CAWTHRON

# Water Quality Guidelines To Protect Trout Fishery Values

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Joe Hay

John Hayes

Roger Young

Prepared for

horizons regional council

Cawthron Institute
98 Halifax Street East, Private Bag 2,
Nelson, New Zealand.
Ph. +64 3 548 2319,
Fax. + 64 3 546 9464
www.cawthron.org.nz

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# 1. INTRODUCTION

Horizons Regional Council (Horizons) have sought water quality guidelines for the protection of trout fisheries values in streams identified as significant trout fisheries in their area of jurisdiction.

This report provides three tables of water quality guidelines, the first containing guideline values for fisheries identified as "Outstanding" or "Regionally significant", the second containing guideline values for fisheries classified as "other significant fisheries", and the third containing guideline values for protection of spawning and incubation in recognised spawning streams. Some background information is also provided for each of the water quality guidelines contained in the tables.

It is our opinion that the four key parameters for the protection of adult trout are water temperature, dissolved oxygen, water clarity/ turbidity, and food (represented in this case by the Macroinvertebrate Community Index (MCI)). Temperature and dissolved oxygen have obvious direct effects on metabolism, while water clarity can influence foraging efficiency for drift feeding trout (the most common feeding behaviour of trout in New Zealand streams).

For spawning and incubation the main issues are also temperature and dissolved oxygen, as well as maintaining a relatively low fine sediment fraction in the substrate. Suitable guidelines for water clarity and the MCI could act as surrogate controls on fine sediment load in this instance, since the fine sediment fraction is laborious to measure.

# 2. GUIDELINE VALUES

**Table 1.** Water quality guidelines intended to maintain near optimal conditions for trout in rivers designated "Outstanding" or "Regionally significant" trout fisheries (N.B. the parameters in the tables are hyperlinked to the appropriate section in the background section).

| Parameter | Standard | Flow at which the standard applies | Season during which the standard applies | Reference(s) |
|---|---|---|---|---|
| pH | 7.3-8.0 Upland streams & 7.2-7.8 Lowland streams | Baseflow conditions (< median flow) | All year | ANZECC 2000 |
| Temperature | 19 °C as daily maximum | | | Elliott 1994 |
| | Shall not be changed by more than 3 °C | | | RMA 1991 |
| Dissolved Oxygen | > 80 % saturation | | | ANZECC 1992 & RMA 1991 |
| Periphyton Biomass | For Lowland streams,<br>Diatoms and cyanobacteria: 200 mg/m² Chlorophyll *a* 35 g/m² AFDW †<br>Filamentous algae: 120 mg/m² Chlorophyll *a* 35 g/m² AFDW | | | Biggs 2000 |
| | For Upland streams,<br>50 mg/m² Chlorophyll *a* maximum<br>15 mg/m² Chlorophyll *a* mean monthly | | | Biggs 2000 |
| Periphyton cover | 30 % > 2 cm long for filamentous algae | | | Biggs 2000 |
| MCI | > 120 | | | Stark 1985 |
| Soluble Inorganic Nitrogen (SIN) | < 10 µg-N/ L to <295 µg-N/ L depending on accrual period | | | Biggs 2000 |
| Soluble Reactive Phosphorus (SRP) | < 1 µg-N/ L to < 26 µg-N/ L depending on accrual period | | | Biggs 2000 |
| Ammoniacal N | 10 µg-N/ L in upland streams 21µg-N/ L in lowland streams | | | ANZECC 2000 |
| Clarity (Black Disc) | 5 m | | | |
| Turbidity | 0.5 NTU | | | |
| Faecal contamination | 260 *E.coli*/ 100 mL | | | MfE 2003 |
| Other Toxicants | Level for protection of 99 % of species | | | ANZECC 2000 |

† Ash Free Dry Weight

**Table 2.** Water quality guidelines intended to minimize adverse effects on trout in rivers designated "other significant trout fisheries" (N.B. the parameters in the tables are hyperlinked to the appropriate section in the background section).

| Parameter | Standard | Flow at which the standard applies | Season during which the standard applies | Reference(s) |
|---|---|---|---|---|
| pH | 7.3-8.0 Upland streams & 7.2-7.8 Lowland streams | Baseflow conditions (< median flow) | All year | ANZECC 2000 |
| Temperature | 24 °C as daily maximum | | | Elliott 1994 |
| | Shall not be changed by more than 3 °C | | | RMA 1991 |
| Dissolved Oxygen | > 80 % saturation | | | ANZECC 1992 & RMA 1991 |
| Periphyton Biomass | For Lowland streams,<br>Diatoms and cyanobacteria: 200 mg/m$^2$ Chlorophyll *a* 35 g/m$^2$ AFDW<br>Filamentous algae: 120 mg/m$^2$ Chlorophyll *a* 35 g/m$^2$ AFDW | | | Biggs 2000 |
| | For Upland streams,<br>50 mg/m$^2$ Chlorophyll *a* maximum<br>15 mg/m$^2$ Chlorophyll *a* mean monthly | | | Biggs 2000 |
| Periphyton cover | 30 % > 2 cm long for filamentous algae | | | Biggs 2000 |
| MCI | > 100 | | | Stark 1985 |
| Soluble Inorganic Nitrogen (SIN) | < 10 µg-N/ L to <295 µg-N/ L depending on accrual period | | | Biggs 2000 |
| Soluble Reactive Phosphorus (SRP) | < 1 µg-N/ L to < 26 µg-N/ L depending on accrual period | | | Biggs 2000 |
| Ammoniacal N | 10 µg-N/ L in upland streams 21µg-N/ L in lowland streams | | | ANZECC 2000 |
| Clarity (Black Disc) | 3.75 m | | | |
| Turbidity | 0.7 NTU | | | |
| Faecal contamination | 260 *E.coli*/ 100 mL | | | MfE 2003 |
| Other Toxicants | Level for protection of 99 % of species | | | ANZECC 2000 |

