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Review

Understanding the influence of suspended solids on water quality and aquatic biota

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ABSTRACT

Over the last 50 years the effects of suspended solids (SS) on fish and aquatic life have been studied intensively throughout the world. It is now accepted that SS are an extremely important cause of water quality deterioration leading to aesthetic issues, higher costs of water treatment, a decline in the fisheries resource, and serious ecological degradation of aquatic environments. As such, government-led environmental bodies have set recommended water quality guidelines for concentrations of SS in freshwater systems. However, these reference values are often spurious or based on the concept of turbidity as a surrogate measure of the concentration of SS. The appropriateness of these recommended water quality values is evaluated given: (1) the large variability and uncertainty in data available from research describing the effects of SS on aquatic environments, (2) the diversity of environments that these values are expected to relate to, and (3) the range of conditions experienced within these environments. Furthermore, we suggest that reliance solely upon turbidity data as a surrogate for SS must be treated with caution, as turbidity readings respond to factors other than just concentrations of SS, as well as being influenced by the particle-size distribution and shape of SS particles. In addition, turbidity is a measure of only one of the many detrimental effects, reviewed in this paper, which high levels of SS can have in waterbodies. In order to improve the understanding of the effects of SS on aquatic organisms, this review suggests that: First, high-resolution turbidity monitoring should be supplemented with direct, measurements of SS (albeit at lower resolution due to resource issues). This would allow the turbidity record to be checked and calibrated against SS, effectively building a rating-relationship between SS and turbidity, which would in-turn provide a clearer picture of the exact magnitude of the SS problem. Second, SS should also be characterised in terms of their particle-size distribution and chemical composition. This would provide information to develop a more comprehensive understanding of the observed variable effects of a given concentration of SS in aquatic habitats. These two suggested improvements, combined with lower-resolution concurrent measures of aquatic ecological status, would improve our understanding of the effects of SS in aquatic

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environments and together with a more detailed classification of aquatic environments, would provide an environment-specific evidence base for the establishment of effective water quality guidelines for SS.

Contents

1. Introduction

The term suspended solids (SS) refers to the mass (mg) or concentration (mg L^{-1}) of inorganic and organic matter, which is held in the water column of a stream, river, lake or reservoir by turbulence. SS are typically comprised of fine particulate matter with a diameter of less than 62 μm (Waters, 1995), though for the majority of cohesive solids, research has demonstrated that transport frequently occurs in the form of larger aggregated flocs (Droppo, 2001; Droppo et al., 1997; Phillips and Walling, 1995). All streams carry some SS under natural conditions (Ryan, 1991). However, if concentrations are enhanced through, for example, anthropogenic perturbations, this can lead to alterations to the physical, chemical and biological properties of the waterbody. Physical alterations caused by SS include reduced penetration of light, temperature changes, and infilling of channels and reservoirs when solids are deposited. These physical alterations are associated with undesirable aesthetic effects (Lloyd et al., 1987), higher costs of water treatment (Ryan, 1991), reduced navigability of channels and decreased longevity of dams and reservoirs (Butcher et al., 1993; Verstraeten and Poesen, 2000). Chemical alterations caused by SS include the release of contaminants, such as heavy metals and pesticides (Dawson and Macklin, 1998; Kronvang et al., 2003; Miller, 1997), and nutrients such as phosphorus (Harrod and Theurer, 2002; Haygarth et al., 2006; Russell et al., 1998), into the water body from adsorption sites on the sediment. Furthermore, where the SS have a high organic content, their *in-situ* decomposition can deplete levels of dissolved oxygen in the water, producing a critical oxygen shortage which can lead to fish kills during low-flow conditions (Ryan, 1991). The biological effects of high levels of SS on different groups of organisms are discussed below and are summarised in Tables 1–3.

The effects of SS on various aquatic biota have been reviewed in the past (see Alabaster and Lloyd, 1982; Cordone and Kelley, 1961; Gammon, 1970; Newcombe and MacDonald, 1991; Owens et al., 2005; Petticord, 1980; Ryan, 1991; Wood and Armitage, 1997). In this review paper, we first provide an in-depth overview of the different mechanisms by which SS can affect different types of aquatic biota (Section 1). This section of the paper presents essential knowledge that underpins why we cannot rely solely on turbidity data to monitor and assess the effects of SS in aquatic environments. Section 2 identifies and reviews the key factors that determine the effect of SS on water quality and aquatic biota. This section of the paper demonstrates the complexities involved behind understanding the effects of a given concentration of SS on aquatic biota. Section 3 of the review paper discusses the various conventional methods applied in environmental monitoring of SS, highlighting, with reference to international water quality guidelines, several key issues and deficiencies with the existing measurement techniques for both turbidity and SS. This section of the paper also examines how these flaws may limit our understanding of the effects of SS in waterbodies and will inhibit attempts to mitigate SS-related

Table 1 – Summary of data on effects of various concentrations of, and durations of exposure to, suspended solids on periphyton and macrophytes

