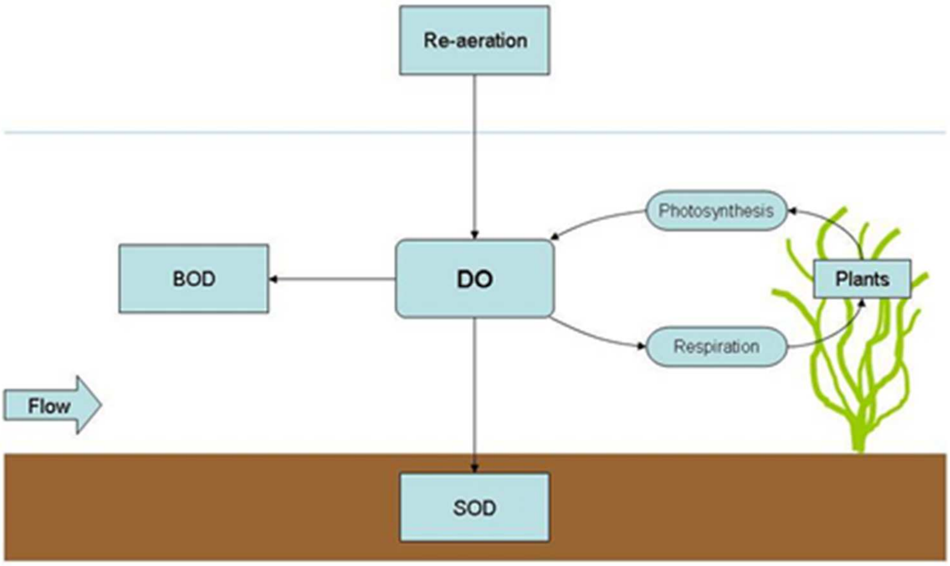

Figure 3-1: Schematic of the major processes influencing dissolved oxygen concentration in rivers. DO = dissolved oxygen; BOD = biochemical oxygen demand; SOD = sediment oxygen demand.

Re-aeration is affected by water temperature (colder water can hold more oxygen), mixing of the water and turbulence. This is the main factor controlling oxygen replenishment in water. In general, faster flowing, shallower, more turbulent water (e.g., in rapids or below waterfalls) has a higher re-aeration rate than slow flowing or still water (e.g., in pools). BOD or SOD are frequently associated with pollutant inputs, for example waste water outfalls. The rate at which biochemical oxidation occurs is typically proportional to the amount of organic matter remaining in the water (i.e., the less organic matter, the lower the BOD) leading to the classic dissolved oxygen sag curve (Figure 3-2).

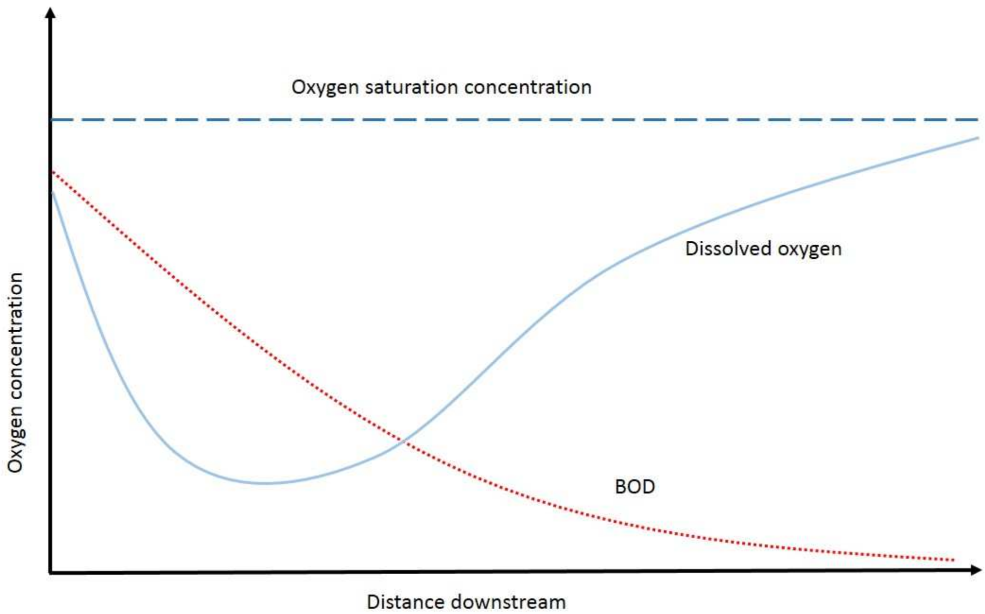

Figure 3-2: The dissolved oxygen sag curve. When BOD is high, the consumption of oxygen by micro-organisms is greater than re-aeration meaning dissolved oxygen concentration declines. As BOD declines with distance downstream, consumption rate of oxygen by micro-organisms declines proportionally until it is lower than the re-aeration rate. Dissolved oxygen concentrations in the stream then begin to recover.

Aquatic vegetation (algae and plants) can have a significant influence on dissolved oxygen levels in rivers. Oxygen is produced by photosynthesis during the day and consumed by respiration continuously. The combination of these two processes can impart significant seasonal and daily cycles in dissolved oxygen (e.g., Goodwin et al. 2008, Wilcock et al. 1998). Photosynthesis varies with light availability and vegetation biomass. Consequently, photosynthesis begins at dawn and ends at dusk and is greatest during the growing season. This means that during the growing season large variations in dissolved oxygen concentration are possible, with a maximum occurring in early afternoon when solar insolation is greatest, and a minimum occurring just before dawn following depletion by respiration overnight. The effects of plants on dissolved oxygen are illustrated in Figure 3-3 for two streams with different vegetation biomass. The magnitude of diel variation is significantly greater in the stream with a higher biomass of aquatic vegetation. More extreme variations have been observed in some lowland river systems (e.g., Wilcock et al. 1998).

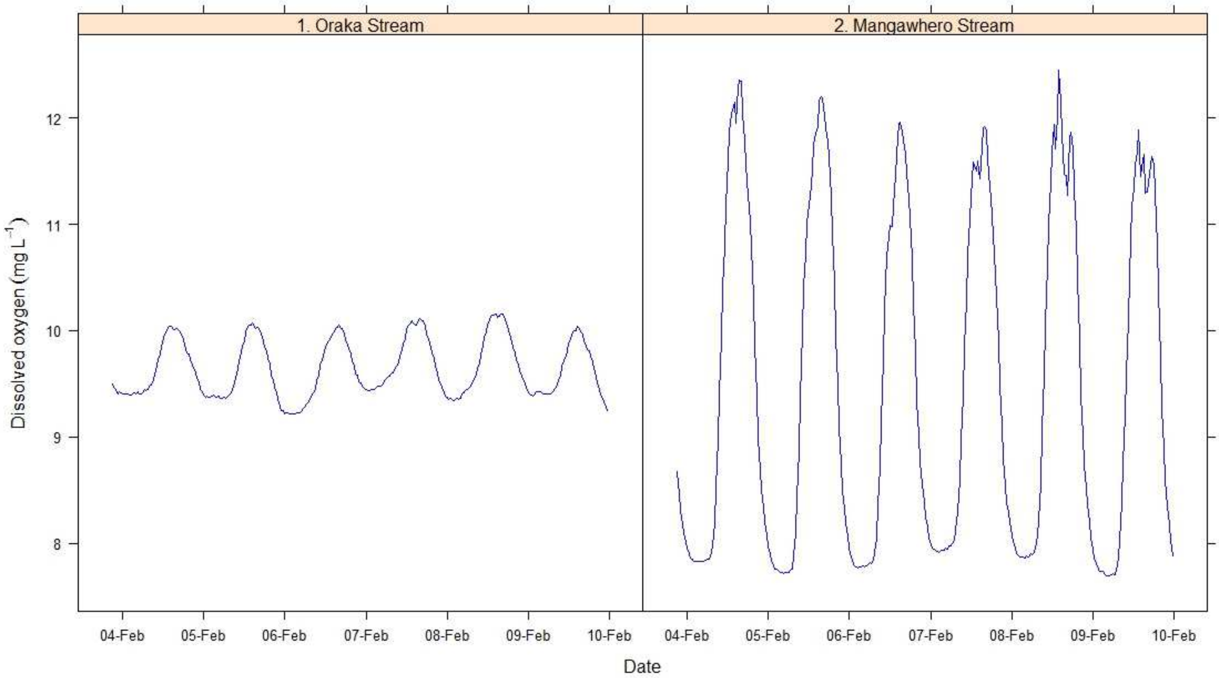

Figure 3-3: Temporal variations in dissolved oxygen concentrations in two contrasting stream, a stream with low vegetation biomass (left) and a stream with high vegetation biomass (right). Data were collected using continuous monitoring during the summer of 2012. The two streams are tributaries of the Waihou River, Waikato.

3.1.3 Where does low dissolved oxygen most commonly occur?

In New Zealand, low dissolved oxygen is most commonly encountered in warm, un-shaded, slow-flowing, lowland rivers where aquatic plants or algae are abundant. There is increasing evidence to suggest that in some areas, dissolved oxygen concentrations in these lowland streams and rivers are falling below the recognised lethal thresholds for some fish species (e.g., Wilcock & Nagels 2001, Wilding et al. 2012).

Other areas susceptible to low dissolved oxygen concentrations include below dams where water is released from the hypolimnion, areas of streams where upwelling of oxygen-depleted groundwater occurs, in estuarine river reaches associated with turbidity peaks (Mitchell et al. 1999, Wilding et al. 2012), and downstream of point sources of pollution with a high organic content. There is also evidence to suggest that dissolved oxygen concentrations decline with reduced flow in some streams (Figure 3-4). Conversely, low dissolved oxygen is likely to be less of a problem in higher gradient, cooler streams and rivers.

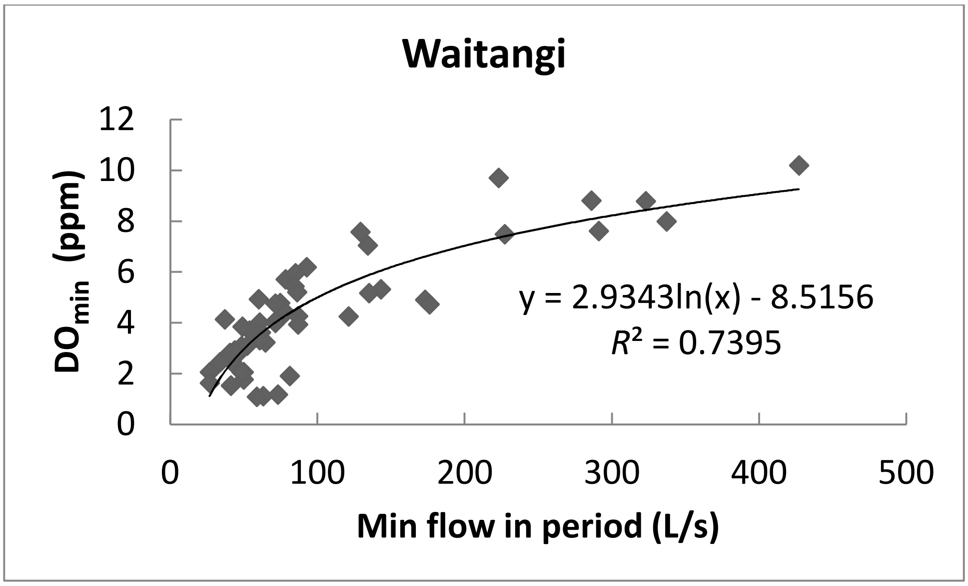

Figure 3-4: Decreasing dissolved oxygen concentrations (ppm; equivalent to mg L^{-1}) with decreasing instream flows (Wilcock, R.J. unpublished data). Data for weekly intervals in January-March and monthly intervals in April-December, during 2008-2011.

3.2 How much oxygen do aquatic organisms need?

Different aquatic organisms vary in their dissolved oxygen needs (Davis 1975, Dean & Richardson 1999, USEPA 1986a). Requirements vary both between species and across different life stages of the same species. The organisms most likely to be impacted by low dissolved oxygen in freshwater environments are fish and macro-invertebrates. The potential impact of low dissolved oxygen on these organisms will depend on their ability to tolerate low dissolved oxygen through physiological and behavioural adaptations, and the magnitude, duration, frequency and timing of low dissolved oxygen conditions. This will also vary under the influence of other environmental stressors, such as water temperature.