**Table 3.** Water quality guidelines intended to maintain near optimal conditions for trout spawning and incubation in streams managed for spawning (N.B. the parameters in the tables are hyperlinked to the appropriate section in the background section).

| Parameter | Standard | Flow at which the standard applies | Season during which the standard applies | Reference(s) |
|---|---|---|---|---|
| pH | 7.3-8.0 Upland streams & 7.2-7.8 Lowland streams | Baseflow conditions (< median flow) | Spawning and incubation (approximately May-October) | ANZECC 2000 |
| Temperature | < 11 °C as daily maximum | | | Bell 1986 & Jowett 1992 |
| | Shall not be changed by more than 3 °C | | | RMA 1991 |
| Dissolved Oxygen | > 80 % saturation | | | ANZECC 1992 & RMA 1991 |
| Periphyton Biomass | For Lowland streams,<br>Diatoms and cyanobacteria: 200 mg/m$^2$ Chlorophyll *a* 35 g/m$^2$ AFDW<br>Filamentous algae: 120 mg/m$^2$ Chlorophyll *a* 35 g/m$^2$ AFDW | | | Biggs 2000 |
| | For Upland streams,<br>50 mg/m$^2$ Chlorophyll *a* maximum<br>15 mg/m$^2$ Chlorophyll *a* mean monthly | | | Biggs 2000 |
| Periphyton cover | 30 % > 2 cm long for filamentous algae | | | Biggs 2000 |
| MCI | > 100 | | | Stark 1985 |
| Soluble Inorganic Nitrogen (SIN) | < 10 µg-N/ L to <295 µg-N/ L depending on accrual period | | | Biggs 2000 |
| Soluble Reactive Phosphorus (SRP) | < 1 µg-N/ L to < 26 µg-N/ L depending on accrual period | | | Biggs 2000 |
| Ammoniacal N | 10 µg-N/ L in upland streams 21µg-N/ L in lowland streams | | | ANZECC 2000 |
| Clarity (Black Disc) | 3.75 m | | | |
| Turbidity | 0.7 NTU | | | |
| Other Toxicants | Level for protection of 99 % of species | | | ANZECC 2000 |

# 3. BACKGROUND TO GUIDELINE VALUES

## 3.1. Temperature

The incipient lethal temperature is usually defined as that which fish (usually 50% of a given sample) can tolerate for a prolonged period (seven days is the usual standard) but beyond which fish cannot tolerate for an indefinite period (Elliott 1994). The ultimate lethal temperature is that which fish cannot tolerate for even a short period (10 minutes is the usual standard). The incipient lethal temperature for brown trout increases with acclimation to a plateau at 24.7 °C, while the ultimate lethal temperature reaches a plateau at 29.7 °C (Elliott 1981; Elliott 1994; Elliott & Elliott 1995). Behavioural disturbances can be expected at temperatures less than the incipient lethal temperature. For example, brown trout cease feeding at temperatures above 19 °C. Less detailed information on temperature requirements is available for rainbow trout. Their upper incipient lethal temperature is 26.2 °C, the ultimate lethal temperature is approximately 30 °C, and behavioural disturbances occur above 19 °C (Elliott 1994).

The above information on lethal and sub-lethal temperatures is for relatively short term exposure. From these data, one might get the impression that a trout population could "cope" with short to longer periods of high water temperature, providing they did not exceed the short term lethal tolerances. This is an incorrect assumption. In fact the impacts of "sub-lethal" high water temperatures are expressed not only in fish behaviour and growth rate but also in survival rates and population production. For example, mortality increases as water temperature rises above the growth optima (14 °C – 17 °C for brown trout, 16 °C – 18 °C for rainbow trout). A study on rainbow trout found that the maximum temperature at which a population can be expected to maintain its weight (biomass) was a constant temperature of 23 °C and a fluctuating mean temperature of 21 °C (Hokanson *et al.* 1977) (*i.e.* the temperature at which population production is zero). Given the differences in temperature preference between the species, an equivalent zero production temperature for brown trout is likely to be 19 °C. These maximum temperatures apply to fish on maximum rations; they will be even lower for food-limited populations (Hokanson *et al.* 1977). Stress, increased risk of disease and greater activity of predators are reasons for the increased mortality with increasing "sub-lethal" temperatures. The above information indicates that the productivity of a trout population (especially brown trout) will suffer as water temperature approaches and exceeds 19 °C.

Trout deaths have been reported in New Zealand rivers when water temperatures have equalled or exceeded 26 °C (Jowett 1997).

