# Preliminary information for developing sediment guidelines for streams of the West Coast, New Zealand

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# Preliminary information for developing sediment guidelines for streams of the West Coast, New Zealand

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# 1. Introduction

## 1.1 Suspended and deposited sediment impacts

Increased sedimentation due to landuse changes in surrounding catchments is a major form of pollution in streams. Some suspended particulate matter arises from point sources such as sewage outfalls, mining, industrial wastes and stormwater drains, but most is contributed from diffuse land runoff due to soil erosion (ANZECC, 2000). Sediment may be deposited on stream beds or remain in suspension. Most of the suspended sediment is <2 mm (Owens et al., 2005), with suspended particle size distribution dependent on flow velocities and source characteristics. Suspended sediments are often regarded as the single most important pollutant of freshwaters, in terms of the quantities discharged and the damage that they cause to aquatic ecosystems (Henley et al., 2000; Owens et al., 2005). The functioning and productivity of streams can be altered by suspended sediment, which can reduce photosynthesis of in-stream autotrophs, clog gills and filter feeding structures of certain faunal species, increase invertebrate drift, and alter the behaviour of fish (Ryan, 1991; Waters, 1995; Wood and Armitage 1997; Henley et al., 2000). Some fish may avoid streams with high suspended sediment (Jowett et al., 1996), reducing the value of these streams as recreational fisheries. The reduction in clarity due to excess suspended sediment also adversely affects the aesthetics of streams, reducing their value as recreation resources (Davies-Colley et al., 1993).

Depending on the amount contributed to streams, deposited sediment can block interstitial spaces among larger substrates, reduce availability of stable attachment sites for biota, cover fish spawning sites, reduce exchange of oxygen and metabolic wastes for benthic biota, interfere with feeding by covering and reducing the quality of food sources, or scour and smother the animals themselves (Ryan, 1991). Excess sediment discharged to marine environments can also have adverse effects on the communities in these ecosystems (Thrush et al., 2004). The impact on aquatic biota depends on the species and lifestages present in communities, and the concentration and duration of exposure (Newcombe and MacDonald, 1991). Continuous high level inputs of sediment are likely to have most deleterious effects on aquatic communities, as some sediment input is natural and necessary for ecosystems, and animals are presumably adapted to cope with smaller pulsed inputs similar to those that occur naturally (Ryan, 1991).

## 1.2 West Coast regional sediment guidelines

The West Coast Regional Council (WCRC) deals with many activities that increase sediment discharges to streams, and requires meaningful, scientifically-based criteria to regulate sediment discharges from consented activities. Setting ecoregion-specific standards has been the approach for the development of sediment guidelines in New Zealand and Australia (ANZECC, 2000), as well as the U.S. (USEPA, 2000; 2006) and Canada (Culp et al., 2009). The main consented activities conducted in the West Coast are alluvial and open cast gold mining, underground and open cast coal mining, and forestry. For each resource consent the council must determine appropriate conditions, which can be complicated due to: variability in the type and intensity of each activity; natural variability of the physical environment, including topography, geology, and soil type; and, the range of other instream activities (e.g., swimming, recreational fishing, rafting, kayaking) that may be impacted by upstream landuse intensification. Currently, each resource consent is dealt with on a case by case basis, with some variability in consents for similar activities (pers. comm. J. Horrox, West Coast Regional Council). This variability in consent conditions may be reduced by a clearer understanding of appropriate thresholds based on the most relevant and up to date science.

## 1.3 Objectives of this document

The information contained in this document is provided to assist in the development of regulations to control sediment discharges that balance protection of the environment within pragmatic social and economic constraints. However, the judgement on standards that balance protection of the environment within other constraints must lie with the Regional Council, because NIWA is not in a position to judge where this balance lies. This report aims to summarise the latest scientific knowledge and outline sound statistical principles that should be used when setting thresholds and evaluating environmental impacts of elevated sediment discharges.

# 2. Previous studies of sedimentation impacts in New Zealand

## 2.1 Impacts of sedimentation from West Coast mining

Alluvial mining can produce continuous high concentrations of point source sediment inputs (Ryan, 1991). Alluvial gold mining (also termed placer mining) occurs in New Zealand, as well as interior Canada, and Alaska (Winterbourn and Ryan, 1994; Milner and Piorkowski, 2004). Few studies have been conducted in New Zealand streams to determine impacts of sediment associated with mining. Further, it is often difficult to separate the effects of increased sedimentation from those of other pollutants due to intensified landuse (Ryan, 1991). However, studies conducted in the early 1990s on West Coast streams impacted by placer mining were able to examine impacts of elevated suspended sediments because, unlike other forms of mining, the main impact of placer mining is almost exclusively due to elevated suspended sediment (Davies-Colley et al., 1992; Quinn et al., 1992). Other studies of mining impacts in New Zealand have largely focused on acid mine drainage and heavy metals (see review by Harding and Boothroyd, 2004).