Organism	SS concentration ($mg\,L^{-1}$ or NTU)	Duration of exposure (h)	Effect on organism	Country of study	Reference
Periphyton and macrophytes					
Macrophytes and algae	$8\,mg\,L^{-1}$	–	3–13% reduction in primary productivity	United States	Lloyd et al. (1987)
Phytoplankton	$10\,mg\,L^{-1}$	1344	40% reduction in algal biomass	New Zealand	Quinn et al. (1992)
Macrophytes and algae	$40\,mg\,L^{-1}$	–	13–50% reduction in primary productivity	United States	Lloyd et al. (1987)
Aquatic moss	$100\,mg\,L^{-1}$	504	Severe abrasive damage to the leaves	Wales	Lewis (1973)
Periphyton	$100\,mg\,L^{-1}$	–	Stimulated growth and filament length (under low flow velocities)	United States	Birkett et al. (2007)
Macrophytes and algae	$200\,mg\,L^{-1}$	–	50% reduction in primary production	United States	Van Nieuwenhuyse and LaPerriere (1986)
Periphyton	$200\,mg\,L^{-1}$	–	Significant reduction in biomass and filament length	United States	Birkett et al. (2007)
Aquatic moss	$500\,mg\,L^{-1}$	168	Severe abrasive damage to the leaves	Wales	Lewis (1973)
Macrophytes and algae	$2100\,mg\,L^{-1}$	–	No primary production	United States	Van Nieuwenhuyse and LaPerriere (1986)
Periphyton	$0–6500\,mg\,L^{-1}$	–	Abrasive damage and reduced biomass	New Zealand	Francoeur and Biggs (2003)

Table 2 – Summary of data on effects of various concentrations of, and durations of exposure to, suspended solids on invertebrates

Organism	SS concentration ($mg\,L^{-1}$ or NTU)	Duration of exposure (h)	Effect on organism	Country of study	Reference
Invertebrates					
Benthic invertebrates	$8\,mg\,L^{-1}$	2.5	Increased rate of drift	Canada	Rosenberg and Wiens (1978)
Invertebrates	$8–177\,mg\,L^{-1}$	1344	Reduced invertebrate density by 26%	New Zealand	Quinn et al. (1992)
Benthic invertebrates	$62\,mg\,L^{-1}$	2400	77% reduction in population size	United States	Wagener and LaPerriere (1985)
Stream invertebrates	$130\,mg\,L^{-1}$	8760	40% reduction in species diversity	England	Nuttall and Bielby (1973)
Macro-invertebrates	$133\,mg\,L^{-1}$	1.5	Seven-fold increase in drifting invertebrates	Australia	Doeg and Milledge (1991)
Cladocera	$82–392\,mg\,L^{-1}$	72	Survival and reproduction harmed	United States	Robertson (1957a)
Invertebrates	Pulses	456	Reduced abundance and richness	Canada	Shaw and Richardson (2001)
Cladocera and Copepoda	$300–500\,mg\,L^{-1}$	72	Gills and gut clogged	Germany	Alabaster and Lloyd (1982)
Chironomids	$300\,mg\,L^{-1}$	2016	90% decrease in population size	United States	Gray and Ward (1982)
Benthic invertebrates	$743\,mg\,L^{-1}$	2400	85% reduction in population size	United States	Wagener and LaPerriere (1985)
Mayfly (leptophlebiid)	1000 NTU	336	No increased mortality	New Zealand	Suren et al. (2005)
Invertebrates	20,000 NTU	24	No increased mortality	New Zealand	Suren et al. (2005)
Invertebrates	$25,000\,mg\,L^{-1}$	8760	Reduction or elimination of populations	England	Nuttall and Bielby (1973)

Table 3 – Summary of data on effects of various concentrations of, and durations of exposure to, suspended solids on salmonids

Organism	SS concentration (mg L^{-1} or NTU)	Duration of exposure (hours)	Effect on organism	Country of study	Reference
Salmonids					
Atlantic salmon	20 mg L^{-1}	–	Increased foraging activity	Canada	Robertson et al. (2007)
Arctic grayling	25 mg L^{-1}	24	6% mortality of sac fry	Canada	Reynolds et al. (1988)
Rainbow trout	47 mg L^{-1}	1152	100% mortality of incubating eggs	Canada	Slaney et al. (1977)
Arctic grayling	65 mg L^{-1}	24	15% mortality of sac fry	Canada	Reynolds et al. (1988)
Atlantic salmon	60–180 mg L^{-1}	–	Avoidance behaviour. Reduced foraging activity	Canada	Robertson et al. (2007)
Arctic grayling	185 mg L^{-1}	72	41% mortality of sac fry	Canada	Reynolds et al. (1988)
Chinook salmon	488 mg L^{-1}	96	50% mortality of smolts	United States	Stober et al. (1981)
Sockeye & Coho salmon	800–47,000 mg L^{-1}	–	80% reduction in egg fertilisation success when SS > 9000 mg L^{-1}	Canada	Galbraith et al. (2006)
Coho salmon	2000–3000 mg L^{-1}	192	Reduced feeding efficiency and immunity	United States	Redding et al. (1987)
Rainbow trout	Pulses	456	Reduced growth	Canada	Shaw and Richardson (2001)
Brown trout	5838 mg L^{-1}	8,670	85% reduction in population size	England	Herbert et al. (1961)
Coho salmon	40,000 mg L^{-1}	96	Physical damage to gills, stress response	Canada	Lake and Hinch (1999)
Chinook salmon	207,000 mg L^{-1}	1	100% mortality of juveniles	United States	Newcomb and Flagg (1983)

water quality problems. Lastly, Section 4 of the paper suggests ways in which the techniques for measurement can be improved and how this should, in-turn, feed into more environment-specific water quality guidelines to alleviate the impacts of SS on water quality.