Bell (1986) notes that fish eggs are sensitive to temperature changes, especially during the first half of their incubation period. He goes on to provide thermal tolerance limits, the exceedance of which can be expected to result in increased mortality. For brown trout spawning he gives a range of 3 - 20 °C, with an optimum of 10 °C; and for hatching a preferred range of 2 – 11 °C, but with a maximum of 20 °C (the ranges he provides for rainbow trout are largely similar).

On the other hand brown trout embryos have been known to incubate at 1 to 2 °C for many weeks (Raleigh *et al.* 1986). Raleigh *et al.* (1986) suggest optimal incubation temperatures for brown trout of 2 to 13 °C, with a tolerance range of 0 to 15 °C. Raleigh *et al.* (1984) provides a temperature range for rainbow trout spawning of approximately 2 – 15.5 °C, and for egg incubation of 5.5 – 12 °C as the desirable range, but approximately 1.5 – 16 °C as the extremes. The temperature range suggested by Raleigh *et al.* (1984) for rainbow trout spawning and incubation closely match the findings of Humpesch (1985), who experimentally assessed incubation and hatching success at a range of temperatures for five species of salmonid. However, Humpesch's findings for brown trout would suggest a lower optimum range (1 - 9 °C) than that given in Raleigh *et al.* (1986).

Winter water temperature was one of the main predictive factors in Jowett's (1992) "100 rivers models" predicting brown trout abundance in New Zealand rivers. Jowett's study indicated that rivers with winter water temperatures > 10 °C contained very few, or no, brown trout. Winter temperatures exceeded 10 °C in only eight of the 89 sites in 82 rivers used to construct his models. When these sites where excluded, no significant correlations remained between any temperature variable and brown trout abundance. It appears from this that high water temperatures in winter (the spawning and incubation period) may limit brown trout recruitment in New Zealand rivers. This is supported by Scott & Poynter (1991), who showed that temperature increases, predicted under climate change scenarios, had the potential to reduce the northern range of both brown and rainbow trout in New Zealand. Increased winter temperatures, affecting spawning and incubation, appeared likely to have the greatest impact.

In our opinion, a conservative approach would be to maintain water temperature below 19 °C in rivers designated as "Outstanding" or "Regionally significant" trout fisheries, to avoid the behavioural disturbances (*e.g.* cessation of feeding) observed at higher temperatures. For those designated "Other significant trout fisheries" we recommend that water temperatures be maintained below 24 °C to avoid the lethal effects of high temperature. During the spawning and incubation period we suggest that the water temperature in streams managed for trout spawning be maintained at < 11 °C.

## 3.2. Dissolved Oxygen

The dissolved oxygen requirements of salmonids are higher than for most other freshwater fishes (Dean & Richardson 1999). A minimum oxygen concentration of 5.0 – 5.5 mg/L can be tolerated by free swimming brown trout but should be at least 80 % saturation (Mills 1971 cited in Elliott 1994). The incipient lethal level of dissolved oxygen concentration for free swimming brown and rainbow trout is about 3 mg/L (Raleigh *et al.* 1984, 1986). The oxygen requirements of salmonids increase with water temperature due to increased metabolic rate (Elliott 1994). When water temperature exceeds 10 °C, rainbow trout generally avoid water with dissolved oxygen concentrations < 5 mg/L (May 1973). However, as oxygen concentration falls toward this level, the health, growth, reproduction, and survival of the fish may be jeopardised. Consequently the > 6 mg/L and 80 % saturation guidelines stipulated in ANZECC (1992) would be best interpreted as short term exposure levels (*i.e.* days). If spot measurements are used, these should be taken in the early morning when DO levels are likely to be at their lowest. Long term exposure to dissolved oxygen levels of even 6 mg/L can chronically impair the growth of salmon, by up to 20 % depending on the water temperature (BCME 1997). Following the BCME (1997) guidelines, 8 mg/L is an appropriate long term (*e.g.* 30 day mean) level for best protection of salmonids and other aquatic life.

Large fluctuations in oxygen concentration may also cause a reduction in food consumption and impaired growth (Doudoroff & Shumway 1970).

Brown trout egg development ceases when DO levels fall below 4.5 mg/L (Raleigh *et al.* 1986). Oxygen concentrations within redds can be reduced if fine sediments within them are rich in organic material (Bjornn & Reiser 1991), due to the oxygen used in decomposition. This can result in asphyxiation of embryos or alevins (larvae) if oxygen concentrations become very low (Bjornn & Reiser 1991). BCME (1997) suggest an instantaneous minimum of 9 mg/L and a 30 day mean of 11 mg/L (based on five approximately evenly spaced measurements, targeting the time of day when DO is expected to be lowest, measured in the water column) to protect developing eggs and alevins in the substrate. This assumes that DO levels in the water column are likely to be in the order of 3 mg/L higher than those experienced in the substrate interstitial space.

In our opinion, the > 80 % saturation guideline suggested in ANZECC (1992) should provide adequate protection on its own, so long as the temperature guidelines are adhered to. The weight per volume of dissolved oxygen required to achieve 80 % saturation does not fall below 6 mg/L until temperatures of > 30 °C. Also, 80 % saturation is approximately equivalent to the 9 mg/L guideline (suggested in BCME 1997) for protection of eggs and fry in the substrate at the 11 °C maximum water temperature guideline for this value, and is more conservative at lower temperatures.