In West Coast streams on the plains between the Taramakau River Valley and Hokitika, Davies-Colley et al., (1992) and Quinn et al., (1992) examined the effects of clay inputs from placer gold mining on optical properties and epilithon, and benthic macroinvertebrate communities, respectively. The clay severely reduced optical quality, with visual clarity reduced from a few metres (typically ~2 m) to between 0.03 and 0.66 m (median 0.33 m). Turbidity increased from a median of 2.4 NTU (nephelometric turbidity units) upstream of mining often to >100 NTU (median 15 NTU) downstream, with highly variable turbidity below mining activities (Davies-Colley et al., 1992). The changes downstream of mines equated to reductions in stream-bed lighting of 12-73% (mean 44%) of upstream levels. Reduction in light due to the suspended sediment reduced benthic primary production, benthic algal biomass and phototrophic content of epilithon downstream of mining activity (Davies-Colley et al., 1992). Turbidity increases of as little as 9 NTU were shown to reduce algal biomass by as much as 40 percent (Davies-Colley et al., 1992). The organic content of epilithon was also reduced due to fine sediment deposition, lowering the quality of this food source for invertebrate primary consumers (Davies-Colley et al., 1992).

Densities of invertebrates downstream of mining activities were negatively correlated with the logarithm of the turbidity loading ($r$ = -0.82), with densities at downstream sites ranging from 9 to 45% (median 26%) of those at matched upstream sites (Quinn et al., 1992). These reductions in invertebrate densities were associated with as little as 7 NTU increase in turbidity above background (Quinn et al., 1992). Taxon richness was significantly lower at four sites that had mean turbidity increases between 23 and

154 NTU (Quinn et al., 1992). Reduced invertebrate densities below mining activities may have been due to a combination of lower epilithon biomass and productivity, degraded food quality, reduced bed permeability and interstitial dissolved oxygen, and increased downstream drift (Quinn et al., 1992). Total invertebrate density provided a better indicator of sediment pollution than either changes in taxon richness or densities of particular species (Quinn et al., 1992).

## 2.2 Other studies of sedimentation impacts on communities in New Zealand streams

There have been a range of surveys and experiments conducted in New Zealand, which have demonstrated impacts of sediment on stream ecosystems. The major findings of some of these studies are presented below:

In the Tongariro River, a natural sand input from Rangipo Desert increased the amount of sand cover deposited on the streambed from 0% upstream to 7-16% downstream (measured by a Wolman (1954) pebble count). Increased sand cover did not affect taxonomic richness, but favoured some chironomid species and restricted the densities of the mayfly *Deleatidium* (Quinn and Vickers, 1992). Suspended inorganic sediment values (measured using the “Quorer” methodology, see http://www.niwa.co.nz/our-science/freshwater/tools/quorer) were negatively correlated with both *Deleatidium* densities and QMCI (Quinn and Vickers, 1992). However, these sand deposits did not limit water flow or oxygen supply within the stream bed (Quinn and Vickers, 1992).

Townsend et al., (2008) conducted experiments involving the addition of fine sediment to stream reaches in Otago. They suggested that an increase in the loading of fine deposited sediment (from 36% cover prior to experiments, up to 83% after additions) had more widespread effects on stream ecosystems than augmented nutrient concentrations. Also in Otago, Matthaei et al., (2006) added fine river sand to stream reaches surrounded by ungrazed tussock grassland, grazed pasture, dairying or deer farming. They found that prior to sediment additions the cover of fine deposited sediment was lowest in tussock (7%), intermediate in pasture and dairying (30% and 47%, respectively) and highest in deer farming streams (88%). Sediment additions resulted in reduced cover of mosses, reduced taxonomic richness and EPT richness relative to control reaches, with the impact most severe in pasture streams where pre-treatment fine sediment cover was moderate and the richness and diversity of the invertebrate community highest (Matthaei et al., 2006). Matthaei et al., (2010) found that sediment additions to experimental channels (addition of fine sand, up to 80% cover) had the largest negative effect on algal biomass, invertebrate abundance and taxonomic richness in experimental manipulations with nutrient enrichment, fine sediment addition and water abstraction. It was also found that there were often interactions among the stressors, with interactions between increased sediment and reduced flow particularly common (Matthaei et al., 2010).