1.1. *The effects of SS on*

1.1.1. *Phytoplankton, periphyton and macrophytes*

Phytoplankton (algae suspended in the water column), periphyton (algae attached to stream substrates) and macrophytes (visible plants that are either rooted in the substrate in the case of *emergent* and *floating-leaved* macrophytes, floating beneath the surface in the case of *submersed* macrophytes, or floating on the surface, in the case of *free-floating* macrophytes), are important sources of food and producers of oxygen in the aquatic environment (Bronmark, 2005; Brown, 1987). SS can influence macrophytes and algae, primarily through affecting the amount of light penetrating through the water column. The reduction in light penetration through the water column will restrict the rate at which periphyton and emergent and submersed macrophytes can assimilate energy through photosynthesis, which will impact directly on primary consumers. However, it is worth noting that this mechanism is not so important for the planktonic species including surface phytoplankton, and floating-leaved or free-floating macrophytes. Furthermore, the importance of in-stream primary producers within food chains varies amongst different stream communities. For example, the small forested streams studied by Cowie (1983, 1985) in New Zealand, obtain a considerable proportion of their energy inputs from allochthonous sources such as decaying leaf matter. Under these circumstances the SS entering the waterbody are an important part of the ecosystem.

Periphyton abundance can also be influenced by SS through mechanisms other than reduced light penetration; (1) High levels of SS in transport by fast flow rates can act to scour these organisms away from streambed substrates as well as being abrasive and damaging to the photosynthetic structures of organisms (Alabaster and Lloyd, 1982; Steinman and McIntire, 1990). (2) SS can indirectly affect the abundance of phytoplankton, periphyton and macrophytes through acting as a vector of nutrients such as phosphorus (Heathwaite, 1994), and toxic compounds such as pesticides and herbicides from the land surface to the waterbody (Kronvang et al., 2003).

1.1.2. *Aquatic invertebrates*

Invertebrates can be divided into those that remain suspended in the water column (i.e. zooplankton), and those that inhabit the zone surrounding the streambed (i.e. benthic invertebrates). Benthic invertebrates include numerous species of insects, molluscs and crustaceans. SS can affect benthic invertebrates by subjecting them to abrasion and scouring as SS being carried in the flow move over the channel bed. This can damage exposed respiratory organs or make the organism more susceptible to predation through dislodgement (Langer, 1980). A number of studies have shown that increased SS are associated with an increase in invertebrate drift (down- or up-channel migration of organism). For example, Gammon (1970) showed that increases in

SS of 40–80 $mg\,L^{-1}$ above background levels caused an increase in invertebrate drift of 25–90%. Ryder (1989) showed that a sudden increase in the drift densities of stream insects occurred when suspended sediment was introduced into a natural stream. Ryder (1989) noted that in the normal course of events there would be a compensating drift from upstream, however, the introduction of fine material to the substrate and the associated turbidity may inhibit re-attachment to the stream bed and encourage fauna to continue drifting. For grazing invertebrates, Graham (1990) demonstrated that suspensions of clay-sized particulates can be trapped by epilithic periphyton and reduce its attractiveness for grazing. For filter-feeding invertebrates, high levels of SS can clog feeding structures, reducing feeding efficiency and therefore reducing growth rates, stressing and even killing these organisms (Hynes, 1970). For those invertebrates that graze periphyton for their energy and nutritional requirements, any changes in SS concentrations that adversely affect algal growth, biomass, or species composition can adversely affect populations of these types of invertebrates (Newcombe and MacDonald, 1991). Changes in the abundance of invertebrates have knock-on effects higher up in the food chain as discussed below.

1.1.3. *Salmonid fish*

As well as being important members of the aquatic food chain, salmonids, including trout, grayling, whitefish and salmon, are valuable game fish and an important economic and nutritional resource for humans (Cordone and Kelley, 1961; Ryan, 1991). As such, there has been a large amount of research into the effects of SS on salmonid fish (e.g. Alabaster and Lloyd, 1982; Cordone and Kelley, 1961; Greig et al., 2005; Harrod and Theurer, 2002; Lloyd, 1987; Newcombe and MacDonald, 1991; Redding et al., 1987). Salmonid fish can be affected by SS in several ways. The most intensively studied of these mechanisms involves the deposition and settling of SS in gravel-bed rivers. This has been recognised as a major cause for the reduced development and survival of salmonid eggs and larvae within salmonid redds (Harrod and Theurer, 2002). This is because the deposited material blocks the pores in the gravel-redd structure, preventing the sufficient exchange of dissolved oxygen and carbon dioxide between the respiring eggs/larvae and the flowing water (Greig et al., 2005; Walling et al., 2003). The presence of SS can also act directly on the free-living fish by clogging and being abrasive to their delicate gill structures (Cordone and Kelley, 1961; Ellis, 1944; Kemp, 1949), and/or stressing the fish and suppressing their immune system, leading to increased susceptibility to disease and osmotic dysfunction (Ellis, 1981; Redding and Schreck, 1983; Redding et al., 1987). Salmonid fish can also be affected by SS interfering with their natural movement and migration (Bisson and Bilby, 1982; Whitman et al., 1982).

It is worth noting, whilst salmonid fish are regarded as being particularly sensitive to SS, cyprinid fish (including carp, barbell, tench, rudd) on the other hand, are somewhat more tolerant to higher levels of SS (Alabaster, 1972; Cordone and Kelley, 1961). It is also worth noting that whilst fish are known to respond to SS fluxes, the fish themselves can also cause fluxes of SS through activities such as bioturbation whilst foraging and through excretion of waste products. For example, Matsuzaki et al. (2007) demonstrated that common carp (*Cyprinus carpio*, L.) could have a dramatic influence on sediment and nutrient dynamics resulting in a modification of the littoral community structure and triggering a shift from a clear water state dominated by submerged macrophytes, to a turbid water state dominated by phytoplankton. Even salmonids that prefer relatively clear waters carry out activities that may raise in-stream SS concentrations. For example, semelparous species such as the Pacific salmon invest all of their reproductive energy into one season which culminates in them digging redds in which to deposit and fertilise their eggs and then they die in the vicinity of their redd (Petticrew, 2006). The redd-digging procedure mobilises fine material previously stored in the gravel matrices and re-suspends this material in the water column (Chapman, 1998). More importantly, the decomposition of the post-reproductive salmon carcass releases organic material and nutrients into the waterbody (Ben-David et al., 1998; Bilby et al., 1996; Johnston et al., 2004; McConnachie and Petticrew, 2006).