## 3.3. Clarity and Turbidity

Water clarity (measured by the black disc method) and turbidity (NTU) are fairly well correlated, although there is also a fair degree of variability in this relationship between rivers (Davies-Colley & Close 1990). This means that neither of the two measurements can be deduced based on the other with a great deal of precision. Ideally the river specific relationships between black disc and NTU should be determined on a case by case basis.

Since trout are visual predators and drift feeding is the predominant foraging behaviour in most rivers (especially those of moderate to steep gradient), increased turbidity (*i.e.* lower water clarity) would be expected to have an adverse effect on trout because it reduces their foraging radius and their foraging efficiency. Indeed, experiments with brook trout in an artificial stream show that although increased turbidity had no significant effect on mean daily consumption, specific growth rates were significantly reduced (Sweka & Hartman 2001). This is because trout abandoned drift feeding in favour of active searching - which is energetically more expensive - as turbidity increased. In other words, where food is reasonably abundant, reduced water clarity reduces foraging efficiency with the result that trout spend more time (and energy) foraging in order to meet their food requirements (either by drift feeding or active searching). Extending this argument further, reduced water clarity can be expected to compound an already reduced growth rate where food is scarce and trout cannot achieve maximum daily consumption.

Rowe *et al.* (2003) found that turbidity had no effect on feeding rate for rainbow trout. However, these experiments were undertaken in tanks, where the lack of current would have precluded drift feeding, and trout would have to actively search for prey. An important confounding feature of Rowe *et al.*'s study design was that prey density in their experiments was exceptionally high relative to densities normally expected in nature (a result of the small size of the experimental tanks), meaning that there was a high probability of fish encountering prey items regardless of the level of turbidity.

Gregory & Northcote (1993) reported a log linear decline in reaction distance to invertebrate prey with increasing turbidity for juvenile Chinook salmon. Barrett *et al.* (1992) also found that increased turbidity strongly reduced reaction distances of juvenile rainbow trout to drifting prey items in artificial stream channels.

These data suggest that an understanding of the mechanics of drift foraging might inform on the level of water clarity required to maintain drift foraging opportunity. Hughes & Dill (1992) developed a drift foraging model for Arctic grayling, which describes the geometry of drift foraging and how this is related to fish and prey size, water temperature and velocity. This model has since been employed in a coupled drift foraging and bioenergetics model for brown trout developed by the Cawthron Institute (Hayes 2000), and subsequently successfully tested in the Maruia River (Hayes *et al.* 2000).

Hughes and Dill's (1990) drift foraging model predicts that reaction distance for a given prey size plotted against fish length reaches an asymptote (Figure 1). For 12 mm prey this asymptote is at approximately 1.4 m. However, the reaction distance to larger prey items is obviously greater (*e.g.* for a 60 cm fish with a 30 mm prey item the reaction distance is predicted to be approximately 3.6 m). The majority of drifting prey eaten by trout in New Zealand rivers is 12 mm or less (because most drifting invertebrates are in this size range). Therefore, it may seem reasonable that if water clarity is maintained above 1.4 m, the foraging area of drift feeding trout should not be substantially reduced. However, larger drifting prey items (including stoneflies, large swimming mayflies and large terrestrial invertebrates), although encountered less frequently, make an important energetic contribution to the diet of larger trout. Therefore, in order to maintain optimum foraging conditions a higher level of water clarity may be justified (*e.g.* the 3.6 m reaction distance for 60 cm fish to 30 mm prey item, as discussed above).

On the other hand, it is not certain whether the visual clarity measurements based on black disc are directly comparable to the ability of trout to perceive prey items (much smaller than the black disc) under similar conditions. This is an area that requires more research. However, as discussed above, there is research supporting a negative relationship between turbidity and reaction distance in salmonids.

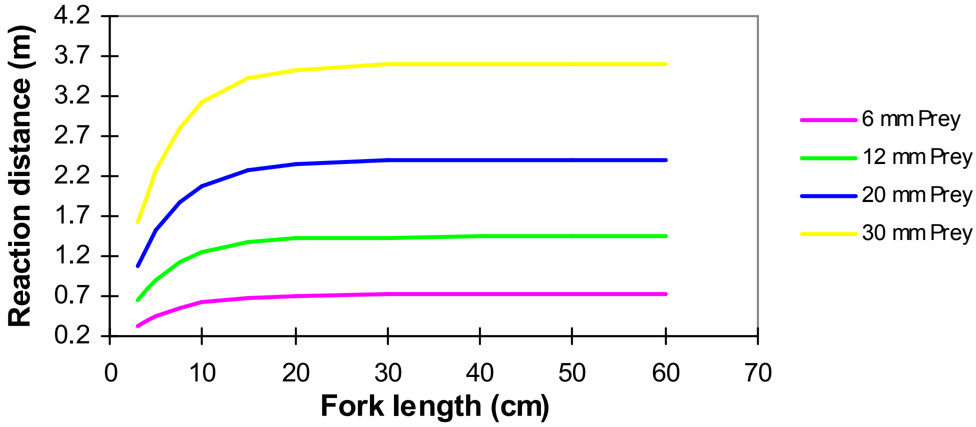


**Figure 1.** Reaction distance to drifting invertebrate prey relative to fish size, based on Hughes & Dill's (1990) drift foraging model, for a range of sizes of invertebrate prey.