Some New Zealand studies have directly examined the influence of increased sediment loads on fish and invertebrate behaviour. High turbidity (>20 NTU) limited the upstream migration and recruitment of juvenile banded kokopu (a whitebait species) in New Zealand streams, whilst the low abundance of redfin bully in rivers with high suspended sediment concentrations was likely due to siltation of benthic habitats (Richardson et al., 2001; Rowe et al., 2009). Suren and Jowett (2001) found increased macroinvertebrate drift in cobble lined experimental channels to which 12 kg $m^{-2}$ of fine sediment was added.

The aforementioned studies demonstrate significant ecological impacts in New Zealand streams, and levels at which these impacts occur, but impacts may also occur at levels lower than those tested.

## 2.3 Sediment impacts on recreational values

When setting guidelines on limits to a pollutant, the potential amenity values (e.g., fishing or swimming) also influence what is acceptable in a water body, In some situations aesthetic considerations may be considered more important than ecological issues (Ryan, 1991). The visual clarity of water is an important consideration because it affects the recreational and aesthetic quality of water (MfE, 1994; ANZECC, 2000), as well as plant growth and animal behaviour. For swimmers, visual clarity is important for judging depth and sighting subsurface hazards. In normally clear water a clarity reduction of 10-15% is noticeable, whilst in water that is already turbid a 20-50% reduction in clarity is distinguishable (Ryan, 1991). Davies-Colley (1988) suggested that a 20-50% reduction in clarity may be the detectable threshold for the human eye when the change in colour due to suspended sediment is the only cue, and this should be the starting point for setting standards in streams where aesthetics are important. A horizontal black disc sighting of at least 1.2 m, corresponding to Secchi disc depth of 1.5 m, is required before water is generally perceived as acceptable for bathing (increasing to a horizontal black disc sighting of 2.2 m if 90% of people perceiving water quality as suitable for bathing) (Davies-Colley, 1988). Ministry for the Environment Guidelines recommend that to maintain visual clarity relevant to swimmer safety in wadeable areas, the black disc visibility should be not less than 1.6 m (Davies-Colley, 1994). Acceptable thresholds may vary depending on changes in use at different times of the year. For example, in the early 1990s in Nelson’s Creek, a popular location for swimming in summer, the West Coast Regional Council allowed a 2 NTU increase over ambient levels during warmer months (November to March, inclusive) and 10 NTU during the remainder of the year in recognition of values of the creek varying at different times of the year (Ryan, 1991).

# 3. Existing guidelines for quantities of sediment in streams

There have been very few studies aimed at examining ecological thresholds or developing standards for either suspended or deposited sediment. However, in New Zealand the Resource Management Act 1991 (RMA) and ANZECC guidelines (ANZECC, 2000) provide guidance on criteria that may be useful when developing region-specific standards.

## 3.1 Resource Management Act 1991

The RMA has increased the need for resource users to provide information on the environmental effects of their activities. The RMA does not directly legislate for sediment quantity, but it enables region-specific guidelines to be developed. Specific guidance related to water clarity in the RMA requires that activities on land, and discharges to waters be managed so that, after reasonable mixing with the receiving water, there is no "conspicuous change in the colour or visual clarity" (Sections 70 and 107, mandatory). Furthermore, in waters classified for contact recreation "the visual clarity of the water shall not be so low as to be unsuitable for bathing" (Schedule 3). The Ministry for the Environment (MfE, 1994) provides a reduction of up to 33 - 50% as the guideline for protection of water clarity from *point source* discharges. Interpretation of effects on visual clarity data for *diffuse source* effects is not addressed specifically in the MfE guidelines (MfE, 1994). However, in analysing whether water clarity effects from forest harvest operations comply with Sections 70 and 107, Wright-Stow et al., (2010) argued that assessment of changes in annual median values relative to pre-harvest levels is a reasonable approach, given the variable nature of water clarity with flow. This approach has been presented in Environment Court evidence (Quinn, 2006) without challenge.

Compliance with the MfE (1994) guidelines for compliance with RMA Sections 70 and 107 requirements is quite restrictive in relatively clear waters where relatively small increases in suspended solids reduce water clarity by more than 50%. For example, in Davies-Colley et al.,'s (1992) study of placer mine effects on West Coast streams, the smallest median downstream increase in total suspended solids (TSS) – 8.5 g $m^{-3}$ over a background of 0.3 g $m^{-3}$ – reduced black disc visibility by 80% (i.e., from 3.1 m to 0.62 m). Data in Figure 4 of Davies-Colley et al., (1992) indicate that TSS increases due to clay discharge would need to be restricted to below approximately 2 g $m^{-3}$ to avoid reductions in black disc of >50% in such clear streams. This suggests that avoiding aesthetic effects by applying the MfE (1994) guideline to protect against conspicuous clarity effects will also protect against ecological impacts.