2. Factors determining the effect of SS on aquatic biota

The effect of SS on aquatic biota is dependent on several key factors, these include: (1) the concentration of SS, (2) the duration of exposure to SS concentrations, (3) the chemical composition of SS and (4) the particle-size distribution of SS. These factors are discussed in the following section.

2.1. Concentration

Both the scientific literature and international water quality guidelines relating to SS are dominated by the implicit assumption that the concentration-response model applies to SS effects on aquatic biota (i.e. increase in SS = increase in effect on aquatic biota) (Newcombe and MacDonald, 1991). Indeed, numerous authors have reported that the magnitude of the effects of SS on aquatic organisms generally increases with SS concentrations. However, other factors such as the duration of exposure, particle-size distribution and chemical composition of the SS, and the presence of other contaminants on the solids, also appear to have an important control over the effect of SS on aquatic biota. These additional factors (discussed below) complicate the relationship between the magnitude of effect of SS and the concentration, making it difficult to predict the effect of SS on an organism merely by considering just the concentration. Furthermore, in contrast to the simple concentration-response model, there is also evidence that certain aquatic biota can be adversely affected by exceptionally low concentrations of SS. For example, systems such as the small forested streams studied by Cowie (1983, 1985) in New Zealand, rely upon allochthonous material (primarily decaying leaf matter) for their energy requirements. Removal or reduction of the source of this matter will therefore have deleterious impacts on the aquatic biota in the stream, particularly the various bacteria, fungi, larvae and invertebrates that mechanically break-down and consume this material (Winterbourn, 1987).

2.2. Duration of exposure

As mentioned previously, the duration of exposure to a concentration of SS is an important factor determining its effect on aquatic biota. A study by Suren et al. (2005) investigated the response of six common New Zealand invertebrates to short-term exposures of high SS concentrations. Suren et al. (2005) discovered that even with repeated exposures lasting less than 24 h, there was no pattern of increased mortality in 'sensitive' invertebrates. On this basis, Suren et al. (2005) suggested that the absence of these invertebrates from streams with high SS concentrations is therefore likely to reflect adverse long-term changes to in-stream habitat conditions such as filling of interstitial spaces and contamination of food sources. A study by Newcombe and MacDonald (1991) collated the results from more than 70 studies from the scientific literature and plotted the ranked severity of effect on aquatic organisms against either; (1) the suspended sediment concentration or (2) the suspended sediment intensity (defined as the concentration multiplied by duration of exposure), found that the ranked response of aquatic biota was poorly correlated with concentration of suspended sediment ($r^2 = 0.14$, $p > 0.05$), whereas the ranked response of aquatic biota was more strongly correlated to suspended sediment intensity ($r^2 = 0.64$, $p < 0.01$). This study suggests that suspended sediment effects on aquatic biota cannot be predicted simply by using concentration data, but also need to include measurements of the duration of exposure, and even with this information, further parameters are required for the effects on aquatic biota to be accurately predicted.

Related to the duration of exposure, and perhaps equally as important, is the timing of the SS delivery to streams and the timing of its transport in streams relative to the stage in the **life-cycle of aquatic biota. A given sediment concentration and duration of exposure will have different effects depending on seasonality. For example, fluxes of SS are likely to have a more significant impact on spawning salmon if they take place during the period of redd construction and egg incubation. Conversely, fluxes of SS during the winter or periods of senescence are likely to have a lower impact on aquatic biota, at least in the short term.

2.3. Geochemical composition

The geochemical composition of the suspended load in a waterbody is an important factor in determining its effect on aquatic organisms. The geochemical composition will influence both the physical characteristics of the solids (including the shape, angularity and particle-size of the SS) and the chemical characteristics of the solids, including the likelihood of any chemical alterations in the receiving waters (e.g. pH, salinity, dissolved oxygen, phosphorus concentration, toxicity). Despite the large number of studies on the effects of SS on aquatic biota (see Tables 1–3), relatively few of these studies have considered the geochemical composition or characteristics of the suspended load. A study by Stephan (1953) investigated the effects of SS, of various composition, on the invertebrates; Cladocera and Copepoda noting that the harmful effect of SS was primarily through clogging of their filter-feeding apparatus and digestive organs, and this damage was worse in suspensions of clay, followed by earth, and then sand. A study by Robertson (1957b) investigated the effects of SS, of various geochemical composition, on the survival and reproduction of *Daphnia magna* and found that the concentrations considered to be harmful to populations of *Daphnia*, varied from 392 mg L^{-1} for kaolinite suspensions, to 102 mg L^{-1} for montmorillinite suspensions, to 82 mg L^{-1} for charcoal suspensions (i.e. the sensitivity of the organism to a given concentration of SS depended on the geochemical composition of the SS). Other studies have looked at the effect of SS in effluent from localised sources, such as coal mining (e.g. Lewis, 1973), gold mining (e.g. Van Nieuwenhuyse and LaPerriere, 1986; Wagener and LaPerriere, 1985) and china-clay works (e.g. Herbert et al., 1961; Nuttall and Bielby, 1973), where the composite solids have distinct properties. Nevertheless, despite the research that has been done on this subject area, the chemical composition of the SS remains a neglected factor in many studies on the effects of SS and is even neglected in the development of SS water quality guidelines.