3.2.1 Oxygen requirements of fish

Dissolved oxygen is one of the most important environmental variables affecting the biology of fish (Alabaster & Lloyd 1982). During respiration fish, like other animals, take in oxygen and give out carbon dioxide. In most fish this is done using the gills, although some can also use the skin or have lung like structures used in addition to gills. When a fish respires, water is passed across the gills and oxygen diffuses into the blood through the gill filaments, subsequently being transported to the tissues in the bloodstream. Simultaneously, carbon dioxide in the bloodstream diffuses into the water and is carried away from the body. A reduction in external dissolved oxygen levels can result in a shortage of oxygen in the tissues and elicit physiological and behavioural responses to compensate (Kramer 1987). Typical responses may include a reduction in activity to reduce energy expenditure, increased ventilation of the gills, increased use of aquatic surface respiration (ASR), increased use of air breathing and vertical or horizontal habitat changes (Dean & Richardson 1999, Kramer 1987).

Reduced oxygen availability inevitably results in changes in fish activity due to the coupling between oxygen and energy budgets within organisms (Kramer 1987). If oxygen availability is reduced, then the energy allocated to breathing must be increased in order to maintain oxygen supply to the tissues. Alternatively, if the energy allocated to breathing is to remain constant, then the oxygen allocated to other energetic requirements must be reduced. Consequently, changes in both breathing and activity are likely under conditions of reduced oxygen availability.

The most frequently observed alteration in behaviour of fish following exposure to reduced dissolved oxygen levels is an increase in ventilation of the gills (Doudoroff & Shumway 1970, Kramer 1987, McNeil & Closs 2007, Urbina et al. 2011). This increases the flow rate of water across the gills in a bid to compensate for the reduced concentration of dissolved oxygen within the water. As the oxygen deficit increases, non-essential activity is often reduced in order to conserve energy. Feeding is often strongly affected because search, digestion and food assimilation are significant components of many fishes energy budget and thus are limited by oxygen availability (Doudoroff & Shumway 1970, Remen et al. 2012). Predator avoidance may also be altered as a result of differential tolerances to low dissolved oxygen, reduced swimming capability or enforced changes in habitat selection (Kramer 1987, Landman et al. 2005, Robb & Abrahams 2002, Roussel 2007). Another compensatory response displayed by fish is ASR. Diffusion of oxygen from the atmosphere into the water occurs at the air-water interface meaning that oxygen levels are elevated in the surface film. Under low dissolved oxygen levels, ASR utilises this thin layer of higher dissolved oxygen to help meet the oxygen demand of fish (Dean & Richardson 1999, Kramer 1987, McNeil & Closs 2007, Urbina et al. 2011). However, the use of ASR comes at a cost of increasing predation risk by being close to the surface.

As dissolved oxygen progressively decreases, the energetic costs of breathing will increase and eventually, where possible, fish are likely to move to habitats with a higher dissolved oxygen concentration (Miranda et al. 2000). Fish have frequently displayed a preference for locations with higher levels of dissolved oxygen (Doudoroff & Shumway 1970, Poulsen et al. 2011), and have shown avoidance of normally preferred locations in the presence of hypoxic water (Richardson et al. 2001). However, habitat shifts may have costs in terms of food availability, predation risk and less desirable physico-chemical conditions. If movement to a higher dissolved oxygen environment is not possible and low dissolved oxygen conditions persist, oxygen supply may be insufficient to meet the minimal energy demands of essential functions and fish will ultimately suffocate.

3.3 Dissolved oxygen tolerances of New Zealand fish species

A wide range of methods have been used to evaluate the dissolved oxygen tolerances of organisms. For example, some studies focus on establishing the time it takes for 50% of organisms to die whilst exposed to a constant dissolved oxygen concentration. Others impose a condition of progressively declining dissolved oxygen and identify the concentration at which 50% of individuals die. Some experiments focus on lethal effects and others on characterising sub-lethal impacts, such as on behaviour and growth. Some authors also describe results in terms of a prescribed effect level (e.g., 50% mortality), whilst others focus on identifying the incipient thresholds (i.e., the no effect level). This difference in methods employed complicates the interpretation of results from different studies (Franklin 2013).

Information regarding the dissolved oxygen tolerances of New Zealand's native fish species is relatively limited, particularly with respect to sub-lethal effects (Franklin 2013). Those studies that do exist have all used different approaches to assessing impacts making direct comparison of results difficult. Dean and Richardson (1999) assessed the tolerances of seven native New Zealand freshwater fish species and rainbow trout (*Oncorhynchus mykiss*) to low levels of dissolved oxygen by holding them in the laboratory at constant dissolved oxygen levels of 1, 3 and 5 mg L^{-1} for 48 hr at 15°C. Common smelt at both the juvenile and adult life stages, juvenile common bullies (*Gobiomorphus cotidianus*) and juvenile rainbow trout were found to be the most sensitive to low dissolved oxygen, with 50% mortality at dissolved oxygen levels of 1 mg L^{-1} occurring after 0.6-0.7 h, 0.6 h and 1 h respectively, and 100% mortality for all species within four hours. Juvenile banded kokopu (*Galaxias fasciatus*) were also relatively sensitive with 50% mortality at dissolved oxygen levels of 1 mg L^{-1} occurring in less than eight hours and 100% mortality by twelve hours. Juvenile torrentfish (*Cheimarrichthys fosteri*) showed no mortality for the first 24 hours of exposure, but 100% mortality by 48 h. Juvenile inanga (*Galaxias maculatus*) were shown to be more sensitive than the adult life stage, with 61% mortality after 48 h at 1 mg L^{-1} relative to 38% for the adult life stage. At a dissolved oxygen level of 3 mg L^{-1}, only juvenile trout responded, with fish moving to the surface to breathe indicating stress, but mortality was only 5% after 48 h. Shortfin (*Anguilla australis*) and longfin (*Anguilla dieffenbachii*) eels showed no response under the conditions tested.

Landman et al. (2005) tested the effects of constant low dissolved oxygen under laboratory conditions on a number of fish and invertebrate species by evaluating the dissolved oxygen concentration at which half of individuals died over 48 hours exposure at 15°C (48-h LC_{50}). Their experimental set-up also prevented aquatic surface respiration by blocking access to the water surface. They found that juvenile inanga was the most sensitive fish, with a 48-h LC_{50} at a concentration of around 2.6 mg L^{-1}. Common smelt and juvenile trout displayed similar thresholds with lethal concentrations of 1.8 mg L^{-1} and 1.6 mg L^{-1} respectively. Shortfin eel and common bully were the most tolerant species at this temperature with lethal thresholds of less than 1 mg L^{-1}.

The results of these two studies provide a good illustration of the effects of differences in experimental methodology and hence the need for caution when interpreting such results for management purposes. Landman et al. (2005) observed 50% mortality of adult inanga after 48 h at 2.6 mg L^{-1}. However, Dean and Richardson (1999) observed only 38% mortality after 48 h at a lower dissolved oxygen concentration of 1 mg L^{-1}. Urbina et al. (2011) demonstrated the importance of ASR in inanga exposed to low dissolved oxygen and observed the use of emersion as an avoidance strategy. The disparity in results between Landman et al. (2005) and Dean and Richardson (1999) therefore most likely reflects the importance of these strategies, which were prevented in the Landman et al. (2005) study, as a behavioural response for inanga to overcome low dissolved oxygen concentrations. The difference in results for smelt (50% mortality in 0.7 h at 1 mg L^{-1} (Dean and Richardson) compared to 50% mortality in 48 h at 1.8 mg L^{-1} (Landman et al. 2005)) suggests that ASR is less important as a coping strategy for this species. It also demonstrates that there is a very narrow threshold range over which the lethal effect of low dissolved oxygen is rapidly increased, an effect that has been observed in other species (Seager et al. 2000).

Another factor that is important to recognise about these studies is that in both cases fish were acclimated and trials carried out at a temperature of 15°C. This is significantly lower than summer water temperatures in some lowland streams of the North Island (e.g., Wilcock. & Nagels 2001, Wilcock et al. 1999). The metabolism and hence oxygen demand of fish varies with temperature. Consequently, oxygen tolerance thresholds have been shown to get higher with increasing temperature. Downing and Merkens (1957), for example, observed that lethal dissolved oxygen concentrations for a number of fish species increased by an average factor of 2.6 over a temperature range of 10°C to 20°C. Criteria based on the data presented by Landman et al. (2005) and Dean and Richardson (1999) are therefore likely to be under-protective at higher water temperatures.

Richardson et al. (2001) and Bannon and Ling (2003) considered sub-lethal effects of low dissolved oxygen on behaviour for some New Zealand fish species. Richardson et al. (2001) investigated the avoidance behaviour of smelt, inanga and common bully to low dissolved oxygen. In this study, the fish were acclimated and trials carried out at 20°C. Fish were placed in a fluvarium, one half of which was held at a dissolved oxygen of approximately 2 mg L^{-1} and the other at 8.5 mg L^{-1}, with free access between the two sides of the fluvarium. The behaviour of fish in response to the differences in dissolved oxygen was then observed over a 15 minute trial period. Only smelt displayed avoidance behaviour to the low dissolved oxygen water, with inanga showing no significant negative response and adult bullies displaying a significant preference for low dissolved oxygen. No explanation was suggested for the preference for low dissolved oxygen displayed by bullies; however, they have been shown to have quite a high tolerance to low dissolved oxygen levels (Dean & Richardson 1999).

Bannon and Ling (2003) explored the effects of low dissolved oxygen and temperature on sustained swimming capability of rainbow trout parr and inanga. Trials were carried out at 10°C, 15°C and 20°C under both normoxic (>96% saturation; c. 11.3, 10.1 and 9.1 mg L^{-1} respectively) and mildly hypoxic (75% saturation; c. 8.5, 7.6 and 6.8 mg L^{-1} respectively) conditions, with fish acclimated to the respective trial temperatures prior to testing. Maximum sustained swimming speed for trout parr occurred at 15°C under normoxic conditions, but decreased at lower and higher temperatures. Under conditions of mild hypoxia, no effect was observed at temperatures of 10°C and 15°C, but at 20°C a significant reduction in swimming capability was observed. Inanga juveniles also displayed temperature dependency of sustained swimming capability. Maximum sustained swimming speed was displayed between 15°C and 20°C under normoxic conditions. Under mild hypoxia no effect was observed at 10°C, but swimming capability was significantly reduced at 15°C and 20°C, with the optimal temperature reduced to between 10°C and 15°C and maximum swimming speed also reduced. The results of the Bannon and Ling (2003) study indicate that the influence of dissolved oxygen on sustained swimming speeds in these species varies with water temperature. It is unclear, however, whether this reflects increased metabolic demands for oxygen at higher temperatures, or whether it is an indication that the fish are responding to the concentration (≥7.6 mg L^{-1} for inanga; ≥6.8 mg L^{-1} for trout) of dissolved oxygen (which decreases with increasing temperature), rather than the percentage saturation (which remains constant) (Franklin 2013).

Urbina et al. (2011) investigated behavioural and physiological responses of inanga to exposure to progressive hypoxia. They observed significant changes in swimming activity as dissolved oxygen declined below 7.3 mg L^{-1}. The time that inanga spent performing ASR also increased progressively as dissolved oxygen concentrations declined from normoxia (9.7 mg L^{-1}), but only significantly so when a concentration of 1.9 mg L^{-1} was reached. At this level, fish spent an average of 16.4% of the time performing ASR and at 1.5 mg L^{-1}, this increased to 29.0% of the time (Urbina et al. 2011). Avoidance behaviour, defined as when inanga tried to jump out of the water, was only observed at the two lowest oxygen concentrations that were tested. On average, 70% and 94% of fish exhibited this behaviour at 1.9 and 1.5 mg L^{-1} respectively.