## 3.2 Australian and New Zealand Environment and Conservation Council (ANZECC) guidelines

The overarching objective of the Australian and New Zealand Guidelines for Fresh and Marine Water Quality guidelines (developed by ANZECC) is to define water quality objectives required to sustain current, or likely future, environmental values for water resources (ANZECC, 2000). The ANZECC guidelines include information for visual clarity, but no specific information for setting limits on deposited sediment. The guidelines state that to protect the aesthetic quality of streams the natural visual clarity should not be reduced by more than 20% (based on studies in New Zealand streams by Davies-Colley, 1991), whilst to protect the visual clarity of water used for swimming, the horizontal sighting of a (200 mm) black disc should exceed 1.6 m (which is also the aesthetic standard provided by USEPA, 2006). In areas used for diving, the water clarity would need to be considerably greater than this (ANZECC, 2000).

## 3.3 West Coast Regional Council fine sediment recommendations and standards cited in publications

In the early 1990s, West Coast Regional Council imposed discharge standards on alluvial miners that allowed increases of 10 mg $L^{-1}$ suspended solids, or 10 NTU turbidity, in small streams outside a 200 m mixing zone (Ryan 1991). Based on their research on the West Coast, Quinn et al., (1992) recommended that average increases be limited to <5 mg $L^{-1}$ suspended sediments or turbidity to <5 NTU to prevent substantial impacts on invertebrate communities of West Coast streams. These limits applied to non-flood conditions. In the instance of a major gold dredge operation adjacent to the regionally significant Grey River, the West Coast Regional Council applied a sliding scale of allowable turbidity increase, depending on background conditions: requiring that turbidity increase be less than 3.2 NTU when ambient turbidity was below 2 NTU, whereas an increase up to 60% was allowed when ambient turbidity was 2 – 16 NTU above which the allowable increase was capped at 10 NTU (Ryan, 1991).

## 3.4 Sediment guidelines from the U.S. and Canada

Overseas water quality guidelines generally specify a maximum suspended solids concentration. Some of these recommended concentrations are summarised below, with the reader referred to documents by the USEPA (2006) and Culp et al., (2009) for more information. A full review of these recent detailed documents is beyond the scope of the present Envirolink project, but they provide a valuable source of information that may be useful for development of region-specific sediment guidelines if due consideration is given to local influences.

From agricultural catchments in Canada, Culp et al., (2009) provide best estimates of sediment thresholds below which good environmental conditions are maintained, with values varying depending on which attribute was being protected. For maintenance of physical values, thresholds ranged from 1.5 to 8.0 mg $L^{-1}$ for total suspended solids (TSS) and from 0.3 to 4.0 NTU for turbidity, whilst for maintenance of biotic values the thresholds were slightly higher, i.e., 13 mg $L^{-1}$ and 8 NTU (Culp et al., 2009). These authors suggest that seasonal or annual averaging is most appropriate (rather than requiring that values should fall within these limits on every sampling occasion).

The majority of U.S.A states use turbidity and TSS as indicators of suspended sediment, although there is considerable variability in the threshold values provided for each region (see USEPA, 2006). Some guidelines are in the form of exceedances above background (e.g., not more than 10% above background, or not more than 10 NTUs above background), whilst other criteria define absolute values (e.g., not greater than 100 NTU) (USEPA, 2006). Further, some states also have numeric deposited sediment standards based on different bed substrate criteria, with a range of thresholds depending on particle sizes: e.g., 1) maximum of 25-30% of surface substrate smaller than fine gravel (~6.4 mm); 2) maximum 10-15% of surface substrate smaller than coarse sand (~0.85 – 2 mm); 3) maximum 20% of fine sediment (<2 mm) in riffles; 4) maximum 20 – 27% subsurface fine sediment by mass; 5) maximum substrate embeddedness of 25 – 33 %; and, 6) median particle size should increase towards 69 mm, and not be below 37 mm, over the long term (Culp et al., 2009). However, it is recognised that standards should be region-specific. Sediment particle sizes vary naturally within individual catchments with factors that affect stream power, such as discharge, slope, frequency and magnitude of flushing flows, and sediment supply.