2.4. Particle-size distribution

The particle-size distribution of the suspended load determines: (1) the duration of time that the particles will remain in suspension for, (2) the depth-distribution of SS within the water column and (3) the sorption-capacity of the SS. First, according to the semi-empirical equation known as Stoke's law, smaller particles (or aggregates of particles) will generally remain in suspension in the water column for longer periods than larger particles (for a given turbulence, when particle density and shape are the same) (Schindl et al., 2005). Therefore, because of the importance of the duration of exposure, SS loads with a fine particle-size distribution present a longer-term threat to aquatic organisms. Second, under low or zero-flow conditions, the finer particles tend to occupy the surface zone of the waterbody, whilst the coarser particles tend to occupy the deeper zone of the waterbody (Schindl et al., 2005). This determines which organisms will be affected (i.e. plankton or benthic organisms). The coarser particles (i.e. sand, silt and clay) are most likely to be deposited on the waterbody bed, potentially influencing salmonid redds (Greig et al., 2005), whilst the finer material may pose more of a threat to the organs of fish and invertebrates involved in respiration and feeding. Third, the particle-size distribution influences the sorption capacity, with finer particles tending to have a higher sorption capacity due to the presence of colloidal properties (large surface area to volume ratio, and surface charges) (Brady and Weil, 1999). The sorption capacity of the SS will determine how effective the particles are at acting as a vector of contaminants from the land surface, and the potential for SS to modify the chemistry of the waterbody (Brady and Weil, 1999; Schindl et al., 2005; Stone and Droppo, 1994).

Traditionally, sediment researchers working in a range of environments have measured the chemically dispersed (i.e. disaggregated) mineral fraction of sediment in order to characterise the particle-size distribution (Droppo, 2001; McConnachie and Petticrew, 2006). However, it is now

Table 4 – Summary of water quality guidelines for levels of suspended solids in the surface waters of Canada, United States, European Union member states, Australia and New Zealand

Organisation	Policy name	Countries/ states involved	Narrative statement		
Canadian Council of Ministers of the Environment (CCME)	Canadian Environmental Quality Guidelines (CEQG) for Protection of Freshwater Aquatic Life CCME (2007)	Canada	**Low Flow**: Concentrations should not increase by more than 25 mg L^{-1} from background levels for any short-term exposure (e.g. 24-h period). Concentrations should not increase by more than 5 mg L^{-1} from background levels for any long-term exposure (e.g. inputs lasting between 24 h and 30 d). **High Flow**: Concentrations should not increase by more than 25 mg L^{-1} from background levels at any time when background levels are between 25 and 250 mg L^{-1}. Concentrations should not increase by more than 10% of background levels when background is $>$250 mg L^{-1}.		
United States Environment Protection Agency	National Recommended Water Quality Criteria US EPA (2007)	United States	Settleable and suspended solids should not reduce the depth of the compensation point for photosynthetic activity by more than 10% from the seasonally established norm for aquatic life.		
European Union Freshwater Fisheries Directive	Freshwater Fisheries Directive (78/659/EEC) and (2004/44/EC)	European Union	Apart from in exceptional circumstances, such as storms or droughts, concentrations of suspended solids should not exceed 25 mg L^{-1} in waters suitable for both salmonid and cyprinid fish populations. These values are guideline standards that should be achieved where possible. Suspended solids are not included in the imperative standards which must be achieved if the stretch is to pass the directive.		
Australian and New Zealand Environment and Conservation Council (ANZECC)+Agriculture and Resource Management Council of Australia and New Zealand	Australian and New Zealand Guidelines for Fresh and Marine Water Quality ANZECC (2000)		Upland Rivers ($>$150 m $<$1500 m altitude)	Lowland Rivers ($<$150 m altitude)	Freshwater lakes and Reservoirs
		South East Australia	2–25 NTU[a]	6–50 NTU[a]	1–20 NTU[a]
		Tropical Australia	2–15 NTU[a]	2–15 NTU[a]	2–200 NTU[a]
		South West Australia	10–20 NTU[a]	10–20 NTU[a]	10–100 NTU[a]
		South Central Australia	1–50 NTU[a]	1–50 NTU[a]	1–100 NTU[a]
		New Zealand	4.1 NTU[a]	5.6 NTU[a]	N/A[a]

[a] NTU is Nephelometric Turbidity Units.

accepted that in the aquatic environment cohesive sediments (i.e. silts and clays <62 μm) tend to be transported in the form of composite or aggregate particles, termed flocs, that are bound together by living (e.g. bacteria and algae) and non-living (e.g. organic detritus, extra-cellular polymeric substances) substances (Droppo, 2001; McConnachie and Petticrew, 2006; Petticrew, 1996; Phillips and Walling, 1995). This is an important consideration as it has significant implications on the particle structure (size, shape, density) and behaviour (settling velocity) and ultimately on the influence that the solids have on aquatic organisms and water quality and our understanding of this process.

3. SS and international water quality guidelines

Clearly, SS are an important pollutant in surface waters. In recognition of this, government-led environmental bodies have responded by establishing water quality guidelines and standards, which state recommended concentrations of SS in surface waters (see Table 4 for examples). This review questions the appropriateness of these recommended water quality values given; (1) the large variability and uncertainty in data available from research describing the effects of SS on aquatic environments, (2) the diversity of environments that these values are expected to relate to (e.g. gravel bed channels to channels in eroding alluvial plains), (3) the range of conditions experienced within these environments (e.g. storm flows to droughts) and (4) the use of turbidity as a surrogate measure of SS has several limitations, principally that turbidity responds to factors other than just the concentration of SS, and turbidity is only one of the many effects of SS. These limitations are discussed in more detail in the following paragraphs.