Relative to New Zealand's native fish species, much more is known about the effects of low dissolved oxygen on salmonid and other exotic cyprinid species present in New Zealand. Doudoroff and Shumway (1970), Davis (1975), Alabaster and Lloyd (1982) and USEPA (1986a) provide detailed reviews of the literature describing the effects on some of these species, particularly salmonids (i.e., trout and salmon). For salmonid species, research has been carried out on both acute (lethal) and chronic (sub-lethal) impacts of low dissolved oxygen across all life stages (eggs, larval, juvenile and adult). It has been shown that lower dissolved oxygen concentrations can retard egg development (Coble 1961, Côte et al. 2012, Ingendahl 2001, Malcolm et al. 2011, Shumway et al. 1964, Silver et al. 1963), reduce growth and alter behaviour of larvae and juvenile life stages (Jones 1952, Remen et al. 2012, Roussel 2007, Whitworth 1968) and impact on the growth, behaviour and habitat use of adults (Bushnell et al. 1984, Plumb & Blanchfield 2011, Poulsen et al. 2011)..

The exotic cyprinid fish species that have been introduced to New Zealand typically have a higher tolerance for low dissolved oxygen concentrations than those displayed by the salmonid species (Doudoroff & Shumway 1970, USEPA 1986a). Doudoroff and Shumway (1970) summarised the results of a range of studies indicating acute thresholds of <1 mg L^{-1} for species including goldfish, carp, tench (*Tinca tinca*), perch (*Perca fluviatilis*), and mosquito fish (*Gambusia affinis*), which are all present in New Zealand. Downing and Merkens (1957) reported 24-hr constant dissolved oxygen LC_{50} values for carp ranging from 0.4 mg L^{-1} at 10°C to 0.8 mg L^{-1} at 20°C. Equivalent thresholds for tench at 10°C were even lower at 0.2 mg L^{-1}. McNeil and Closs (2007), investigated the behavioural response of a number of fish species to progressive hypoxia and found that goldfish (*Carassius auratus*) and carp (*Cyprinus carpio*) were highly tolerant of hypoxia under laboratory conditions and that they may be able to survive in hypoxic (<1 mg L^{-1}) habitats for sustained periods of time through the use of ASR.

The USEPA (1986a) summarised the likely impact of differing dissolved oxygen levels on the production of salmonid species (Table 3-1). These figures are the basis of the dissolved oxygen criteria for freshwater ecosystems in the US and Canada (see Section 3.5 for further details). Dean and Richardson (1999) also suggested that the USEPA figures may be an appropriate basis for defining dissolved oxygen guidelines and standards in New Zealand based on their own results showing that the most sensitive of New Zealand's native fish species display similar acute tolerances to trout. Franklin (2013) has also proposed dissolved oxygen guidelines for freshwater fish in New Zealand partially founded on the USEPA figures in Table 3-2.

Table 3-1: Summary of how dissolved oxygen concentrations (mg L^{-1}) impact on production of salmonid species as described by the USEPA (1986). *Recommended water column concentrations to achieve the required inter-gravel concentrations shown in parentheses.

Degree of impairment	Early life stages	Other life stages
No production impairment	11.0* (8.0)	8.0
Slight production impairment	9.0* (6.0)	6.0
Moderate production impairment	8.0* (5.0)	5.0
Severe production impairment	7.0* (4.0)	4.0
Limit to avoid acute mortality	6.0* (3.0)	3.0

Alabaster and Lloyd (1982) suggested that where conditions are otherwise favourable, acute lethal effects on fish are likely to be avoided for most species by maintaining dissolved oxygen levels above 3 mg L^{-1} and that a minimum of 5 mg L^{-1} should be sufficient to satisfactorily support activities at most life-stages. However, in several studies initial responses have been detected at higher concentrations of 6-7 mg L^{-1} (Ingendahl 2001, Remen et al. 2012, Silver et al. 1963, Urbina et al. 2011). The severity of impact then increases as dissolved oxygen concentrations progressively fall below this threshold. Overall, the consequences of reduced dissolved oxygen concentrations is dependent on the influence of other stressors (e.g., temperature), and will vary between species and life stages. Typically, salmonid species are more sensitive and species such as carp and goldfish are more tolerant of low dissolved oxygen concentrations. It is likely that the tolerance of the more sensitive of New Zealand's native fish species is similar to that of salmonids, whilst the less sensitive species, e.g., eels, may be more similar to the cyprinid species.

A knowledge gap exists regarding the effects of varying dissolved oxygen concentrations, such as those observed in macrophyte and periphyton dominated streams, on fish (USEPA 1986a). As is the case for water temperature (Cox & Rutherford 2000b), exposure duration and frequency are likely to be important in determining the consequences of reduced dissolved oxygen on fish. Short-term, occasional exposure may be tolerated through changes in behaviour, however, as the duration and frequency of exposure increases, behavioural responses may no longer be sufficient to avoid negative impacts. One study by Seager et al. (2000) showed that for rainbow trout, for a given duration of exposure, there is a narrow threshold concentration range (<1 mg L^{-1}) above which mortality does not occur and below which mortality rapidly becomes high. This threshold increased as the exposure duration increased with, for example, mortality being significant at 1.6 mg L^{-1} for 6-h exposure, but not at 1.5 mg L^{-1} for 1-h exposure. Another area of uncertainty is over the impact of super-saturation of dissolved oxygen (i.e., dissolved oxygen >100% saturation). This can occur downstream of dams and where there is a high abundance of periphyton or macrophytes. A number of studies have identified negative effects of gas super-saturation below dams (Backman & Evans 2002, Lutz 1995), but in aquaculture super-saturation with oxygen is used to enhance productivity and a review by Dong et al. (2011) suggested that no adverse effects or abnormal behaviour is observed in fish when exposed to dissolved oxygen saturations up to 200%.

3.4 Oxygen requirements of macroinvertebrates

The highly diverse nature of macroinvertebrate species means that responses to low dissolved oxygen are extremely variable (Davis 1975, USEPA 1986a). Both the USEPA (1986a) and Davis (1975) cited studies of macroinvertebrate tolerances indicating lethal thresholds ranging from <1 mg L^{-1} to >8 mg L^{-1}. Some macroinvertebrate species therefore have a lower tolerance to reduced dissolved oxygen than fish. Davis (1975) suggested that broadly speaking, those species typical of high oxygen habitats, e.g., riffles, were more susceptible to low dissolved oxygen conditions. This is supported by the studies of Gaufin et al. (1974) who identified mayflies (typical of faster flowing streams) as being the most sensitive group of insects. The use of macroinvertebrate communities as an index of organic pollution (e.g., Stark & Maxted 2007) evolved from the differing sensitivities of species to the dissolved oxygen sag associated with inputs of organic pollutants with high BOD (Chapman 1996).

There have been very few studies of the dissolved oxygen tolerances of native New Zealand macroinvertebrate species. Dean and Richardson (1999) included shrimp (*Paratya curvirostris*) in their studies of dissolved oxygen tolerances and recorded mean mortality of 26.7% after 48 hours at dissolved oxygen concentrations of 1 mg L^{-1}. The first deaths were recorded after 2 hours. Landman et al. (2005) evaluated the 48 hour LC_{50} of *P. curviostris* (0.82 mg L^{-1}) and the freshwater crayfish (*Paranephrops planifrons*; 0.77 mg L^{-1}).

The high variability in tolerance between species and lack of data specific to New Zealand species makes defining dissolved oxygen limits for macroinvertebrates challenging.

3.5 Approaches to setting dissolved oxygen limits

3.5.1 US

US Environmental Protection Agency (USEPA)

The USEPA has proposed guidelines for dissolved oxygen based on the results of an extensive literature review (USEPA 1986a). Data on growth, development, reproduction and survival of fish, rather than solely incipient sub-lethal responses, were used to define production impairment levels at a range of dissolved oxygen concentrations (Table 3-1). These thresholds were then used as a basis for deriving dissolved oxygen limits (Table 3-2). The limits are 0.5 mg L^{-1} above their 'Slight Production Impairment' values to account for natural conditions and are represented as being protective of the more sensitive populations of freshwater organisms from impairment rather than being assured no-effect levels. In recognition of the natural daily fluctuations in dissolved oxygen and influence of exposure duration, dual-level (average and minimum) limits were defined across multiple durations (instantaneous, 7-day and 30-day). Where natural conditions do not meet 110% of the applicable means or minima, the minimum acceptable concentration was defined as 90% of the natural concentration.

Table 3-2: USEPA (1986a) water quality limits for dissolved oxygen (mg L^{-1}). Coldwater criteria. *These are water column concentrations recommended to achieve the required interstitial dissolved oxygen concentrations shown in parentheses.

Duration	Early life stages	Other life stages
30-day mean	-	6.5
7-day mean	9.5* (6.5)	-
7-day mean minimum	-	5.0
1-day minimum	8.0* (5.0)	4.0

3.5.2 Canada

National Research Council of Canada

Davis (1975) undertook a comprehensive review of dissolved oxygen tolerances and derived dissolved oxygen criteria for the National Research Council. His approach was to gather incipient oxygen response levels for primarily Canadian species and calculate mean thresholds for various fish groups. These thresholds were considered biological indicators of the onset of hypoxic stress and were designated as Level B in a three-tiered protection scheme. Levels A and C were derived by taking one standard deviation above and below the mean average threshold (i.e., Level B). Level A is close to full saturation in many cases and is meant to represent near-ideal conditions, Level B assumes some degree of stress and Level C may allow severe and widespread deleterious effects, especially if prolonged beyond a few hours (Davis 1975). For adult salmonids, Level A was 7.75 mg L^{-1}, Level B 6.0 mg L^{-1}, and Level C 4.25 mg L^{-1}. These values were defined as dissolved oxygen minima at each level of protection. There has been some criticism of the approach used to define Levels A and C, with these values considered unrepresentative of biological responses (e.g., USEPA 1986a).

Canadian Council of Ministers of the Environment (CCME)

The Canadian water quality guidelines are intended to provide protection of freshwater and marine life from anthropogenic stressors and build on the work carried out by Davis (1975). The guideline values are meant to protect all forms of aquatic life and all aspects of the aquatic life cycles, including the most sensitive life stage of the most sensitive species over the long-term (CCME 1999). The guidelines were designed to provide a science-based benchmark for a nationally consistent level of protection for aquatic life in Canada.

The Canadian water quality guidelines for the lowest acceptable dissolved oxygen concentrations (i.e., instantaneous minima) are 6 and 5.5 mg L^{-1} for the early and other life stages, respectively, in warm-water ecosystems, and 9.5 and 6.5 mg L^{-1} for the early and other life stages, respectively, in cold-water ecosystems (CCME 1999). The guidelines were derived from the USEPA's "slight production impairment" estimates (USEPA 1986), with an additional safety margin of 0.5 mg L^{-1} to estimate threshold dissolved oxygen concentrations. Where natural conditions alone create dissolved oxygen concentrations <110% of the guideline values, the minimum acceptable concentration is 90% of the natural concentrations. The guidelines also state that degradation of existing water quality is to be avoided.

3.5.3 Europe

European Inland Fisheries Advisory Committee (EIFAC)

The EIFAC proposed dissolved oxygen limits for European freshwater fish in 1973 (EIFAC 1973). That report highlighted the difficulties in defining robust criteria based primarily on laboratory experiments, with little validation in the natural environment. Criteria were based on percentile distributions to reflect the natural fluctuations in dissolved oxygen observed in rivers. For resident populations of moderately tolerant freshwater species, it was proposed that the annual 50th-percentile and 5th-percentile values should be greater than 5 and 2 mg L^{-1}, respectively, and for salmonids these percentiles should be 9 and 5 mg L^{-1}, respectively (EIFAC 1973). These values were intended to provide general guidance of suitable dissolved oxygen conditions, but recognised that under certain circumstances (e.g., for juvenile life stages or under higher water temperatures) higher thresholds may be required.

Water Framework Directive (WFD)

The Water Framework Directive (2000/60/EC) came into force on 22 December 2000 (see Section 2.3.3. for further information).