# 4. Methods for measurement and statistical analyses

## 4.1 Deriving standards or trigger values

Numeric standards are best determined by dose-response studies along natural or artificial gradients to find thresholds/tipping points and safe levels. In the absence of such data, “trigger values” for non-toxic stressors, including TSS and turbidity, are usually determined by examining naturally occurring background values, or variability around these values, to establish critical reference thresholds within an eco-region (Culp et al., 2009). A common approach for defining trigger values above which ecological impacts may occur, whilst accounting for natural variability, is to select a particular percentile of naturally occurring values (see ANZECC, 2000; USEPA, 2006). The ANZEEC (2000) guidelines use the 80$^{th}$ percentile for distributions from least disturbed reference streams in an ecoregion. The choice of this percentile is arbitrary and considered a reasonably conservative threshold between background and potentially impaired condition for stream ecosystems. Ultimately, the method used to define thresholds will depend upon the ecosystem type, the desired level of protection, and the availability of suitable reference systems and adequate data for these systems (ANZECC, 2000; USEPA, 2006).

## 4.2 Correlations between various direct and indirect measures of suspended sediment

The amount of suspended sediment in water influences water clarity and turbidity. TSS is a measure of the mass of inorganic particles suspended in a water sample, whereas turbidity is a relative measure of the amount of scattering of light by particles (Davies-Colley and Smith 2001). Clarity is a measure of how clear or transparent water is and can be measured as the horizontal visibility of a black disc ($y_{\mathrm{BD}}$, Davies-Colley, 1988) that is related directly to the water’s light attenuation coefficient, a fundamental optical property measured by a beam transmissometer. Measurements of horizontal and vertical black disc visibility (i.e., $y_{\mathrm{BD}}$ and $z_{\mathrm{BD}}$) can be used to calculate the water’s attenuation coefficient ($K_d$) for photosynthetically available radiation (PAR) and hence to calculate how much light is available for photosynthesis at different depths (ANZECC, 2000). Reduced clarity is caused by the presence of suspended particulate and colloidal matter consisting of suspended clay, silt, phytoplankton and detritus. The amount, size, shape and composition of the suspended matter will affect turbidity and clarity measurements. Water clarity is also influenced by the amount of dissolved colour (aquatic humus or “yellow substance”), such as the beech forest leachate that gives many West Coast streams their brown colour.

Davies-Colley et al., (1992) reported close correlations between the log-transformed measures of turbidity and TSS ($r$ = 0.94), turbidity and black disc visibility ($r$ = -0.96), and TSS and black disc visibility ($r$ = -0.94) in West Coast streams, so that any one of these three closely interrelated variables can probably be used to define the loading of suspended inorganic sediment from mining in the region (Davies-Colley et al., 1992). However, the correlation should be checked for each water body, as the relationships can be affected by numerous confounding factors, including the colour of the water and the size of suspended particles.

In Canadian streams from numerous regions, the correlations between turbidity and TSS, both log-transformed, were highly variable, ranging from 0.37 to 0.95 (Culp et al., 2009). This high variability between regions likely reflects different particle size characteristics of the suspended material. For example, the correlation between turbidity and TSS would be expected to be closer in streams where the suspended material is dominated by clays and/or silts, such as for West Coast streams impacted by placer mining (Davies-Colley et al., 1992), than for streams where coarser sediments were present in suspension.

## 4.3 Survey and statistical design

Optimal monitoring approaches (including methodology and frequency of surveys) will vary depending on the nature of inputs and values of each impacted waterbody. Given the staff resource constraints within the Regional Council and the need to manage the costs of compliance monitoring, we recommend that consented activities generally should be monitored by the consent holder, with quality assurance checks by an independent group.

Monitoring frequency should reflect the potential magnitude of the effects of non-compliance, the timescale over which compliance is intended to be assessed, and seasonal influences. For example, continuous monitoring using turbidity sensors is recommended where potential impacts of non-compliance are rated high, whereas weekly to monthly black disc measurements are recommended where potential impacts are moderate to low. Monitoring results should be reported regularly (e.g., quarterly to biannually, depending on the risks). Clarity/turbidity/TSS measurements should be indexed against flow or water level to aid data/trend interpretation and/or allow interpretation of sediment loads. Ideally, flow or water level would be measured on the same stream, but reference to a flow recorder on a nearby stream with similar hydrological response (e.g., elsewhere in the catchment or in an adjacent catchment) will often be sufficient. Adverse impacts from excess sedimentation associated with intensified landuse are usually greatest at low flow (Ryan, 1991), and most recreational activities also occur at baseflows. However, it is also important to ensure that monitoring programs include storm events. Most (70–90%) suspended sediment is transported during high-flow events and, if the load or flux of suspended sediment is required, it is particularly important that TSS concentrations and discharge are measured during these high-flow events (ANZECC, 2000). Consideration of loads is particularly important when streams discharge to potentially sensitive marine areas, wetlands or lakes (e.g., Lake Brunner).