Given the apparent lack of equivalent empirical data for brown trout, the relationship reported by Gregory & Northcote (1993), for juvenile Chinook salmon, was used by Hayes 2000 to adjust the reaction distance predicted by Hughes & Dill's (1990) model for the effect of turbidity. Based on this adjusted model the reaction distance, at 0.5 NTU (the lowest practical value for comparison), is predicted to be reduced by approximately 50 % as turbidity increases to about 10 NTU (Figure 2). The maximum reaction distances to various prey sizes shown in Figure 1 are for the clear water (0.5 NTU) condition. It is possible to approximate the level of water clarity (as measured by black disc) that would be required to maintain reaction distances based on a relationship between NTU and black disc water clarity.

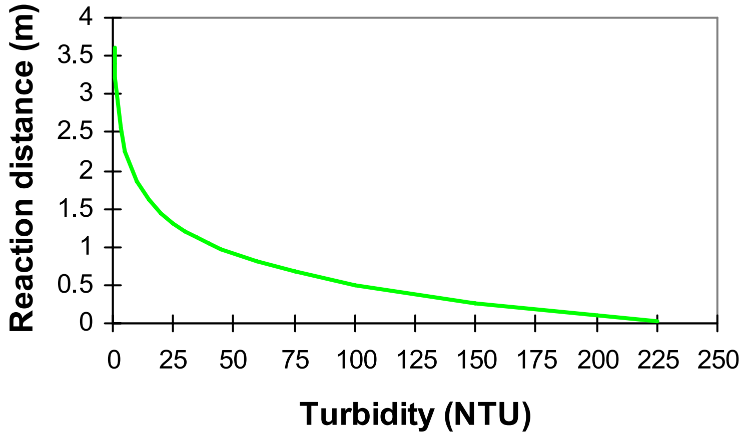


**Figure 2.** Attenuation of the predicted reaction distance of a drift foraging salmonid with increasing turbidity. Based on Hughes & Dill's (1990) foraging model predictions (for a 60 cm trout with 30 mm prey), modified by the NTU versus reaction distance relationship from Gregory & Northcote (1993)

Based on the relationship reported in Davies-Colley & Close (1990) for 190 observations in 96 New Zealand rivers at base flow, the clear water condition, represented by 0.5 NTU in Figure 2, would equate to approximately a 5 m black disc water clarity reading. This level of water clarity would arguably be appropriate to maintain optimum drift feeding conditions in rivers managed for their "Outstanding" or "Regionally significant" trout fishery values. For the "Other significant trout fisheries" some level of reduction in foraging area would be permissible. Reducing foraging area by 5 % or 10 % would be expected to produce similar reductions in energetic returns from foraging. Foraging area is proportional to the square of reaction distance, so a reduction in reaction distance will result in a larger proportional reduction in foraging area. Reductions of approximately 5 % and 10 % in foraging area would result from reduction in reaction distance of approximately 3 % and 5 %, respectively. These reductions would equate to approximately 0.6 NTU and 0.7 NTU, respectively, which in turn would translate to approximately 4.75 m and 3.75 m black disc.

These levels of water clarity would place rivers managed for their "Outstanding" or "Regionally significant" trout fishery values just below the 80th percentile of the 96 rivers sampled by Davies-Colley & Close (1990) at base flow, while the level suggested for "Other significant trout fisheries" would place them between the 60th and 75th percentiles. These appear to be reasonable levels, especially given that aesthetics (including water clarity) generally play an important role in the level of enjoyment derived from trout fishing.

As well as reducing foraging efficiency, high loads of fine sediment in rivers are known to impact fish, both through direct physical effects, and less directly as a result of effects on habitat and food availability. Suspended sediments can scour and abrade fish, particularly the gill-rakers and gill filaments, making those in turbid waters more susceptible to disease and even causing mortality in extreme cases (Ryan 1991; Wood & Armitage 1997). Deposited fine sediments can cause a reduction in suitable spawning habitat, reducing survival or hindering development of eggs and fry, and can reduce habitat and cover for both juvenile and adult fish. The severity of effects increases with both the concentration of suspended sediment and the duration of exposure, with these two factors in combination providing the strongest predictions of ill effects (Newcombe & MacDonald 1991; Newcombe & Jensen 1996). There is a strong correlation between water clarity and suspended solid concentrations (Davies-Colley & Close 1990). Therefore, in the absence of suspended solids data, water clarity during baseflow may serve as an indicator of fine sediment loads, with implications for the suitability of substrate for spawning and as habitat for macroinvertebrates important as trout food. Monitoring of water clarity may also provide a pragmatic approach to detecting/ avoiding the detrimental effects of excessive levels of siltation, given that assessments of embeddedness are notoriously subjective (Phillips & Basher 2005).

The discussion above suggests that maintaining water clarity (as measured by black disc) at 5 m in rivers designated “Outstanding” or “Regionally significant” trout fisheries and a lesser level of say 3.5 m in those designated “Other significant trout fisheries” should maintain reaction distances of drift-feeding trout at reasonable levels.