In the UK, dissolved oxygen standards were derived based on conditions statistically associated with “good” (biological) quality macroinvertebrate communities in thousands of surveyed sites (UKTAG 2008a). Standards are defined for two river types (Upland and low alkalinity; Lowland and high alkalinity) that broadly correspond with salmonid and cyprinid fish communities respectively. In contrast to other countries, standards are based on percentage saturation rather than concentration (Table 3-3).

Table 3-3: UK dissolved oxygen limits for freshwater. Standards are percent saturation and are measured against the 10th percentile of saturation (UKTAG 2008a).

Type	Status			
	High	Good	Moderate	Poor
Upland and low alkalinity (Salmonid)	80%	75%	64%	50%
Lowland and high alkalinity (Cyprinid)	70%	60%	54%	45%

In support of these standards, fundamental intermittent standards have also been derived to improve protection from intermittent discharges (Table 3-4; UKTAG (2013)). These standards refer to management of events of particular frequency and duration (UKTAG 2013). The standards for salmonid fisheries are presented in Table 3-4, but standards for cyprinid fisheries were also presented.

Table 3-4: UK fundamental intermittent standards for dissolved oxygen. Ecosystem suitable for a sustainable salmonid fishery. Dissolved oxygen concentration (mg L^{-1}).

Return period	Duration		
	1 hour	6 hours	24 hours
1 month	5.0	5.5	6.0
3 months	4.5	5.0	5.5
1 year	4.0	4.5	5.0

3.6 Dissolved oxygen limits for New Zealand

3.6.1 Essential characteristics of robust dissolved oxygen limits

Deriving dissolved oxygen limits for the maintenance of freshwater ecosystem health and general protection for indigenous species must account for the fact that different species and life stages vary in their tolerance and behavioural responses to low dissolved oxygen. Many species are capable of adapting their behaviour to compensate for short-term exposure to depressed dissolved oxygen levels. However, as the severity, duration and frequency of exposure increases, the costs in terms of energy expenditure and vulnerability to predation increase. Subsequently, changes in aquatic community structure and functioning become increasingly likely.

Critical considerations when deriving dissolved oxygen limits must therefore include:

- Lethal v. sub-lethal effects: Defining standards based solely on avoiding lethal effects will not be sufficient to protect ecosystem health. Sub-lethal impacts, for example on recruitment success or growth, may result in reduced abundance and eventual loss of more sensitive species and a shift in community composition to favour more tolerant (possibly exotic) species.
- Representative: Robust data must be available for a representative range of freshwater species (e.g., fish, macroinvertebrates, amphibians).
- Duration, magnitude and frequency of exposure: Limits must account for not only how low oxygen concentrations get, but also how often and for how long they are at that level. This includes accounting for natural fluctuations in dissolved oxygen.
- Multiple stressors: The impacts of low dissolved oxygen on aquatic organisms may be affected by the influence of additional stressors, e.g., water temperature, and therefore may require special consideration.

Ideally limits for dissolved oxygen to protect ecosystem health should be based on incipient concentrations where organisms first display a reaction, i.e., in growth, reproduction or avoidance in migratory species (CCME 1999, Davis 1975, Franklin 2013, USEPA 1986a). At this level, an organism must adjust its available energies to counteract the influence of oxygen starvation. When this stress is a chronic occurrence, it could have a detrimental effect on long-term survival. This, or any departure from a “no-effect” level, may therefore result in degradation of ecosystem health through changes in community structure and function.

3.6.2 NOF limits for dissolved oxygen

Table 3-5 sets out a proposed national objective framework (NOF) for dissolved oxygen in running waters. Narrative band descriptors are proposed in terms of stress on aquatic fauna caused by low dissolved oxygen concentrations. Tentative suggestions for numeric dissolved oxygen limits (band boundaries) are proposed.

Potential narrative descriptors

Consistent with other NOF attributes, Class A waters are considered to be of a standard equivalent to achieving reference or near-pristine status for New Zealand rivers. The expectation is that the dissolved oxygen regime at Class A sites will be sufficient to sustain intact aquatic communities and maintain ecological integrity and health to a high standard. Conversely, Class D waters are those in which the dissolved oxygen regime imposes significant and persistent stress on both the structure and functioning of aquatic ecosystems. There is a high likelihood that sensitive fish and macroinvertebrate species will be absent, whilst other species will be subject to chronic stresses. Ecosystem integrity and health is highly likely to be compromised in Class D waters.

The dissolved oxygen regime in Class B waters is proposed to maintain good ecosystem health. Aquatic communities should be similar to those expected in Class A, but may be subject to elevated stress caused by reduced dissolved oxygen for short periods of time (e.g., associated with diel fluctuations in dissolved oxygen caused by macrophytes or algae). There is a small risk to the long-term sustainability of some of the most sensitive species (particularly macroinvertebrates) unable to make the short-term behavioural responses to overcome exposure to temporary lower dissolved oxygen events.

The ecosystem health of Class C waters is likely to show indications of slight impairment as a result of the dissolved oxygen regime. This is likely to take the form of reduced populations or loss of sensitive fish and macroinvertebrate species caused by exposure to increasing duration of low dissolved oxygen under conditions of diel fluctuations or longer-term chronic impacts.

Potential numeric boundaries

Numeric band boundaries are proposed in Table 3-5, subject to peer review and discussion among New Zealand ecologists and experts on fish and invertebrate responses to stress from dissolved oxygen.

For simplicity, a single set of numeric NOF limits for dissolved oxygen have initially been proposed for application to all rivers and streams. It is acknowledged that natural variability in environmental characteristics means that these values may be slightly over-protective for some environments (e.g., lowland streams) whilst also being slightly under-protective for others (e.g., upland streams). The intention is that these values should provide a point of discussion for the expert groups to critically evaluate and modify where appropriate, rather than being set out as a definitive set of guidelines.

The proposed limits have been derived from expert analysis of the (somewhat limited) published data available on dissolved oxygen tolerances of New Zealand's native species, augmented by the international literature available on better studied species, such as trout and salmon. The rationale for each of the limits is described below, but it is acknowledged that other experts may choose to interpret the available data differently. A more empirical approach to deriving the limits is to be preferred and would result in more robust and transparent limits. The published data on New Zealand species is unsuitable for achieving this, but it is likely that there are unpublished data available for some New Zealand fish species that may be suitable for deriving more quantitative limits. Achieving that is outside the scope of this project, but may be beneficial.

A hierarchical format has been adopted for the proposed thresholds to account for the influence of both the magnitude and duration of exposure in determining the risk to ecosystem health. We decided this was more appropriate for an attribute with high natural diel variability than use of a single summary statistic. A similar tiered approach is used by the USEPA. Furthermore, specifying limits in the form of dissolved oxygen concentration (mg L^{-1}) was considered more appropriate than using saturation (as is currently used in the RMA). There is a generally good fit between dissolved oxygen concentration and ecological response thresholds in the existing literature. By defining a standard as a percentage of maximum saturation, the threshold dissolved oxygen concentration decreases as water temperature increases (i.e., 80% saturation at 10°C is 9.0 mg L^{-1} and at 25°C is 6.6 mg L^{-1}). This seems counter-intuitive for ecosystem protection purposes given that the oxygen demand of aquatic fauna generally increases with increasing temperature (Davis 1975, Downing & Merkens 1957)

Setting limits for daily dissolved oxygen minima primarily provides protection against short duration exposure to dissolved oxygen concentrations that exceed the acute mortality thresholds of sensitive aquatic species. The longer term (7-day) averages should avoid chronic impacts as a consequence of continuous or regularly occurring low dissolved oxygen events (Table 3-5).

The Class A 1-day threshold was based on 7 mg L^{-1} being observed as a threshold for initial behavioural responses in inanga (Bannon & Ling 2003, Urbina et al. 2011) and salmonids (Ingendahl 2001, Remen et al. 2012). The 7-day mean minimum threshold reflects the USEPA no impairment threshold for salmonids (8.0 mg L^{-1}; Table 3-1) which, based on the results of Dean and Richardson (1999), should also be sufficient for providing protection to New Zealand native fish species. The 7-day mean threshold was set at 9 mg L^{-1} to provide general protection for the most sensitive macroinvertebrate species. Setting the mean dissolved oxygen threshold for achieving Class A status at 9 mg L^{-1} means that water bodies with a mean water temperature of greater than 20°C will not achieve Class A status, because this exceeds the saturation capacity of the water. However, the proposed thermal threshold for Class A is 18°C (Table 2-2) meaning that if this is achieved, Class A status for dissolved oxygen is also achievable.

The 7-day mean threshold for Class B status is set at 8 mg L^{-1}. On average, this should minimise risk of impairment of fish populations and also provides a level of protection likely to protect all but the most sensitive of macroinvertebrate species. The 7-day mean minimum threshold is set at 7 mg L^{-1}, again based on minimising risk of chronic impacts on fish populations. However, the 1-day threshold has been set lower for Class B at 5 mg L^{-1}. The 5 mg L^{-1} threshold was suggested by Alabaster and Lloyd (1982) as providing general protection from any greater than moderate chronic effects in most fish communities. It is also the threshold for moderate impairment proposed by (USEPA 1986a; Table 2-1). In combination with the 7-day mean minimum threshold, this should provide reasonable protection for most organisms for most of the time, while allowing for occasional, short duration low dissolved oxygen excursions. However, it is possible that some of the most sensitive species, particularly macroinvertebrates, may not be fully protected such that small shifts in community composition could occur.

The thresholds for Class C are set at a level which will likely lead to some degree of impairment of ecosystem health in this class. Largely as a consequence of increasing frequency and duration of exposure to dissolved oxygen concentrations below the thresholds for chronic impacts, there is elevated risk of sensitive species of both fish and macroinvertebrates being lost. The 4 mg L^{-1} 1-day minimum threshold is set as a conservative protection level to avoid acute limits for sensitive fish species being exceeded. Most fish species are able to tolerate short duration exposure to these concentrations through behavioural adaptations (e.g., moving to higher dissolved oxygen habitats). However, longer term, repeated exposure to these levels is likely to lead to chronic impacts for sensitive species of fish. As this limit is approached, there is therefore increasing risk of a shift in community composition towards more tolerant, generalist species, including exotic cyprinid species. The 7-day mean minima is set at 5 mg L^{-1}, which is generally considered sufficient to avoid significant chronic impacts on most fish species. However, it is likely that some sensitive species and life stages may be impaired by regular and extended exposure to these concentrations. The 7-day mean threshold is set just above the slight impairment threshold suggested by the USEPA (1986a). The 6.5 mg L^{-1} limit has been used in both the USA and Canada as the main threshold for avoiding significant impacts on fish communities. It is likely to provide similar protection to most indigenous fish species in New Zealand, but we are uncertain whether this is sufficiently protective of sensitive macroinvertebrate species.

When dissolved oxygen concentrations fall below the thresholds for Class C and into Class D, there is a risk of significant degradation of ecosystem health. As dissolved oxygen concentrations drop, they will exceed both the chronic and eventually the acute thresholds for an increasing number of species, and ecosystem structure and function will become increasingly compromised.

Table 3-5: Proposed NOF for dissolved oxygen regime in rivers and streams. The term regime refers to the diel fluctuation of dissolved oxygen around the daily mean. Proposed thresholds are tentative suggestions for class boundaries.