3.1. *Variability and uncertainty in data*

As can be seen in Tables 1–3, there is a large amount of *variability* (i.e. differences in response to SS related to natural factors, such as organism type, species, stage in life-cycle, environmental characteristics) and *uncertainty* (i.e. differences in response related to the ability to measure accurately and quantify factors such as SS concentrations and organism response) in the data from research describing the effects of SS on aquatic organisms. This means that developing a simple recommended water quality guideline for SS is complicated and challenging. One reason for the variability and uncertainty in available data describing the effects of various concentrations of SS on aquatic biota is that there have been a variety of techniques (with variable precision of measurement) used to study the effects of SS (Newcombe and MacDonald, 1991). In terms of measuring the concentration of SS in water, uncertainties arise from the sampling (Phillips and Walling, 1995; Schindl et al., 2005) and/or monitoring method deployed (Henley et al., 2000) as well as the analytical technique utilised (Clark and Siu, 2008; Schindl et al., 2005). In terms of measuring the response of organisms to SS uncertainties arise from the method of biological assessment and ecological survey, as well as the choice of experimental design. Carefully controlled experiments, subjecting organisms to different conditions, are rare (Cordone and Kelley, 1961). A second reason is that, unfortunately, not all of the important factors that influence the reactions of biota have always been measured in the field (Cordone and Kelley, 1961). A third reason for the variability and uncertainty in response of aquatic biota to concentrations of SS is that although the effect of SS is related to the concentration of SS in the water column, it also depends on the type/species of organism, the stage that the organism is at within its life cycle, the duration and seasonal timing of exposure to the SS, as well as the chemical composition and particle-size distribution of the SS mentioned above. The literature and indeed the guidelines used by the US EPA and EU FFD (for example) are dominated by the implicit assumption that the concentration-response model applies to SS effects on aquatic biota (Newcombe and MacDonald, 1991). However, the study by Newcombe and MacDonald (1991) (mentioned in Section 2.2), suggested that suspended sediment effects on aquatic biota cannot be predicted simply by using concentration data, therefore, it is questionable as to whether the existing guidelines for SS, based solely on concentrations, are appropriate for their cause.

3.2. *Diversity of environments*

The water quality guidelines (Table 4) are used to cover a wide range of environments. The EU FFD divides waters into two categories; those suitable for (1) salmonid fish (salmon and trout)—these are generally fast flowing stretches of river that have a high oxygen content and low levels of nutrients, and those suitable for (2) cyprinid fish (coarse fish—carp, tench, barbel, rudd, roach)—these are slower flowing waters that often flow through lowlands (Environment Agency, 2007). Although there are two categories, this is not enough given the diversity of aquatic environments within the countries subscribed to this legislation. As mentioned previously, all streams carry some SS under natural conditions (Ryan, 1991), however, the large variability in the natural background levels of SS concentrations throughout different aquatic environments means that setting just two SS guidelines for two categories of waters is not appropriate. For example, the natural background levels for SS in a gravel-bed river tend to be much lower than that in a river which runs through a meandering alluvial floodplain with a sand and silt bedded channel (Church, 2002; Dade and Friend, 1998). A study by Bond (2004) calculated that rates of fine sediment transport were up to 100 times greater in granite- and sandstone-bed streams than for cobble- and gravel-bed streams. This difference is not necessarily due to human impacts and does not necessarily need to be mitigated against; different environments have different ecological roles, supporting different organisms. The Canadian Environmental Quality Guidelines (CEQG) and US EPA water quality criteria attempt to address this issue of producing environment-specific guidelines by referring to a change in SS from background values (CEQG), or from seasonally established norms (US EPA) using simple statistics such as mean concentration/turbidity. The issue with this type of guideline is that there is potential for anthropogenically enhanced concentrations of SS to be included in the background levels or seasonally established norms. Where this is the case, the water quality monitoring would therefore suggest that no water quality remediation is required, even though the SS input from human activities may be seriously affecting the aquatic environment. Mean or 'background' SS concentrations and/or turbidity does not necessarily reflect the 'natural' levels in the stream, or the optimum levels for aquatic biota, the mean values may still be elevated above 'natural' levels as a result of human activity. Whilst it is difficult to determine the 'natural' characteristics of any given waterbody (Montgomery, 2008), it is more achievable and sensible to establish an optimum guideline SS value for a given waterbody based on ecological monitoring and scientific evidence as opposed to adopting the potentially anthropogenically altered state as being the optimum. From Table 4, it is clear that the ANZECC guidelines (ANZECC, 2000) are the most flexible for different environments, with different guidelines for different regions (five different regions), altitudes (0–150 m, >150–<1500 m) and types of waters (lakes/reservoirs, streams/rivers).

3.3. *Range of conditions*

The water quality guidelines are used to cover a wide range of flow conditions, from droughts to floods. The amount of SS transported in the water column is highly dynamic and will generally increase with transport capacity (i.e. discharge) until or unless the SS become supply-limited (Rossi et al., 2006). Therefore, we may expect higher background levels of SS during storm events and flood conditions when compared to base-flow conditions. The Canadian environmental quality guidelines attempt to address this by dividing flow conditions into high and low flow and providing different guidelines for each flow class. The US EPA water quality criteria attempt to address this by considering seasonal differences in flow conditions. The EU FFD attempts to address this by suggesting that the guidelines should not apply to exceptional circumstances such as storms and droughts. These divisions of flow types are of little use, and there is a strong need to include the more extreme conditions within our guidelines and monitoring. Instead of overlooking extreme high and low flows, information on the SS response to flow conditions (i.e. sediment to discharge rating curves) should be collected and used to characterise the variability in the relationship between SS and discharge. This rating curve approach can be used in the future to investigate how land management is influencing the SS response to flow, for example, does the land management method increase or decrease the SS response to flow?