Single spot measurements of turbidity and suspended sediment concentration are of little value in terms of understanding dynamic processes in ecosystems (ANZECC, 2000). Relatively inexpensive devices for the continuous *in situ* measurement of turbidity are now available, and the continuous measurement of turbidity is now being incorporated into a number of water quality monitoring programs. In the future, this will result in improved turbidity databases that will be useful for improving our knowledge of the variability of turbidity, thus providing the basis for better defining statistically-based guidelines. Turbidity meters should be maintained and their accuracy cross-checked by regular (e.g., monthly to quarterly) quality assurance (QA) visits during which black disc visibility should be recorded and/or a water sample collected. Water samples from each QA visit should also be kept in storage, and a random subsample of these analysed for TSS and/or turbidity by an independent party to conduct periodic checks on the validity of data being reported by the consent holder.

### 4.3.1 Biological surveys

Periodic biological surveys should be conducted to ensure there are no adverse impacts on community structure from point source sediment discharges (see Culp et al., 2009). Ideally, sampling would follow a before-after-control-impact design (BACI) (Green 1984), with at least a year's data at the impact site before the activity begins. We recommend use of benthic macroinvertebrate communities as indicators of biological response. These communities are relatively easy to monitor, play important ecological roles in stream ecosystem function, and have been shown to be sensitive to fine sediment impacts in previous West Coast studies (Quinn et al., 1992). We recommend quantitative invertebrate sampling (e.g., using Surber samplers) where valid comparisons can be made above and below discharges because total density and density of sensitive species (e.g., the mayfly *Deleatidium*) were found to be more sensitive indicators of fine sediment impact than taxon richness in previous research in West Coast streams (Quinn et al., 1992).

Such surveys could be conducted six-monthly, annually, or in response to high discharge events. Sample replication is necessary to assess the statistical significance of differences in invertebrate communities upstream and downstream of discharges. For one-off surveys (e.g., in response to events or complaints) we recommend collection and individual analysis of 3-7 replicate quantitative samples in matched habitats (ideally in "runs"). Three samples are likely to be sufficient where there is obvious evidence of impact, whereas 7 replicates should detect more subtle effects (Quinn et al., 1992). In long-term surveys (e.g., regular 6 monthly monitoring), the quantitative samples can be composited to make a single sample for analysis to reduce processing time. This precludes statistical analysis of control-impact differences on single occasions (due to lack of replication), but may allow longer term statistical tests of impacts (e.g., as paired t-tests or trends in the difference between control and impact sites).

In some streams it may be appropriate to focus surveys of clarity and/or biota to assess potential impacts during critical periods of key species' lifestages, e.g., spawning season and migrations in streams highly valued for recreational fishing (e.g., trout or whitebait).

A current EnviroLink Tools project is developing protocols for assessment of deposited fine sediment (<2 mm) in wadeable streams and is due to report in late 2011. Methods that are likely to be included in the final protocols include: (1) visual assessment of percentage fine sediment cover using an underwater viewer at multiple random points (likely 20) within a study reach; (2) classification of particle size (b-axis Wentworth classes) of 100 randomly selected surficial sediment particles after Wolman (1954) to provide quantitative assessment of percentage fine sediment; and (3) suspendable inorganic sediment measured as mass or settled volume ("Quorer" method, Quinn et al., 1997). All of these methods are likely to be useful for compliance monitoring in wadeable, gravel bed rivers on the West Coast. Methods 1 and 3 involve replication that allows statistical comparison of percentage cover by fine sediment amongst individual sites/occasions. Natural levels of surficial fine sediments will vary at sites in relation to channel slope, flow regime and sediment supply, so it is important to compare results with those from matched reference sites – again BACI designs are ideal.

### 4.4 Gaps in knowledge of sedimentation impacts on ecosystems

Given the paucity of data for sediment thresholds (both suspended and deposited) at which significant ecological impacts occur, any current guidelines should be considered provisional pending further information. As indicated throughout this document, there is considerable ecoregional variability in the amounts and impacts of stream sedimentation, and information from studies in other latitudes, climates, landscapes, and underlying geology should be used with due care. High flows in mountainous West Coast streams may act to flush out deposited sediments more readily than would occur in spring fed streams, making them potentially less susceptible to long-term impacts from fine sediment deposition. However, some West Coast streams are naturally highly turbid, and therefore may already be relatively close to turbidity thresholds.

## 4.5 Summary of recommendations on sediment standards

The following summary outlines our suggestions for guidelines to control sediment discharges for protection of the environment as required in the RMA to provide a basis for West Coast Regional Council to balance these within pragmatic social and economic constraints in their region.