Value (use)		Ecological Health					
Attribute		**Dissolved oxygen regime**					
Environment		Rivers					
Measurement unit		Milligrams per litre (mg L^{-1})					
Summary statistic		Summer monitoring data for discrete specified periods. All 3 statistics must be met for each band.					
Band descriptors (narrative – what will people notice as the impact on the value)	A	No stress caused by low dissolved oxygen on any aquatic organisms that are present at matched reference (near-pristine) sites.					
	B	Occasional minor stress on sensitive organisms caused by short periods (a few hours each day) of lower dissolved oxygen. Risk of reduced abundance of sensitive fish and macroinvertebrate species.					
	C	Moderate stress on a number of aquatic organisms caused by dissolved oxygen levels exceeding preference levels for periods of several hours each day. Risk of sensitive fish and macroinvertebrate species being lost.					
	D (unacceptable/doesn't provide for value)	Significant, persistent stress on a range of aquatic organisms caused by dissolved oxygen exceeding tolerance levels. Likelihood of local extinctions of keystone species and loss of ecological integrity.					
Band boundaries (numeric)	A/B	7-day mean[a]	≥9.0	7-day mean minimum	≥8.0	1-day minimum	≥7.5[b]
	B/C[c]	7-day mean[a]	≥8.0	7-day mean minimum	≥7.0	1-day minimum	≥5.0
	C/D	7-day mean[a]	≥6.5	7-day mean minimum	≥5.0	1-day minimum	≥4.0
	D (unacceptable/doesn't provide for value)	7-day mean[a]	<6.5	7-day mean minimum	<5.0	1-day minimum	<4.0
Are there circumstances where a water body could naturally fall into the D band?		▪ Geothermally-influenced waters. ▪ Groundwater dominated streams with a large input of low dissolved oxygen groundwater; this can also occur at baseflow conditions in rivers not normally dominated by groundwater. ▪ Sites with a naturally high abundance of macrophytes or periphyton where large diel variations fall below the 4.0 mg L^{-1} 1 day minimum threshold.					
Limitations/gaps/risks		There are limited data available on the dissolved oxygen tolerances of native NZ species, particularly macroinvertebrates. However, those data are complemented by more detailed information on international species "surrogates" including chinook salmon and rainbow trout, which are important recreational fish species in New Zealand. Together these provide a robust basis for establishing thresholds. We note that (as for international guidelines) these were not derived using a rigorous species tolerance approach for resident species. The effects of diel variability in dissolved oxygen are poorly understood and therefore may not be accounted for effectively.					
Notes		[a] 7-day duration alone is insufficient to avoid chronic impacts. It is intended that in any continuous 7-day period throughout the year, this threshold will be met i.e., this is the annual minimum 7-day mean. [b] This corresponds to the 80% dissolved oxygen saturation value at the thermal threshold for achieving Class A (18°C). [c] If the 95^{th} percentile temperature is >24°C, this becomes the threshold for Class D due to the significant interactive effects of high temperature and dissolved oxygen.					
References, supporting documentation S1 links		(Dean & Richardson 1999, Franklin 2013, Landman et al. 2005, USEPA 1986a)					

3.6.3 Confidence in proposed NOF thresholds for DO

There are limited data available on the dissolved oxygen tolerances of native NZ species, particularly macroinvertebrates. Most information is available for inanga (mix of chronic and acute), with limited acute data available for six further native fish and two macroinvertebrate species. Considerable international literature is available on salmonid species. Insufficient published data are available to derive numerical boundaries using a rigorous species tolerance approach for resident species. Expert judgment has been used to benchmark the limited data for NZ native species against the international literature and guidelines for better studied salmonid species allowing for derivation of numerical limits. Greatest uncertainty is associated with the poor understanding of the effects of diel variability in dissolved oxygen on organisms. As the status of a waterbody declines from Class A, the fish species at greatest risk are most likely to be smelt and inanga, followed by other galaxiid fish species (e.g., kokopu) (Dean & Richardson 1999, Landman et al. 2005). Trout and salmon are also sensitive to lowered dissolved oxygen, particularly in the early life stages. We note that the early life stages of the majority of native fish species are poorly understood and no data are available on their dissolved oxygen tolerances. Data on the early life stages of salmonid species have been used as a surrogate (and there is no reason to exclude them in the absence of native species data), but if the early life stages of native species are more sensitive, more conservative limits would be recommended. Data on macroinvertebrates are poor, but it is likely that ephemeroptera, plecoptera and trichoptera taxa (particularly those with a high MCI score) are most at risk in degraded water bodies.

In summary, there is adequate information about dissolved oxygen to establish thresholds sufficiently robust for regulatory purposes. However, we recommend research on sensitive life stages of native fish species and a wider range of macroinvertebrates to ensure the thresholds are sufficiently protective.

3.6.4 Evaluating compliance & current state

The proposed thresholds should be achieved most of the time in New Zealand, but we recognise that natural variation may result in some deviation beyond these limits. It may be appropriate, therefore, to determine a suitable summary statistic, e.g., the 5th percentile value, to use for assessing conditions against the thresholds. The most critical time is likely to be under summer low flows when water temperatures are high. Consequently, evaluation of compliance should focus on these conditions. It is important that compliance be assessed through the use of continuous monitoring of dissolved oxygen. The significant diel variations that can occur in dissolved oxygen concentrations mean that one-off spot measurements are highly dependent on the time of sampling. More details are provided on continuous monitoring of dissolved oxygen in the most recent National Environmental Monitoring Standards (NEMS) DO report (Wilcock et al. 2013 *in press*).

The need for continuous monitoring data to evaluate compliance against the thresholds means that there are limited data available for assessing the current state of rivers at a national scale. Reliable continuous monitoring electrodes for dissolved oxygen only became widely available in New Zealand around 2006. Modern sondes using optode technology are increasingly being adopted by water monitoring agencies and have significantly improved understanding of dissolved oxygen regimes in New Zealand rivers (Wilcock et al. 2013 *in press*). However, due to the investment required for widespread deployment of the optode sondes many agencies still only use the continuous monitoring electrodes for short-term compliance monitoring. Measurement of dissolved oxygen at State of the Environment (SOE) monitoring sites is still primarily undertaken as monthly spot samples. If a tentative assumption were to be made that the spot samples are roughly indicative of mean dissolved oxygen concentrations at a site, these values could be assessed against the 7-day mean thresholds to give an estimate of potential NOF class. However, due to the importance of dissolved oxygen minima in determining NOF class, using only the mean is likely to result in positive bias, with sites more likely to be classified higher than they really are. This is particularly likely for macrophyte- or algal-dominated streams with large diel fluctuations in dissolved oxygen.

It is likely that at a regional scale, suitable continuous monitoring data will be available for some sites where data have been collected as part of targeted short to medium-term investigations of water quality (e.g., Auckland, Waikato, Hawke's Bay, Wellington, Manawatu, Southland and others?). There may also be a small number of sites where longer-term deployments have been made (e.g., Auckland). However, a bias in these data may be expected toward lowland, impacted sites where dissolved oxygen is perceived to be a problem.

3.6.5 Knowledge gaps

Future research to support the derivation of more robust, empirical criteria (e.g., Vaquer-Sunyer & Duarte 2008) for New Zealand should focus on clarifying both acute and chronic incipient thresholds (i.e., no effect thresholds) for New Zealand species and how these vary under differing exposure regimes. Understanding how tolerances vary with increasing duration of constant exposure and under the influence of cyclical dissolved oxygen regimes representative of natural diel variations would be extremely beneficial. It would also be valuable to investigate the influence of temperature and other potential interacting stressors on responses to different dissolved oxygen concentrations.

The early life stages of the majority of native fish species are poorly understood and no data are available on their dissolved oxygen tolerances. Study of dissolved oxygen tolerances of a wider range of native macroinvertebrates is also recommended particularly for ephemeroptera, plecoptera and trichoptera (EPT) taxa. There is a particular need for identifying thresholds for macroinvertebrates, both in terms of lethality and behavioural effects that are likely to increase predatory and competitive pressures (e.g., moving to the upper surfaces of rocks or to higher current areas of streams).

An area currently lacking consensus is the effect of high dissolved oxygen concentrations (>100% saturation) on fish (and other organisms) and hence the need for maximum limits. Internationally, there have been reported incidences of mortality in fish caused by gas bubble trauma under conditions of gas super-saturation below dams (Lutz 1995). However, this effect seems primarily to be driven by total dissolved gas rather than high dissolved oxygen. Elsewhere, investigations into the effects of oxygen super-saturation in aquaculture suggest that no adverse effects or abnormal behaviour is observed in fish when exposed to dissolved oxygen saturations up to 200% (Dong et al. 2011).

4 pH

4.1 Background - What is pH?

The variable pH is a measure of the concentration of hydrogen ions in water (where p stands for – log base 10). The formal definition of pH is given by Eq. 1.

$$pH = -\log_{10} a_{H^+} = -(\log_{10} H^+ + \log_{10} \gamma_{H^+}) \qquad (1)$$

The terms a_{H^+} and γ_{H^+} are the hydrogen ion activity and activity coefficient, respectively

Being a base-10 logarithmic scale, each unit change in pH corresponds to a 10-fold change in hydrogen ion concentration. The pH scale is 0–14 and on average, freshwaters are slightly alkaline (≈7.5) because of buffering by bicarbonate (Butler 1991). Distilled water is slightly acidic (pH 5) because it is saturated with atmospheric CO_2 that forms carbonic acid, which dissociates weakly to give H^+ and HCO_3^- ions, as follows (Eq. 2)

$$CO_2 + H_2O \rightleftharpoons H_2CO_3 \rightleftharpoons H^+ + HCO_3^- \qquad (2)$$

The geology and source of a water often determine its pH, whereas the alkalinity (acid neutralising capacity) defines its resilience to pH change. Waters with a high alkalinity (at least 150 mg L^{-1} as $CaCO_3$ equivalent) are strongly buffered and will have narrow pH ranges. Waters draining limestone have a high concentration of carbonate and have relatively unvarying pH, whereas rivers deriving from glacial meltwater or from rainwater have low alkalinities (less than 20 mg L^{-1} as $CaCO_3$) and often exhibit pH variations of several units because of low concentration of buffering ions that can take up and release hydrogen ions.

During periods of base flow, in many streams with abundant plant biomass (macrophytes, phytoplankton or periphyton), strong diel cycles of dissolved oxygen and pH regularly coincide with the uptake and release of oxygen and carbon dioxide by the plants (Figure 4-1). Maxima in pH and dissolved oxygen occur in the late afternoon when photosynthesis dominates whereas minima occur in early morning when respiration is dominant (Wilcock and Chapra 2005). Late summer pH values of 9-10 are observed in cobble-bed rivers with extensive periphyton mats, such as the Tukituki River.

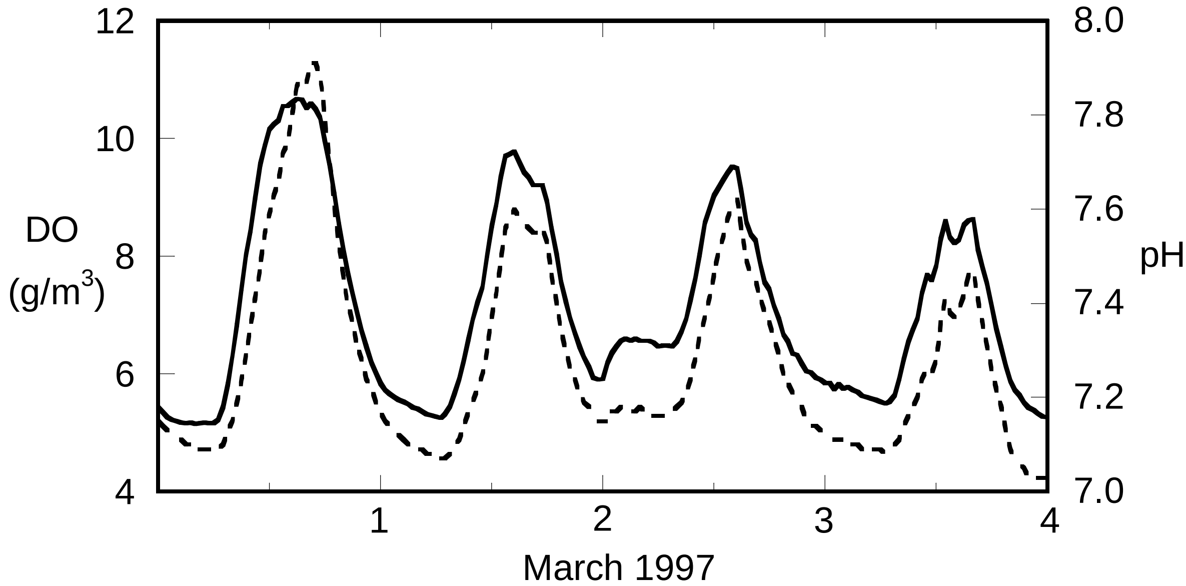

Figure 4-1: Co-variation of dissolved oxygen (continuous line) and pH (dotted line) in a eutrophic stream during summer (from Davies-Colley and Wilcock 2004).

pH is an important ‘master variable’ because it affects other variables. Indeed the NEMaR project (e.g., Davies-Colley et al. 2012a&b) refer to pH as a “supporting” variable – in the sense of one needed to interpret other variables. Ammonia toxicity is affected by pH because the proportion of the toxic un-ionised form, NH_3, of total ammoniacal nitrogen is primarily controlled by pH and temperature. Many point source discharges of dairy effluent and wastewater from community waste-treatment plants have elevated ammoniacal-N concentrations that, coupled with high pH values in late afternoon, may cause ammonia toxicity to stream life if not diluted sufficiently. Diel pH changes affect factors such as arsenate speciation and toxicity from acid mine discharges, the toxicity of metals by reducing the degree of complex formation and increasing concentrations of free ions, and the toxicity of some organic acids, e.g., pentachlorophenol from old timber treatment sites (Wilcock and Chapra 1999).