3.4. *Turbidity as a surrogate measure*

Turbidity as a surrogate measure of SS (as used in the ANZECC and US EPA water quality criteria) has several limitations. Conventionally, SS were quantified directly through collection of a sample of water followed by filtration of this sample through a dried and pre-weighed 0.7 μm pore-size glass fibre filter (Anon, 1980; Gray et al., 2000). SS are operationally defined as the mass retained on the filter per unit volume of water ($mg\,L^{-1}$). This technique, however, can be time-consuming and expensive, particularly if a large number of samples are to be collected and analysed. Consequently, turbidity is often measured and used as a surrogate measure of SS. Turbidity is the measure of the light scattering properties of water. It is typically measured using *in-situ* equipment which record the attenuation (i.e. attenuance turbidimeters—measure the loss in intensity of a narrow parallel beam or dual beams) or scattering (i.e. nephelometric turbidimeters—measure light scattered at an angle to the beam), of a beam of radiation (Lewis, 1996). Nephelometric turbidimeters have been most widely used, recording turbidity data in nephelometric turbidity units (NTU) (Lewis, 1996).

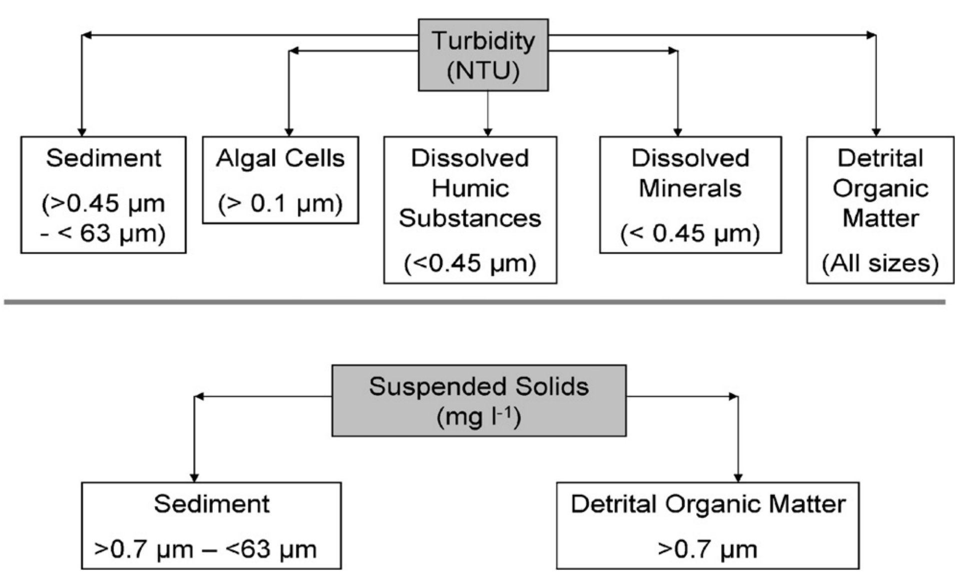

Fig. 1 – A schematic diagram illustrating the various components that are incorporated when measuring turbidity (top) and suspended solids via the conventional method (bottom).

Turbidity has the advantage that it can be measured at high-resolution time-steps, however, there are serious limitations when using turbidity as a surrogate measure of SS. First, turbidity is a measure of only one of the many effects of SS (discussed in this paper). Second, turbidity responds to factors other than just SS concentrations (see Fig. 1). Turbidity readings are influenced by the particle size and shape of SS, the presence of phytoplankton, the presence of dissolved humic substances and the presence of dissolved mineral substances. Consequently, a high turbidity reading can be recorded without necessarily involving a high SS concentration. Therefore, if relying solely on turbidimeter data, it is not straightforward to know exactly what is causing the turbidity and therefore it is difficult to advise those involved in land management as to what the exact problem is when there is one, and how to mitigate against it. Whilst time-series of turbidity may do well at describing the reduction in light penetration and aesthetic issues surrounding SS, it is likely that their use will lead to underestimation of the broader effects of SS in the aquatic environments.

4. Developing more advanced water quality guidelines for SS

The purpose of establishing water quality guidelines is to protect and improve water quality and the health of aquatic ecosystems. This is done through a system of environmental monitoring coupled with the implementation of mitigation measures where waters do not achieve the desired status. In order for the purpose of SS water quality guidelines to be successful this paper suggests that several improvements need to be made to the existing structure of the guidelines.

First, recommendations should be environment-sensitive. At present the guidelines, particularly those for the EU, are not flexible enough for the diversity of environments that they are supposed to be applicable to. The EU Water Framework Directive (WFD) is a piece of legislation which will eventually incorporate the EU FFD, it came into force in December 2000, was translated into UK law by the end of 2003, and will be fully implemented by 2015. This legislation is a considerable advancement in the protection of EU water resources and aquatic habitats. The WFD is more complex than the FFD in terms of the classification of waters because it subdivides waters in terms of their typology (factors include; latitude, longitude, altitude, depth, geology and size), as well as their natural hydro-morphological and physico-chemical conditions (factors such as (1) the power of the water to erode and transport sedments and (2) the size and availability of sediments upon which water can act, from resistant rocks to more easily displaced sands and silts) (UKTAG, 2007, 2008). The EU WFD will also recognise that some waterbodies have been heavily modified by human activity (heavily modified waterbodies—HMWB) and may never achieve the ecological status of a 'natural' stream. Accordingly, the WFD will set specific and realistic guidelines for ecological status in these water bodies.