However, there may be situations where the underlying geology may render these guidelines unattainable (*e.g.* mudstone catchments). There may also be instances where more conservative levels of water clarity may be desirable (especially given that aesthetic values are often an important contributing factor in defining valued trout fisheries). In these situations site-specific water clarity guidelines could be derived by examining historic water clarity records from the relevant streams and setting guidelines based on historic exceedence levels. For example, if the guideline for water clarity were set at the value exceeded 90 % of the time in the existing water quality data set (excluding flood events, when water clarity is likely to be naturally lower), then the guideline would be expected to be breached 10 % of the time if the current state was maintained, but would be exceeded more frequently if the state were to decline. On the other hand, we realise that this approach may be difficult to apply in situations where streams are already degraded. In these instances it may be necessary to survey anglers to assess whether the fishery is still considered to rate as “Outstanding” or “Regionally significant”, and if not, when it began to decline. If a consensus emerges, then water clarity data predating this perceived decline could be used to derive the guidelines. Alternatively, guidelines could be set based on water clarity data from “reference” streams, with similar physical characteristics to those to which the guidelines are intended to apply.

## 3.4. Macroinvertebrate Community Index

The macroinvertebrate community index (MCI) was developed as a biomonitoring tool (Stark 1985). As such, it was designed to summarise information reflected in the macroinvertebrate community present at a site on the likely level of ecosystem degradation. Thus it is relevant as an indicator of the state of ecosystems supporting valued trout fisheries.

The MCI also has the potential to provide an indication of the relative availability of trout food species. Many of the taxa that scored highly in the MCI are also important prey for drift feeding trout. This could be improved upon by developing an index specifically targeted at assessing the relative quality of trout food.

We suggest that the MCI score in "Outstanding" and "Regionally significant" trout fishery rivers be maintained above 120 (indicative of clean water), and in "Other significant trout fishery" rivers that it be maintained above 100 (indicative of possible mild pollution).

## 3.5. Periphyton

Excessive growth of periphyton has the potential to affect food availability for trout and fishing amenity value (by snagging fishing tackle and reducing aesthetic values). At this time, the most applicable guidelines relating to periphyton biomass and cover for the protection of trout fishery values are those contained in Biggs (2000). However, these may need to be revised in the future in light of improved understanding of the inter-relationships between periphyton, invertebrate drift and trout growth and abundance. Although high densities of invertebrates (potential trout food) may be associated with high algal biomass, there is evidence that these invertebrates may not be as readily available to drift feeding trout (Shearer *et al.* 2003). This may be especially pertinent in the event that the invasive alga *Didymosphenia geminata* (didymo) becomes more widespread and established in the North Island.

While the periphyton guidelines for trout fisheries suggested in Biggs (2000) may be sufficient to protect fisheries values in lowland streams it is likely that algal biomass and cover at these levels would be seen as a significant reduction in the "pristine" natural character of many headwater fisheries. Headwater rivers generally have thin diatom films. In these rivers, the guidelines proposed by Biggs (2000) to protect benthic biodiversity values would arguably provide better protection of trout habitat, benthic invertebrate (food producing) habitat and aesthetic values, all of which contribute to these streams being recognised as "Outstanding trout fisheries".

## 3.6. Nitrogen

There are two ways in which elevated levels of nitrogen can potentially impact on trout; indirect effects, via algal proliferation, and direct toxic effects. Nitrogen concentrations need to be reasonably high before direct toxic effects become a concern. However, even at relatively low concentrations, nitrogen has the potential to indirectly affect trout by promoting the proliferation of algae (See Section 3.5 Periphyton).

There is evidence that trout eggs and alevins are sensitive to nitrogen in the form of both nitrate and ammonia. Bell (1986) stated that supersaturation of nitrogen above 104 ppm presents a danger to the development of salmonid eggs and newly hatched juveniles. A report into the toxicity of ammonia produced by the United States Environmental Protection Agency (USEPA 1999) cites two studies into the lethal effects of ammonia on salmonid eggs and alevins. They both found that high ammonia concentrations increased mortality, with one study reporting that survival was reduced by 67 % at a total ammonia nitrogen concentration of 2.55 mg /l. However, the effects of ammonia are dependent on pH and while this testing was carried out at a pH of 7.52, this concentration would be equivalent to 1.44 mg/L at pH 8 (USEPA 1999). Both of these studies also found that long term exposure reduced the concentration required to cause 50 % mortality, as did reduction in the interval between fertilisation and first exposure to ammonia.

It is not just the early life stages of trout that are sensitive to high levels of ammonia. For example the LC50 (concentration at which 50 % of a sample can be expected to die within a 96 hour exposure) for rainbow trout at pH 8 and 15 °C is approximately 11.23 mg/L (Jowett *et al.* 2004).

Another study, cited in a document produced by the State of California Regional Water Quality Control Board (SCRWQCB 2004), investigated the toxic effects of nitrate. It concluded that steelhead trout eggs suffer a statistically significant increase in the rate of mortality when subjected to nitrate concentrations above 1.1 mg/L. However, there appeared to be some doubt as to the reliability of this experiment, so further investigation may be warranted.