The lack of routine diel monitoring data in New Zealand means that the spot measurements that have been made, usually between 9:00 am and 4:00 pm, are of limited value and generally describe a narrow range that may capture afternoon maxima but not minimum values that occur near sunrise (Figure 4-1). Diel pH ranges of 1.5-2 are commonly observed in summer in rivers and streams with high plant or algal biomasses, whereas daytime data may only vary by ± 0.2 units (Wilcock and Chapra 1999).

4.2 Effects of pH on aquatic ecosystems

In Europe and North America, acid rain has caused toxicity problems in freshwaters associated with increased concentrations of the free aluminium ion, Al^{3+}. Freshwater species have varying degrees of tolerance to waters with low pH (Figure 4-2). Generally, the young of most species are more sensitive to environmental conditions than adults. At pH 5, most fish eggs cannot hatch and at lower pH levels, some adult fish die. Although acid rain is not a significant problem in New Zealand, naturally occurring acid streams such as those draining peatland can have pH values as low as 4 (Winterbourn and Collier 1987; Collier et al. 1990).

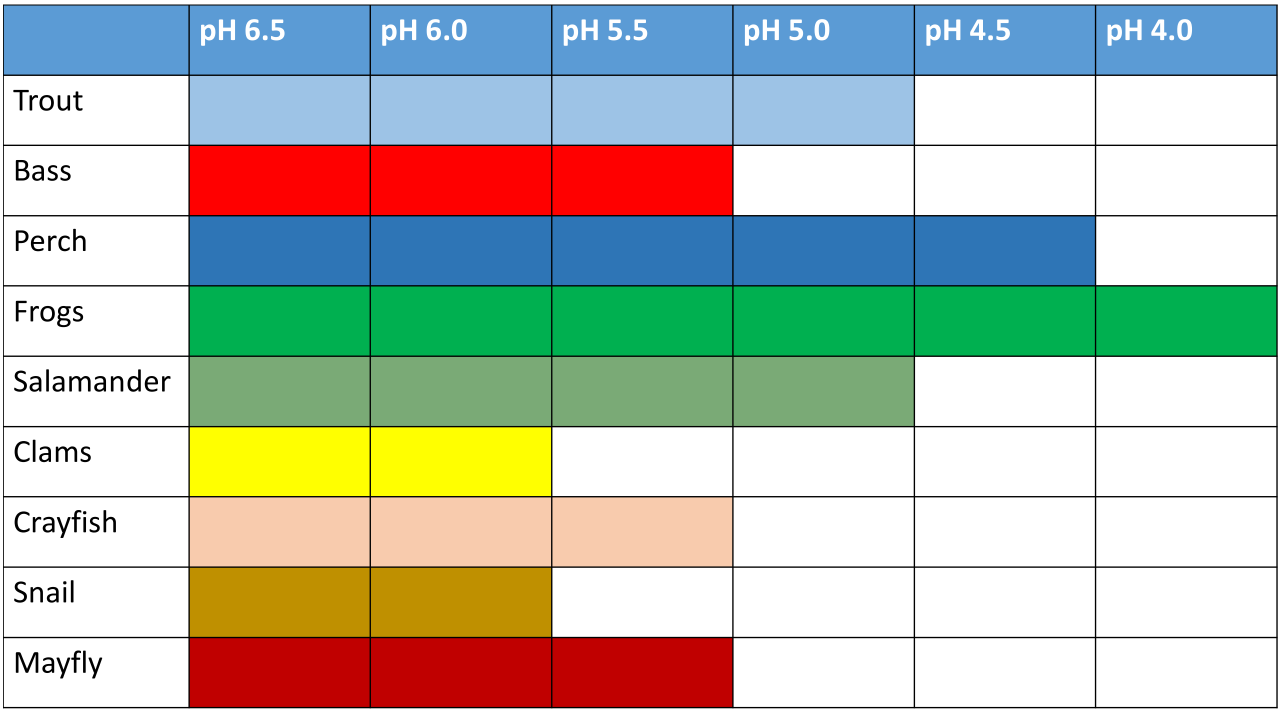

Figure 4-2: Varying tolerances of North American freshwater species to low pH (modified from USEPA)[3].

4.3 Responses of New Zealand species to pH

Responses of nine New Zealand fish species, and a native shrimp, to pH values in the range 3.2-11.2 were measured in a controlled laboratory pH-gradient (West et al. 1997). All species tested exhibited pH preferences and all but inanga avoided pH above 9.5. Adult fish showed stronger preferences than juveniles and an avoidance of pH below 6.5 was evident for most species except shortfinned elvers, koaro and banded kokopu (Figure 4-3). The study concluded that the range of pH encountered in New Zealand lowland streams was unlikely to have a major impact on the distribution of most native freshwater fish (West et al. 1997).

[3] http://www.epa.gov/acidrain/effects/surface_water.html

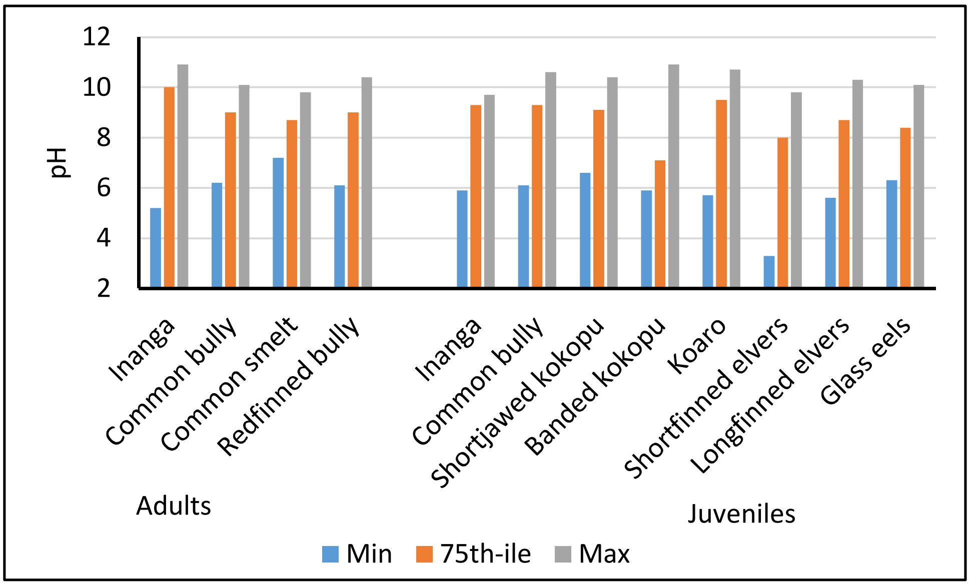

Figure 4-3: pH preferences of some native New Zealand fish at 20°C (West et al. 1997).

Field observations of native fish species in Westland humic-stained streams with naturally low pH (often around 4) showed that 9 out of 14 native fish species were found in waters with pH below 5, and 7 species were present in waters with pH < 4.5. Furthermore, 34 of the 37 most widespread aquatic insect taxa were recorded in Westland streams with pH < 5, and 24 were taken from sites with pH < 4.5 (Figure 4-4) (Collier et al. 1990). Furthermore, “taxonomic richness was not correlated with pH and similar numbers of ephemeropteran, plecopteran and trichopteran taxa were taken from acidobiontic (pH ⩽ 5.5), acidophilic (pH 5.6–6.9) and moderately alkaline (pH ⩾ 7.0) groups of streams. Many species occurred over a wide pH range and had a lower limit of about pH 4.5” (Winterbourn and Collier 1987).

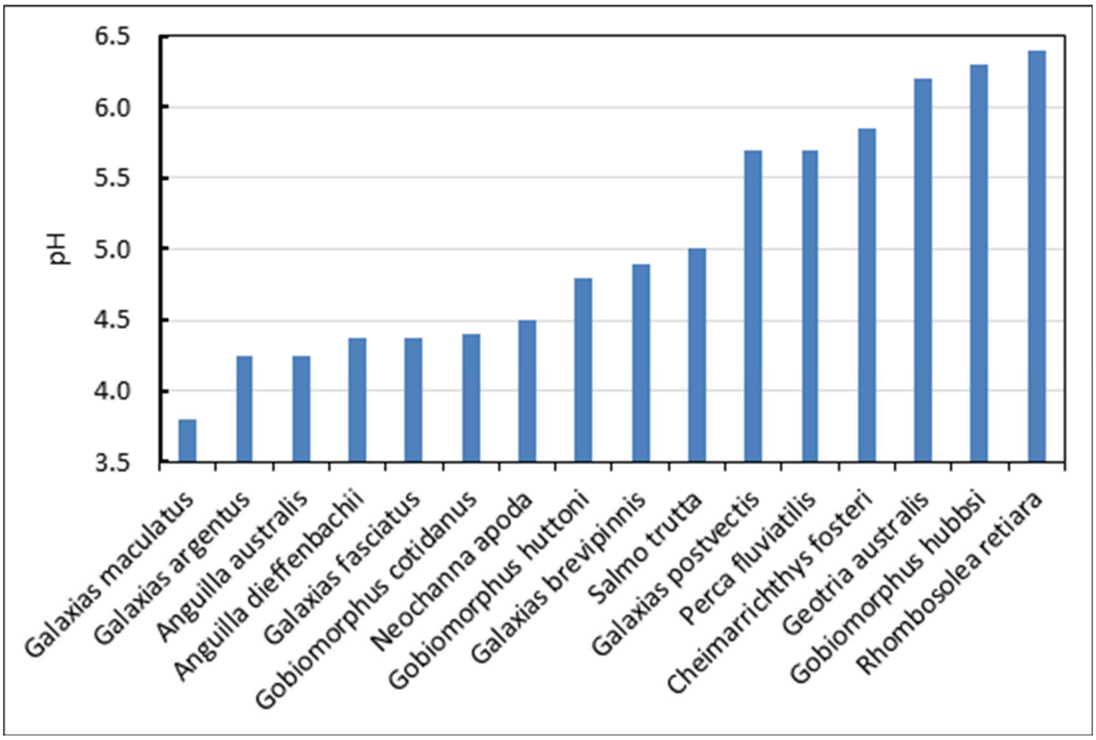

Figure 4-4: Lower pH ranges of fishes in Westland waters during 1984-86 (adapted from Collier et al. 1990).

4.3.1 The special case of ammonia

The indirect effects of pH (e.g., on ammonia toxicity) are generally much greater than pH *per se*. Ammoniacal-N is a relatively common pollutant in New Zealand waters that is sometimes in environmentally harmful concentrations immediately downstream of point source discharges of wastewater, or in sites affected by surface runoff from overloaded effluent land-irrigation schemes. Ammoniacal-N exists in two forms: un-ionised NH_3 that is toxic to aquatic organisms at low concentrations, and the ammonium ion NH_4^+ that is relatively non-toxic and is usually the dominant form (Eqs. 3-5) in waters with pH 7-8. The proportion of total ammoniacal-N $(NH_4 = NH_3 + NH_4^+)$ in the toxic, un-ionised form (NH_3) is affected by pH and temperature, as follows (USEPA 1999) (Figure 4-5).

$$NH_4^+ \leftrightharpoons NH_3 + H^+ \quad (3)$$

$$K_d = \frac{[NH_3][H^+]}{[NH_4^+]}$$ (K_d is the dissociation constant and varies with temperature) (4)

$$f_{NH_3} = \frac{1}{1+10^{pK_d - pH}} \quad (5)$$

(f_{NH_3} is the fraction of total ammonia that is NH_3 and pK_d = -$\log_{10}K_d$)

Regarding terminology for ammonia, which has changed over the years, where possible we indicate total ammoniacal-N (as mg NH_4-N/L), and indicate concentrations of the more toxic un-ionised ammonia as mg NH_3-N/L. Some of the earlier New Zealand research (1990's) on ammonia followed the then USEPA convention of reporting concentrations of un-ionised ammonia as mg NH_3/L, so for comparative purposes we have provided equivalent concentrations of mg NH_4-N/L. More recently the USEPA has adopted the convention of reporting the concentration of total ammoniacal-N (mg NH_4-N/L) and the pH at which it was measured (USEPA 2009). This is considered the most useful way to report ammonia concentrations because while un-ionised ammonia is the most toxic form, the ammonium ion also contributes to toxicity for some species under certain pH and temperature conditions (see Table 8.3.7. ANZECC 2000).