### 4.5.1 Aesthetic guidelines

We consider the MfE (1994) guidelines of limiting reductions in water clarity to <33-50% due to discharges to water, to meet the requirements of Sections 70 and 107 of the RMA, to be robust for managing aesthetic effects of sediment discharge to streams. Compliance for point source discharges can be measured by comparison of black disc visibility at sites immediately above a discharge and below a defined mixing zone. Where black disc measurement is not possible, compliance can be assessed using turbidity or TSS data converted to black disc using locally derived correlations such as in Davies-Colley et al., (1994). We recommend using the median change to evaluate compliance with this guideline. We recommend application of the <33% change guideline in recreationally significant waterbodies and the <50% change guideline elsewhere.

For diffuse effects of land use (e.g., forest harvest, pastoral land development) we recommend assessment of change in median clarity against pre-activity data over a suitable time scale (e.g., annual median of monthly samples for forestry activities in medium–large catchments).

### 4.5.2 Ecological effects on streams of sediment discharge

#### Suspended fine sediment

We recommend following Quinn et al., (1992), limiting the average increase in TSS to <5 g $m^{-3}$ or turbidity to <5 NTU to protect macroinvertebrate abundance in gravel bed West Coast rivers. If the aim is to protect taxa richness, but not abundance, then evidence in Quinn et al., (1992) suggests <20 NTU increase above reference would be an appropriate limit.

#### Settled fine sediment

We suggest increases in deposited fine sediment of <15% above background levels prior to impact or at a hydraulically matched reference site as an interim guideline. The current EnviroLink Tools project is currently collating information on fine sediment levels and macroinvertebrates, and is likely to provide more definitive guidelines for settled fine sediment.

# 5. References

Australian and New Zealand Environment and Conservation Council (ANZECC) (2000). Australian and New Zealand guidelines for fresh and marine water quality. Agricultural and resources Management Council of Australia and New Zealand (ARMCANZ). Canberra, Australia.

Culp, J.M.; Benoy, G.A.; Brua, R.B.; Sutherland, A.B.; Chambers, P.A. (2009). Total suspended sediment, turbidity and deposited sediment standards to prevent excessive sediment effects in Canadian streams. *National Agri-Environmental Standards Initiative synthesis report No. 13.* Environment Canada. Gatineau, Canada.

Davies-Colley, R.J. (1988). Measuring water clarity with a black disc. *Limnology and Oceanography 33*: 616–623.

Davies-Colley, R.J. (1991). Guidelines for optical quality of water and for protection from damage by suspended solids. *Consultancy Report No 6213/1*. Water Quality Centre, Hamilton, New Zealand.

Davies-Colley, R.J.; Hickey, C.W.; Quinn, J.M.; Ryan, P.A. (1992). Effects of clay discharges on streams. 1. Optical properties and epilithon. *Hydrobiologia 248*: 215–234.

Davies-Colley, R.J.; Vant, W.N.; Smith, D.G. (1993). Colour and clarity of natural waters. Ellis Horwood, New York, USA.

Davies-Colley, R.J.; Smith, D.G. (2001). Turbidity, suspended sediment, and water clarity: a review. *Journal of the American Water Resources Association 37(5)*: 1085–1101.

Harding, J.; Boothroyd, I. (2004). Impacts of mining, in *Freshwaters of New Zealand* Harding, J.; Mosley, P.; Pearson, C; Sorrell, B. (eds). The Caxton Press. Christchurch, New Zealand.

Henley, W.F.; Patterson, M.A.; Neves, R.J.; Lemly, A.D. (2000). Effects of sedimentation and turbidity on lotic food webs: a concise review for natural resource managers. *Reviews in Fisheries Science 8*: 125–139.

Jowett, I.G.; Richardson, J.; McDowall, R.M. (1996). Relative effects of in-stream habitat and land use on fish distribution and abundance in tributaries of the Grey River, New Zealand. *New Zealand Journal of marine and Freshwater Research 30*: 463–475.

Matthaei, C.D.; Weller, F.; Kelly, D.W.; Townsend, C.R. (2006). Impacts of fine sediment addition to tussock, pasture, dairy and deer farming streams in New Zealand. *Freshwater Biology 51*: 2154–2172.

Matthaei, C.D.; Piggott, J.J.; Townsend, C.R. (2010). Multiple stressors in agricultural streams: interactions among sediment addition, nutrient enrichment and water abstraction. *Journal of Applied Ecology 47*: 639–649.

Ministry for the Environment (MfE) (1994). Water quality guidelines No. 2: Guidelines for the management of water colour and clarity. Ministry for the Environment. Wellington, New Zealand.