In our opinion, reducing the likelihood of detrimental periphyton proliferation should be the primary focus of these guidelines, since maintaining nitrogen levels low enough to achieve this goal will also avoid toxic effects. Ensuring that soluble inorganic nitrogen (SIN) levels comply with Biggs (2000) guidelines for the control of periphyton proliferation guidelines should be sufficient to protect trout fisheries values. However, we have also provided a guideline value for ammoniacal nitrogen from ANZECC (2000). If Horizons choose to apply the nutrient guidelines from Biggs (2000) only during periods of the year when other physical conditions are favourable for periphyton proliferation, then the ANZECC (2000) guideline could be applied to provide protection against lethal effects of ammonia for the rest of the year.

## 3.7. Phosphorus

Soluble reactive phosphorus (SRP) is most likely to indirectly affect trout, in combination with nitrogen, by promoting proliferation of algae (See Section 3.5 Periphyton).

In our opinion ensuring that SRP levels comply with Biggs (2000) guidelines for the control of periphyton proliferation should be sufficient to protect trout fisheries values.

## 3.8. pH

There is evidence that pH can mediate the effects of some toxicants on trout (*e.g.* Howarth & Sprague 1978, Cusimano *et al.* 1986). Also, Kwak & Waters (1997) found a strong positive correlation between salmonid production and alkalinity in North American streams. Raleigh *et al.* (1986) suggest the tolerable range of pH for brown trout is 5.0 to 9.5, with an optimal range of 6.7 to 7.8 (this is assumed to be apply to both adults and juvenile life stages).

In our opinion, maintaining pH within the circum-neutral range suggested in the ANZECC (2000) guidelines should avoid any adverse effects on trout. However, we recognise that it may be necessary to adapt these guidelines, in some instances, to take account of the influence of underlying geology.

## 3.9. Faecal contamination

We are not aware of any evidence to suggest that mammalian faecal contamination has any adverse effect on trout, although it is often associated with nutrient enrichment, which does potentially have adverse effects. However, trout fishing is a contact recreation. Therefore, we suggest that the Ministry for the Environment guidelines for contact recreation would be appropriate for streams managed for trout fishery values.

## 3.10. Other toxicants

Trout are generally located toward the more sensitive end of the continuum of sensitivity to toxic substances in the environment. For this reason we recommend that the 99 % protection level suggested for "other toxicants" in the ANZECC (2000) guidelines should provide for protection of trout fishery values.

# 4. REFERENCES:


ANZECC 1992. Australian water quality guidelines for fresh and marine waters. Australian & New Zealand Environment & Conservation Council.

ANZECC 2000. Australian and New Zealand guidelines for fresh and marine water quality. Australia and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand.

Barrett JC, Grossman GD, Rosenfeld J 1992 Turbidity-Induced Changes in Reactive Distance of Rainbow Trout. Transactions of the American Fisheries Society 121:437–443.

BCME 1997. Ambient water quality criteria for dissolved oxygen. Water Management Branch of the Lands and Headquaters Division, Ministry of Environment, Lands and Parks, British Colombia, Canada. http://www.env.gov.bc.ca/wat/wq/BCguidelines/do/do-03.htm (accessed on 24 July 2006)

Bell MC 1986. Fisheries handbook of engineering requirements and biological criteria. US Army Corps of Engineers, Portland, Oregon.

Biggs BJF 2000. New Zealand Periphyton Guideline: Detecting, monitoring and managing enrichment of streams. Prepared for Ministry for the Environment by Barry, J.F. Biggs, NIWA, Christchurch. Ministry for the Environment, Wellington. 122p.

Bjornn TC, Reiser DW 1991. Habitat requirements of salmonids in streams. In: Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats, edited by W.R. Meehan. American Fisheries Society, Bethesda, Md. Pp. 83-138.

Cusimano RF, Brakke DF, Champman GA 1986. Effects of pH on the toxicities of cadmium, copper and zinc to steelhead trout (Salmo gairdneri ). Canadian Journal of Fisheries and Aquatic Sciences 43: 1497-1503.

Davies-Colley RJ, Close ME 1990. Water colour and clarity of New Zealand rivers under baseflow conditions. New Zealand Journal of Marine and Freshwater Research 24: 357-365.

Davies-Colley RJ, Smith DG 2001. Turbidity, suspended sediment, and water clarity: a review. Journal of the American Water Resources Association 37: 1085-1101.

Dean TL, Richardson J 1999. Responses of seven species of native freshwater fish and shrimp to low levels of dissolved oxygen. New Zealand journal of marine and freshwater research 33: 99-106.

Doudoroff P, Shumway DL 1970. Dissolved oxygen requirements of freshwater fishes. United Nations Food and Agriculture Organisation, Technical Paper 86.

Elliott JM 1981. Some aspects of thermal stress in teleosts. In Pickering, A.D. (Ed.), Stress and fish. London: Academic Press. Pp. 209-245

Elliott JM 1994. Quantitative ecology and the brown trout. Oxford Series in Ecology and Evolution. Oxford University Press, Oxford.

Elliott JM, Elliott JA 1995. The effect of the rate of temperature increase on the critical thermal maximum for parr of Atlantic salmon and brown trout. Journal of fish biology 47: 917-919.

Gregory RS, Northcote TG 1993. Surface, planktonic, and benthic foraging by juvenile Chinook salmon (Oncorhynchus tshawytscha) in turbid laboratory conditions. Canadian Journal of Fisheries and Aquatic Sciences 50: 233-240.