Toxic concentrations of un-ionised ammonia (NH_3) were assessed for nine native New Zealand freshwater invertebrate species (Hickey and Vickers 1994). The 96-h EC_{50}[4] values at 15°C ranged from 0.18 mg NH_3/L (pH 7.6; 14 mg NH_4-N/L) for the crustacean, *Paracalliope fluviatilis* to >0.8 mg NH_3/L (pH 8.2; 16 mg NH_4-N/L) for the shrimp *Paratya curvirostris* respectively. Richardson (1997) determined the acute toxicity of NH_3 to eight New Zealand indigenous freshwater species, including native fish and reported 96-h LC_{50}[5] values from 0.75 mg NH_3/L (pH 8.1, 15°C; 18 mg NH_4-N/L) to 2.35 mg NH_3/L (pH 7.5, 15°C; 226 mg NH_4-N/L). The most sensitive indigenous species tested to date, is the larvae (glochidia) of the native freshwater mussel *Echyridella menziesii* with a 48 h EC_{50} of 9.4 mg NH_4-N/L (pH 8.0) (Clearwater et al. 2013 *in review*). Comparison of *E. menziesii* data with previous studies on North American freshwater mussels indicates that juvenile mussels would not be adequately protected by current ANZECC water quality guidelines for chronic exposure to ammonia (Wang et al. 2007a and b, 2008, 2011, USEPA 2013).

To take account of the rapid increase in percentage of un-ionised ammonia at increasing pH, guideline values are provided for different pH. For example, the ANZECC & ARMCANZ (2000) toxicant trigger value for total NH_4-N for the 95% level of protection of freshwater species is 0.90 mg NH_4-N/L at pH 8, and 0.18 mg NH_4-N/L at pH 9.0 (see Table 8.3.7. ANZECC 2000). Temperature can also be taken into account (see ANZECC 2000 for guidance and conversion tables).

Thus, for water bodies where total ammoniacal-N concentrations are elevated to near 0.9 mg total NH_4-N/L an upper pH limit of 8 is recommended. At higher NH_4 concentrations pH maximum values should be calculated to maintain non-toxic levels of un-ionised NH_3 (USEPA 1986, 2009, ANZECC 2000). An upper pH limit of 9 is acceptable for waters with total ammonia concentrations no greater than 0.20 mg total NH_4-N/L at 18-22°C. Under these conditions, un-ionised ammonia concentrations are about 0.095 mg NH_3-N/L. The USEPA (1986) chronic ammonia exposure criterion for fresh water fish with early life stages present, is a 30-day average concentration of 0.3-0.4 mg NH_4-N/L that is not exceeded more than once every three years, at temperatures of 18-22°C.

The ANZECC guidelines for ammonia are currently being revised and final USEPA guidelines for ammonia have just been released (USEPA 2013). The USEPA (2013) 1-hour average for acute exposures at pH 7.0, 20°C is 17 mg NH_4-N/L. The chronic criterion

[4] The concentration of a toxicant where 50% of its maximal effect is observed, for an exposure period of 96 hours.
[5] The concentration of a toxicant that is lethal to 50% of the sample population, over an exposure period of 96 hours

concentration (CCC) is 1.9 NH_4-N/L pH 7.0, 20°C as a 30-day rolling average, and is not to exceed 2.5 times the CCC as a 4-day average within the 30-days (i.e., 4.8 mg NH_4-N/L at pH 7 and 20°C). Neither the acute or chronic criteria are to be exceeded more than once in three years on average. For comparison with ANZECC (2000) we note that at pH 8, 20°C the CCC are now 0.78 mg NH_4-N/L (USEPA 2013). Finally the new USEPA criteria allow for the increased sensitivity of salmonid fish to ammonia toxicity at temperatures <16°C, (when if these fish are present in the water body of concern), separate tables can be used for pH and temperature specific acute criteria.

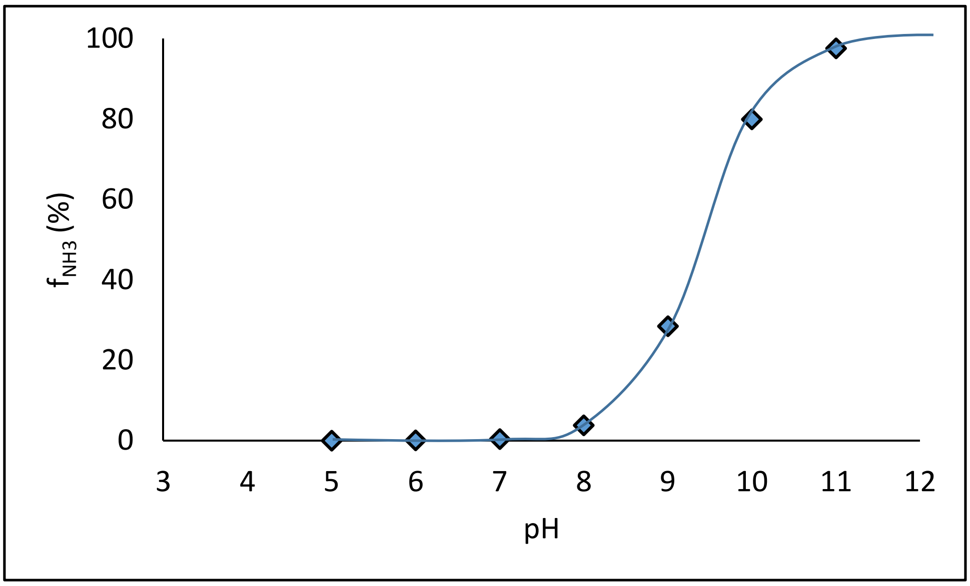

Figure 4-5: Variation if the proportion of total ammonia that is free (NH3) with pH, at 20°C.

At pH values of 7 or less, toxic un-ionised ammonia is not greater than 1% of total ammonia (i.e., what is operationally measured), so that a lower pH bound is not necessary when considering ammonia toxicity.

4.3.2 Low pH and hydrogen sulphide

Low pH (i.e., less than 6) limits should be set when metal toxicity is of concern, taking metal speciation into account. A particular case may be made for limiting low pH to control hydrogen sulphide (H_2S) toxicity where there is gross organic pollution and a likelihood of sulphate reduction. H_2S undergoes dissociation to H^+ and the bisulphide ion (HS^-) such that at pH 7 the ratio of toxic (H_2S) to non-toxic (HS^-) forms is roughly 1:1. At lower pH values the H_2S form becomes more abundant and toxicity to aquatic life is more likely. For example, a combined H_2S+HS^- concentration of 1 mg/L has a H_2S concentration of about 0.5 mg/L at pH 7 and 0.99 at pH 5. Published aquatic toxicity thresholds of H_2S, expressed as threshold limit median concentrations range from 0.007 to more than 1 mg/L[6].

[6] http://avogadro.chem.iastate.edu/MSDS/hydrogen_sulfide.pdf

4.4 Overseas criteria, guidelines and standards for pH

4.4.1 USA

The USEPA water quality criterion for pH in freshwater is currently 6.5-9.0 and was initially promulgated in “The Red Book” (USEPA 1976). The standard is based on the following reported observations (Table 4-1):

Table 4-1: Responses of freshwater fish to pH (USEPA 1976).

pH range	Effect on fish
5.0 – 6.0	Unlikely to be harmful to any species unless either the concentration of free CO_2 is greater than 20 mg/L, or the water contains iron salts which are precipitated as ferric hydroxide, the toxicity of which is not known.
6.0 – 6.5	Unlikely to be harmful to fish unless free CO_2 is present in excess of 100 mg/L.
6.5 – 9.0	Harmless to fish, although the toxicity of other poisons may be affected within this range.

4.4.2 Alaska

The Alaska Department of Environmental Conservation (ADEC 2012) water quality standard for “growth and propagation of fish, shellfish, other aquatic life, and wildlife” is that pH may not be less than 6.5 or greater than 8.5 and may not vary more than 0.5 pH unit from natural conditions.

4.4.3 Canada

The Environment Canada (CCREM 1987) freshwater pH guideline for the protection of aquatic life, is that the pH of water should not vary beyond the range of pH 6.5-9.0. The rationale for this is that “there is no definite pH range within which a fishery is unharmed and outside of which it is damaged, but rather there is a gradual deterioration of water quality as pH values are removed from the normal range (EIFAC 1969). The pH range which is not acutely lethal to fish is pH 5-9. Individual fish species have an optimum pH within this range (Alabaster & Lloyd 1982); however, the toxicity of several common pollutants is markedly affected by pH changes within this range, and increasing acidity or alkalinity may make these pollutants more toxic. Also, an acid discharge may liberate sufficient carbon dioxide from bicarbonate to cause the pH range of 5-6 to become lethal. At lower concentrations of calcium, the toxicity of elevated hydrogen ion concentrations to fish is increased (McDonald et al. 1983).

4.4.4 South-east Australia

The ANZECC (2000) default trigger values for pH are as follows (Table 4-2).

Table 4-2: Default trigger values of pH for south-east Australia (ANZECC 2000).

Ecosystem type	Range	Comment
Upland river	6.5 – 7.5	General
	6.5 – 8.0	NSW upland rivers
	4.0 – 6.5	Humic-rich Tasmanian lakes and rivers
Lowland river	6.5 – 8.0	General
	6.5 – 8.5	NSW lowland rivers

4.4.5 South west Australia

Default trigger values for south-west Australia are 6.5-8.0 for upland and lowland rivers (ANZECC 2000).

4.4.6 European Union

The European Union Freshwater Directive sets pH limit values of 6–9 for the fresh waters capable of supporting fish life. “Artificial pH variations with respect to unaffected values shall not exceed ± 0.5 of a pH unit within the limits falling between 6.0 and 9.0 provided these variations do not increase the harmfulness of other substances present in the water” (EU 2006).

4.4.7 Summary – Development of limits for pH

- There is a lack of continuous monitoring data for pH which, when combined with the absence of adequate biological response data for diel pH change, makes it difficult to recommend upper and lower bounds for pH based on local conditions.
- Laboratory experiments with New Zealand fish and invertebrate species indicate a tolerance range of pH 6.5-9.5 for the species tested. This compares with trigger values and recommended guidelines and standards, as follows:

pH range	Authority
6.5–9.0	USEPA
6.5–8.5	Alaska DOEC
6.5–8.0	NSW upland (ANZECC)
6.5–8.5	NSW lowland (ANZECC)
6.0–9.0	EU

- For peat waters, limits for ecosystem health should be established by monitoring of unimpacted wetland and rivers draining peat bogs to establish ‘natural’ ranges. A pH range for managing peat lakes and streams of 4.0-6.5 is recommended.
- A low pH limit of 6.0 may be necessary to prevent toxicity caused by metals and H_2S. This should be evaluated on a case-by-case basis.

- Ammonia toxicity presents a special case and pH limits to manage ammonia should take into account chronic and acute maximum concentrations of NH_4 and temperature. As a general guideline, where chronic ammonia concentrations are close to the ANZECC toxicant trigger value of 0.90 mg N/L for the 95% level of protection of freshwater species, pH should not be greater than 8.0 at water temperatures up to 20°C. Generally though, at typical ambient NH_4 concentrations in freshwaters (i.e., less than 0.1 mg total NH_4-N/L) a pH limit of 9.5 is adequate to obviate ammonia toxicity.