However, for the immediate future, the EU WFD stands to inherit the simple numeric guidelines for SS from the EU FFD, without consideration of these novel environmental classifications (Collins and Anthony, 2008). This would be a missed opportunity as monitoring of SS in such a range of environments would help to inform water quality policy-makers as to suitable environment-specific guideline values. The ANZECC guidelines already have a more advanced environmental classification system, providing water quality managers with environment-specific quantitative water quality guidelines as well as narrative statements to aid decisions regarding the state of a waterbody. Programmes such as SedNet (2006), the EU WFD Group on Analysis and Monitoring of Priority Substances (http://scientificjournals.com/sj/jss/pdf/aId/6674), the EU Joint Research Council (JRC)—Institute for Environment and Sustainability (http://ies.jrc.cec.eu.int/515.html), and the UK Technical Advisory Group (http://www.wfduk.org/) are all involved in providing recommendations and establishing similarly as advanced sediment initiatives for the EU WFD to adopt in the future.

Second, monitoring within these different environments needs to be on a high-resolution basis (at least hourly, though ideally sub-hourly), in order to ensure that fluctuations in flow and SS are captured. This high-resolution monitoring would enable the calculation of daily exposures and duration of exposure to SS; a factor recognised as important in determining the effect of a given SS concentration on aquatic biota (Newcombe and MacDonald, 1991). Ideally, monitoring would be carried out via the conventional measurement method of filtration and weighing of a sub-sample of flow. Where this cannot be done, however, due to resource issues, turbidity probes could be used in combination with lower-resolution sample analysis for SS, so that the record can be checked and calibrated for SS, effectively building a rating relationship between suspended sediment and turbidity as has been demonstrated by workers such as Gippel (1995), Grayson et al. (1996) and Wass and Leeks (1999). Alternative automated monitoring technologies including (1) acoustic technologies such as the use of acoustic Doppler current profilers (ADCP) (see: Holdaway et al., 1999) and multi-beam echosounding (MBES) (see Simmons et al., 2007) and (2) light technologies (light-scattering, reflectance and attenuation equipment) such as the use of fiber optic in-stream transmissometers (FIT) (see Campbell et al., 2005) should also be considered and developed further.

Third, SS should be characterised in terms of their particle-size distribution and geochemical composition. This would provide more information to enable us to understand the observed variable effects of a given concentration of SS in aquatic habitats. Williams et al. (2007) carried out the first application of a portable laser-diffraction particle-sizer for use on SS in a fluvial environment over the temporal scale of a storm event. This demonstrated that it is possible to make automated, *in situ*, high temporal resolution measurements of the effective particle-size characteristics of SS, negating the need for time-consuming sampling followed by conventional laboratory analysis and the associated uncertainties involved in this and the subsequent sample storage and analysis (Philips and Walling, 1995)**. The guidelines and monitoring used in the EU Air Quality Directive (1999/30/EC) for airborne particulate matter (see http://europa.eu.int/eur-lex/lex/LexUriServ/LexUriServ.do?uri=CELEX:31999L0030:EN:HTML) are more advanced than the EU water quality equivalent (perhaps due to the direct effects that air quality has upon human health), in that high-resolution (1 h time-step) quantitative monitoring of airborne particulates is carried out for the calculation of daily exposures, in addition to analysis of the particle-size distribution of airborne particulates (particulate matter $<10\,\mu m$ and particulate matter $<2.5\,\mu m$), and analysis of the chemical composition of the particulate matter. Here we argue that a true understanding of the effects of SS on water quality will not be possible until this characterisation research has been conducted and linked to the ecological status of our surface waters.

Finally, ecological status should be measured concurrently with turbidity/SS. Concurrent measurement of SS and ecological status (the latter at low resolution) could improve our understanding of the effects of SS in aquatic environments and together with a more detailed classification of environments, would provide an environment-sensitive evidence base for SS water quality guidelines.

The science behind SS water quality guidelines and monitoring appears to be lagging behind the air quality equivalent. There is a need to develop the science behind SS monitoring and characterisation. This should be followed by harmonisation of monitoring strategies, sampling and measuring methods to arrive at comparable measurements throughout different environments, and to provide long-term records and data sets which can be used to investigate trends in SS levels in aquatic environments and their effects on aquatic biota. This in-turn, can feed into more environment-sensitive, evidence-based, water quality guidelines for SS.

5. Conclusions

- The delivery of excessive levels of SS into waterbodies can have significant deleterious impacts on the physical, chemical and biological properties of the waterbody.
- The magnitude of the effect is dependent upon the concentration, duration of exposure, chemical composition, and particle-size distribution of the solids, but also varies between organisms and between environments.
- As a consequence of this complexity, establishing water quality guidelines for SS is a challenging task.
- At present, water quality guidelines for SS are too simplistic for their cause; using turbidity as a surrogate measure of SS, applying simple numeric values to a wide range of environments, and failing to consider duration of exposure and variability of flow conditions. The science behind monitoring of SS is lagging behind the air quality counterpart.
- In the future, the monitoring of SS needs to be carried out in a high-resolution manner, in combination with analysis of the chemical composition and particle-size distribution of the solids.
- This monitoring needs to be carried out in a range of environments concurrently with ecological monitoring.
- Combined, this data will provide information to produce environment-sensitive, evidence-based water quality guidelines and a more holistic understanding of the effects of SS in surface waters.

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