Hayes JW 2000. Brown trout growth models: User guide Version 1.0. Cawthron Report No. 571. [Version 2.0 was used in this report – which includes updates for growth by piscivorous trout and for modelling growth on proportions of maximum rations and combinations of invertebrate and fish diets]

Hayes JW, Stark JD, Shearer KA 2000. Development and test of a whole-lifetime foraging and bioenergetics model for drift-feeding brown trout. Transactions of the American Fisheries Society 129: 315-332.

Hokanson KE, Kleiner CF, Thorslund TW 1977. Effects of constant temperatures and diel temperature fluctuations on specific growth and mortality rates and yield of juvenile rainbow trout, Salmo gairdneri. Journal of the Fisheries Research Board of Canada 34: 639-648.

Howarth RS, Sprague JB 1978. Copper Lethality to Rainbow Trout in Waters of Various Hardness and pH. Water Research 12: 455-462.

Hughes NF, Dill LM 1990. Position choice by drift-feeding salmonids: model and test for Arctic grayling (Thymallus arcticus) in subarctic mountain streams, interior Alaska. Canadian Journal of Fisheries and Aquatic Sciences 47: 2039-2048.

Humpesch UW 1985. Inter- and intra-specific variation in hatching success and embryonic development of five species of salmonids and Thymallus thymallus. Archiv für Hydrobiologie 104: 129-144.

Jowett IG 1992. Models of the abundance of large brown trout in New Zealand rivers. North American Journal of Fisheries Management 12: 417-432.

Jowett IG 1997. Environmental effects of extreme flows. In: Mosley, MP, Pearson, CP (Eds) Floods and droughts: the New Zealand experience. New Zealand Hydrological Society, Caxton Press, Christchurch. Pp. 103-116.

Jowett IG, Kingsland S, Collier KJ 2004. WAIORA User Guide – Version 2.0. National Institute of Water & Atmospheric Research Ltd., Hamilton.

Kwak TJ, Waters TF 1997. Trout production dynamics and water quality in Minnesota stream. Transactions of the American Fisheries Society. Vol. 126: 35-48.

May BE 1973. Seasonal depth distribution of rainbow trout (Salmo gairdnerii) in Lake Powell. Proceedings of the Utah Academy of Science Articles and Letters 50: 64-72.

Newcombe CP, Jensen JO 1996. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16: 693-727.

Newcombe CP, MacDonald DD 1991. Effects of suspended sediments on aquatic ecosystems. North American Journal of Fisheries Management 11:72-82.

Phillips C, Basher L 2005. Catchment channel characteristics and riverbed substrate assessment – a review and trial of a method of fine sediment assessment in the Motueka River. Motueka Integrated Catchment Management Programme Report Series, Landcare ICM Report No. 2004-05/01, Landcare, Lincoln. http://icm.landcareresearch.co.nz/library/project_documents/ICM_2004-05_01.pdf (accessed on 28 July 2006)

Raleigh RF, Hickman T, Solomon RC, Nelson PC 1984. Habitat suitability information: rainbow trout. U.S. Fish and Wildlife Service FWS/OBS-82/10.60.

Raleigh RF, Zuckerman LD, Nelson PC 1986. Habitat suitability index models and instream flow suitability curves: brown trout, revised. U.S. Fish and Wildlife Service Biological Report 82 (10.124).

Rowe DK, Dean TL, Williams E, Smith JP 2003. Effects of turbidity on the ability of juvenile rainbow trout, Oncorhynchus mykiss, to feed on limnetic and benthic prey in laboratory tanks. New Zealand journal of marine and freshwater research 37: 45–52.

Ryan PA 1991. Environmental effects of sediment on New Zealand streams: a review. New Zealand Journal of Marine and Freshwater Research 25:207-221.

Scott D, Poynter M 1991. Upper temperature limits for trout in New Zealand and climate change. Hydrobiologia 222: 147-151.

SCRWQCB 2004. Staff report for regular meeting on February 6, 2004. State of California Regional Water Quality Control Board,Central Coast Region. URL: http://www.swrcb.ca.gov/rwqcb3/TMDL/documents/CHLONutDOSRforFeb04.doc accessed on 18 November 2004.

Shearer KA, Stark JD, Hayes JW, Young RG 2003. Relationships between drifting and benthic invertebrates in three New Zealand rivers: implications for drift-feeding fish. New Zealand Journal of Marine and Freshwater Research 37: 809-820.

Stark JD 1985. A macroinvertebrate community index of water quality for stony streams. Water & Soil miscellaneous publication 87: 53 p.

Sweka JA, Hartman KJ 2001. Effects of turbidity on prey consumption and growth in brook trout and implications for bioenergetics modeling. Canadian Journal of Fisheries and Aquatic Sciences 58: 386-393.

USEPA 1999. 1999 update of ambient water quality criteria for ammonia. Office of Water, U.S. Environmental Protection Agency, Washington D.C. EPA-822-R-99-014. URL: http://www.epa.gov/ost/standards/ammonia/99update.pdf accessed on 18 November 2004.

Wood PJ, Armitage PD 1997. Biological effects of fine sediment in the lotic environment. Environmental Management 21:203-217.