4.5 Proposed framework for managing pH risk for Aquatic Species

4.5.1 Narrative band descriptions

Consistent with other NOF frameworks, we propose that A grade waters are those where no aquatic organisms that would be present in reference or near-pristine New Zealand rivers are subjected to stress caused by pH, or pH change (Table 4-3). Conversely, D grade waters (below the 'bottom line') are those in which aquatic organisms suffer "considerable" stress as a result of overly acidic or alkaline conditions, with mobile animals (fish) moving to more circum-neutral refugia (if they can), and some macroinvertebrates being locally extinguished. There is a significant loss of ecological integrity implied with a D grading, which is therefore to be avoided.

Between these extremes, moving from A to B to C-graded waters, there is a gradient of increasing pH stress on certain (sensitive) aquatic fauna and on aquatic ecosystems.

4.5.2 Numeric boundaries (tentative)

Numeric band boundaries are proposed in Table 4-3. Surprisingly, it is perhaps easier to enumerate the A/B boundary, the no-pH-effect limit, than the C/D ('bottom line') limit. Consistent with the A grade narrative Table 4-3, the A/B boundary should be set where no aquatic organism (that would otherwise be present) is subjected to stress caused by pH.

4.5.3 Confidence in proposed NOF thresholds for pH

There is limited information available on the pH tolerances of native NZ fish and macroinvertebrates species. The pH values preferred and avoided by adult stages of inanga, common bully, common smelt and redfinned bully were determined experimentally, along with juvenile stages of inanga, common bully, shortjawed kokopu, koara, banded kokopu, longfinned eel elvers and glass eels (West et al. 1997). The acute tolerances were measured at exposure durations of 8-12 h but yielded results substantially in agreement with overseas studies and criteria for the protection of freshwater species that were often based on studies with longer exposure periods. Adult native fish species had stronger preferences than juveniles, with avoidance of pH values below 6.5 for all species except shortfinned elvers. Little is known of the preference that native macroinvertebrate species have for pH. Furthermore, little is known about the effects of diel changes in pH on freshwater fish and macroinvertebrate species. Most international guidelines that we have consulted provide some information on species present in New Zealand and (according to standard guideline derivation protocols) these criteria can be used in addition to those for native species. In summary the available information on native species and introduced species is substantially in agreement with long-established international pH guidelines and provides strong support for the proposed thresholds.

4.6 Conclusions - pH

pH is a property of freshwaters that affects many other water quality variables. Therefore data on pH, preferably continuous records defining pH regime, are needed to interpret other variables. The aqueous toxicities of ammonia, toxic metals and organic compounds are affected by pH. Additionally, although fish and invertebrate species demonstrate a wide range of tolerances and preferences to different pH ranges, there is common agreement that a pH range of 6-9 is needed to prevent excessively acidic or alkaline conditions adversely affecting freshwater ecosystems. These define the upper and lower bounds for pH in natural waters, excluding naturally alkaline or acidic waters such as humic-stained waters.

A national objective framework for pH is proposed (Table 4-3) built on a (narrative) gradient of increasing stress from the ‘no effect’ A grade to significant loss of ecological integrity at D grade. The framework should apply throughout the diel (24-hour) regime of pH measurements and not just to the narrower range of daytime ‘spot’ measurements that are commonly reported. This will require more extensive use of continuous pH monitoring.

The limits proposed in Table 4-3 are drawn from a comparatively few experimental and field observations of New Zealand fish and invertebrate species, as well as international standards, guidelines and trigger values based on a mixture of observations and statistical inferences.

Table 4-3: Proposed NOF for pH regime in rivers and streams. The term regime refers to the diel fluctuation of pH (and co-variation with temperature and dissolved oxygen) around the daily mean.

Value (use)		**Ecological Health**
Attribute		**pH regime**
Environment river, lake, groundwater, estuary, wetland		Rivers
Measurement unit		pH units are dimensionless
Summary statistic		Summer monitoring data upper 95th percentile
Band descriptors	A	No stress caused by acidic or alkaline ambient conditions on any aquatic organisms that are present at matched reference (near-pristine) sites.
	B	Occasional minor stress caused by pH on particularly sensitive freshwater organisms (viz. fish and insects).
	C	Stress caused on occasion by pH exceeding preference levels for certain sensitive insects and fish for periods of several hours each day.
	D (unacceptable/doesn't provide for value)	Significant, persistent stress caused by intolerable pH on a range of aquatic organisms. Likelihood of local extinctions of keystone species and destabilisation of river ecosystems.
Band boundaries (numeric)	A/B	6.5 < pH < 8.0
	B/C	6.5 < pH <8.5
	C/D	6.0 < pH < 9.0
	D (unacceptable/doesn't provide for value)	pH < 6 or pH >9
Are there circumstances where a water body could naturally fall into the D band?		Natural humic-stained streams may also have pH values <5.
Limitations/gaps/risks		These criteria do not apply to humic-stained streams. There is a wide range of sensitivities of freshwater fish and invertebrates to pH that is hard to capture with single criteria for each class. Special consideration is needed for (i) naturally acid waters (e.g., humic-stained streams), and (ii) where there are substances for which toxicity is affected by pH (viz. ammonia, toxic metals and sulphide and certain organic contaminants).
Notes:		Summer pH maxima data may need to be used if only limited data are available for a site. Continuous monitoring of pH in summer is required to provide reliable data on the diel pH variability.
Key references:		Alabaster and Lloyd (1982); West et al. 1997)

5 Accounting for the interaction of temperature, DO and pH – multiple stressors and NOF thresholds

There are uncertainties around how to protect against **interaction** of thermal stress with other stressors, notably dissolved oxygen (DO). As a start, and in the absence of explicit interaction criteria, we suggest that if temperature and at least one other stressor (say DO) *both* indicate a “C” grading, that should be interpreted as a “D” (unacceptable) ‘overall’ grading for the water. This ‘two C’s counts as a D’ approach has not been applied, hitherto, so far as we are aware. The point here is that a C grading for both temperature and another key stressor such as DO identifies the water as a high priority for management in terms of reducing temperature (e.g., by riparian shade plantings), or the other stressor, or both.

6 Continuous monitoring of temperature, DO and pH over a diel cycle

Managing the variables, DO, temperature and pH that vary over a diel cycle in rivers requires continuous data so that afternoon maxima or minima (near dawn) can be recognised and compared with NOF limits. However most (current) SoE monitoring in New Zealand involves only 'snap-shot' measurements of these variables during daytime hours. Clearly this will under-estimate potentially stressful low DO near dawn, and may also 'miss' the times of stressful high temperature and pH occurring, typically, in the late afternoon. In the NEMaR (National Environmental Monitoring and Reporting) project, Davies-Colley et al. (2012a,b) strongly recommended that 'snap-shot' visits to river monitoring sites be made at the same time of day (NZST), within about 1 hour, so that the influence of diel fluctuations on trend analysis is minimised. On the other hand it is impractical to measure all sites at the critical time of day by manual or spot monitoring. For interpretation of NOF limits, in the absence of supplementary continuous data, some means of estimating maximum temperatures and minimum DO from such snap-shots is required. This may require operating continuous sensors at key locations, at least during the most stressful time of year (typically mid to late summer), together with modelling.

The NEMS (National Environmental Monitoring systems) project has recently produced standards for continuous monitoring (e.g., for DO) so guidance is available for operating continuous monitoring sensors. Continuous temperature monitoring with inexpensive thermistor sensors (some with built-in logging capability e.g., Hobo loggers http://www.onsetcomp.com/) is probably most straightforward – and have been used routinely for 10 years in NIWA-designed forest stream monitoring programmes. NIWA is currently instrumenting sites in its 'benchmark' network, (that were formerly NRWQN sites), with continuous temperature and turbidity sensors, and regional councils are increasingly deploying temperature loggers over summer months.

Note that there is still a need for independent and frequent verification of these continuous temperature records, most conveniently by comparison with 'snapshot' temperatures measured at times of monthly SoE inspections. NIWA's experience is that instruments such as the Hobo loggers are reliable, relatively inexpensive, and seem to be immune to drift over deployments of several months over summer. The SoE visits also provide an opportunity to check the sensors have not been isolated from the free-flowing water, for example owing to occlusion by moving gravel bars or trash such as plastic bags). Temperature sensors are less prone to fouling problems than those for DO and pH.

Continuous DO and pH monitoring is appreciably more challenging than temperature monitoring because of the nature of sensors and their tendency to drift (Wilcock, B. et al. 2013 *in press*). Fortunately, continuous pH monitoring is probably not required except where other information suggests a pH 'issue'. Continuous DO monitoring demands frequent verification by 'snap-shot' measurements, usually more often than monthly so that visits will be needed between (monthly) SoE visits. There will usually also be a need for appreciable 'post-processing' of data to correct for 'spikes' and drift (Wilcock, B. et al. 2013 *in press*).

7 Recommendations

In the course of preparation of this report we have become aware of numerous limitations of current scientific understanding of New Zealand temperature, dissolved oxygen and pH regimes in our rivers and streams, and on tolerance of New Zealand indigenous animals to temperature, dissolved oxygen and pH extrema. Specific recommendations on just three of the more important research and policy needs arising from this report are as follows:

1. A national-scale project is recommended to evaluate reference temperature regimes in streams throughout New Zealand. This is needed to enable application (and potential refinement) of temperature limits expressed as increments above reference (high) temperatures rather than as absolute temperatures.

2. A national campaign on river and stream temperatures in New Zealand, modelled on the “Keeping our rivers cool” programme sponsored by the Environment Agency of England and Wales (www.environment-agency.govt.uk), seems desirable. This would raise awareness of the issue of heating during summer low flows of poorly shaded streams in cleared land, and encourage wider uptake of stream riparian planting as an adaptation to global warming.

3. Research is needed on dissolved oxygen tolerance of early life stages of New Zealand indigenous fish species and a wider range of New Zealand invertebrates than has been studied hitherto.

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Appendix A Temperature – Background Information

What is temperature?

Temperature is sometimes taken to be synonymous with heat, and indeed temperature and heat are intimately related. But to confuse temperature with heat is an elementary error. Heat is a form of energy, while temperature is a description of state of matter. Temperature can be regarded as a macroscopic expression of the average kinetic energy of molecules or atoms. Adding heat increases temperature (i.e., increases average molecular kinetic energy), but different materials have a different temperature response to the same quantity of heat. Water is noteworthy for having a very high heat capacity meaning that unusually large quantities of heat are needed to increase temperature of a kilogram of water by 1ºC. This high heat capacity of water relates to the thermal regulation of Earth (which has water in all three states – a highly 'improbable' condition, astronomically speaking), and the suitability of this planet for life.

Controls on temperature in river waters

The temperature of rivers expresses the balance point of heat gain mechanisms, primarily solar radiation, *versus* several different heat loss mechanisms including 'long-wave' thermal infra-red radiation and fluid convection. The higher the temperature of water is above its surroundings, due to daytime solar heating, the faster it loses heat by thermal infra-red radiation and other heat loss processes.

Although temperature of water broadly follows that of air the association is only partially causative. In fact water is usually warmer than air, on average, because it directly absorbs most of the solar radiation (all of the solar radiation in the near-infra-red range (~50% of total), and at least some of the radiation in the visible range (~50% of total)) incident on its surface. In contrast, air is transparent to solar radiation and is only heated (indirectly, by heat transfer processes – conduction, convection, net thermal radiation) by hotter underlying soil or water.

Rutherford et al. (1997) give a summary of the components of the energy balance equation in the context of temperature modelling for small streams in New Zealand. The equation itself, and the formulation of its different terms, need not concern us here; the important point to grasp is solar heating *versus* a variety of heat loss mechanisms. The net result of the energy *in*-balance brought about by insolation is that rivers and streams have a strong seasonal fluctuation in (average) temperature due to seasonality in heating by solar radiation. A strong diel fluctuation is superimposed on the seasonal curve of temperature due to the day-night cycle of solar radiation.