Milner, A.M.; Piorkowski, R.J. (2004). Macroinvertebrate assemblages in streams of interior Alaska following alluvial gold mining. *River Research and Applications 20*: 719–731.

Newcombe, C.P.; MacDonald, D.D. (1991). Effects of suspended sediments on aquatic ecosystems. *North American Journal of Fisheries Management 11*: 72–82.

Owens, P.N.; Batalla, R.J.; Collins, A.J.; Gomez, B.; Hicks, D.M.; Horowitz, A.J.; Kondolf, G.M.; Marden, M.; Page, M.J.; Peacock, D.H.; Petticrew, E.L.; Salomons, W.; Trustrum, N.A. (2005). Fine-grained sediment in river systems: environmental significance and management issues. *River Research and Applications 21*: 693–717.

Quinn, J. (2006). Statement of evidence in the matter of RMA appeals between Ernslaw One Ltd, Whangapoua Environmental Protection Society, DOC, and Environment Waikato. Heard before the Environment Court at Paeroa on 11–15 December 2006.

Quinn, J.M.; Davies-Colley, R.J.; Hickey, C.W.; Vickers, M.L.; Ryan, P.A. (1992). Effects of clay discharges on streams. 2. Benthic Invertebrates. *Hydrobiologia 248*: 235–247.

Quinn, J.M.; Vickers, M.L. (1992). Benthic invertebrates and related habitat factors in the Tongariro River. *Consultancy Report No 6025/2*. Water Quality Centre, Hamilton, New Zealand.

Quinn, J.M.; Cooper, A.B.; Davies-Colley, R.J.; Rutherford, J.C.; Williamson, R.B. (1997). Land use effects on habitat, water quality, periphyton, and benthic invertebrates in Waikato, New Zealand, hill-country streams. *New Zealand Journal of Marine and Freshwater Research 31(5)*: 579–597.

Richardson, J.; Rowe, D.K.; Smith, J.P. (2001). Effects of turbidity on the migration of juvenile banded kokopu (*Galaxias fasciatus*) in a natural stream. *New Zealand Journal of Marine and Freshwater Research 35*: 191–196.

Rowe, D.K.; Hicks, M.; Smith, J.P.; Williams, E. (2009). Lethal concentrations of suspended solids for common native fish species that are rare in New Zealand rivers with high suspended solids loads. *New Zealand Journal of Marine and Freshwater Research 43*: 1029–1038.

Ryan, P.A. (1991). Environmental effects of sediment on New Zealand streams: a review. *New Zealand Journal of Marine and Freshwater Research 25*: 207–221.

Suren, A.M.; Jowett, I.A. (2001). Effects of deposited sediment on invertebrate drift: an experimental study. *New Zealand Journal of Marine and Freshwater Research 35*: 725–737.

Thrush, S.F.; Hewitt, J.E.; Cummings, V.J.; Ellis, J.I.; Hatton, C.; Lohrer, A.M.; Norkko, A. (2004). Muddy waters: elevating sediment input to coastal and estuarine habitats. *Frontiers in Ecology and Evolution 2*: 299–306.

Townsend, C.R.; Uhlmann, S.S.; Matthaei, C.D. (2008). Individual and combined responses of stream ecosystems to multiple stressors. *Journal of Applied Ecology 45*: 1810–1819.

USEPA (2000). Draft technical framework to support the development of water quality criteria for sediment. USEPA, Office of Water. Washington DC, USA.

USEPA (2006). Framework for Developing Suspended and Bedded Sediment (SABS) Water Quality Criteria. EPA-822-R-06-001. United States Environmental Protection Agency, Office of Water, Office of Research and Development.

Waters, T.F. (1995). Sediments in streams: Sources, biological effects and control. *American Fisheries Society Monograph 7*. American Fisheries Society, Bethesda, Maryland, USA. 251 p.

Winterbourn, M.J.; Ryan, P.A. (1994). Mountain streams in Westland, New Zealand: benthic ecology and management issues. *Freshwater Biology 32*: 359–373.

Wolman, M.G. (1954). A method of sampling coarse river-bed material. *Transactions of the American Geophysical Union 35(6)*: 951–956.

Wood, P.J.; Armitage, P.D. (1997). Biological effects of fine sediment in the lotic environment. *Environmental Management 21(2)*: 203–217.

Wright-Stow, A. E.; Quinn, J. M. (2010). Effects of progressive harvesting on stream clarity, temperature, habitat and invertebrate communities in Whangapoua Forest: Eighteenth annual report incorporating the results from 1992–2010. *NIWA HAM2009-067*. NIWA, Hamilton, 93 p.