The Biological Effects of Suspended and Bedded Sediment (SABS) in Aquatic Systems: A Review

Internal Report

August 20, 2003

by

Walter Berry, Norman Rubinstein, and Brian Melzian
United States Environmental Protection Agency
Office of Research and Development
National Health and Environmental Effects Laboratory
Atlantic Ecology Division
27 Tarzwell Drive
Narragansett, RI 02882

Brian Hill
United States Environmental Protection Agency
Office of Research and Development
National Health and Environmental Effects Laboratory
Midcontinent Ecology Division
6201 Congdon Boulevard
Duluth, MN 55804

Executive Summary

A review was conducted of the available literature on the biological effects of suspended and bedded sediment (SABS), and the current state standards for SABS to assess the feasability of developing national scientifically-defensible SABS criteria using the traditional “toxicological” dose-response approach. The review has the following take home messages:

1) Some useful models for the biological effects of SABS exist and others are under development. As the water clarity criteria for the protection of SAV in the Chesapeake show, the traditional toxicological dose-response approach can be used if a specific species from a particular habitat is to be protected and the required dose-response data are available. Generalizations are difficult because biological response to both increased suspended sediment and increased bedded sediment varies with species and sediment characteristics.

2) After additional research it may be possible to develop national scientifically-defensible SABS criteria using the traditional “toxicological” dose-response approach. These criteria will presumably have to incorporate some habitat-specificity in order to be widely applicable.

3) Some habitats that have not been well studied (in terms of their sensitivity to SABS) deserve more study, especially those habitats with moderate and variable amounts of SABS.

4) Many states have set standards for SABS, but there is little consistency among them.

Table of Contents

Introduction

Imbalance in loading of suspended and bedded sediment (SABS) to aquatic systems is now considered one of the greatest causes of water quality impairment in the Nation (U.S.EPA, 2003a). Turbidity, suspended solids, sediment, and siltation have been consistently listed in 305(b) Water Quality Reports in rivers and streams, lakes, reservoirs, ponds, wetlands, and ocean shoreline waters (Table 1). In 1998, approximately 40% of assessed river miles in the U.S. had problems arising from sediment stress (U.S.EPA, 2000). The effects of sediments on receiving water ecosystems are complex and multi-dimensional, and further compounded by the fact that sediment flux is a natural and vital process for aquatic systems. We use the term sediment imbalance here to connote significant changes in normal sediment loading to aquatic systems, changes that typically result in increases in sedimentation but can also result in reductions in sedimentation when compared to natural patterns. Sediment stress results from a change in sediment load originating from within the watershed, ultimately compromising the ecological integrity of the aquatic environment (Nietch and Borst, 2001).

Although the lack of sediment supply due to dam construction, bank modification, water diversion, and sea-level rise is a serious problem in some areas, leading to loss of wetlands (e.g., Boesch et al., 1994), lack of sediment was considered more of a physical than biological concern for the purpose of this review. Since very few studies have found organisms with a need or even a preference for increased suspended sediment or sedimentation in the field or laboratory (Cyrus and Blaber, 1987a,b) this paper will concentrate on the deleterious effects of increases in sediment supply to watersheds.

The impacts of suspended and bedded sediment in surface and coastal waters have been reviewed by a number of authors. Recent reviews of sedimentation in aquatic systems include Waters (1995), Naiman and Baily (1998), Reid and Dunne (1996), Wilber and Clarke (2001), and Nietch and Borst (2001). The vast majority of information presented in these reviews pertains to sources and exposure regimes of sediments as a function of geomorphology, erosional processes, catchment basin properties, and other geophysical factors (e.g., Leopold et al., 1964).

In this review we focus on direct and indirect biological effects of sediment (suspended and bedded) imbalance in aquatic systems. The literature on suspended sediment is larger and better summarized than that for bedded sediment, and that is reflected in the greater emphasis on the effects of suspended sediment in this review. We further restrict this review to “clean” (uncontaminated) sediments and do not address biological effects caused by chemical toxicants associated with sediments. In addition to toxicants associated directly with sediments and the sediments themselves, animals in a natural environment are exposed to mixtures of chemical and physical stressors which can combine to cause adverse effects that may not be observed when a stressor (like SABS) is considered individually (Herbranson et al., 2003a). This problem will be considered in the modeling section.

We focus on those studies that describe quantitative dose-response relationships of aquatic organisms exposed to suspended and bedded sediments. One of the goals of this review is to provide a simple, practical compilation of referenced sediment-effects (dose-response) information useful for development of sediment Total Maximum Daily Loads (TMDLs) for receiving waters (see U.S.EPA, 1999 for a description of the TMDL process) and suspended and bedded sediment criteria.

Many of the reviews of the clean sediment literature have been listed above. Most of them have been limited in scope to a particular habitat (e.g., Waters, 1995, dealt with streams) or taxonomic group (e.g., Newcombe and Jensen, 1996, dealt with fishes). This paper is not intended to be an exhaustive review of the primary literature, rather it summarizes the existing literature and models of the biological effects of suspended and bedded sediments on a wide range of organisms from various habitats. We also provide some useful tools for resource managers by providing summaries of existing models for the biological effects of suspended and bedded sediments, providing a table of existing data on the biological effects of suspended sediment, and a table of the current criteria and standards for bedded and suspended sediment.

Approaches to setting numerical targets

Within the regulatory community, the terms “guidelines”, “criteria”, and “standards” all have specific regulatory meaning. Guidelines do not necessarily have any regulatory authority. Criteria are set by U.S.EPA as recommendations, which have the force of law when adopted by states and tribes as standards. The term “criteria” will be used throughout this review to take the place of all three of these terms.

The U.S.EPA’s Office of Water is presently considering how to develop criteria for SABS. The potential approaches for criteria development that U.S.EPA’s Office of Water is considering investigating in the Strategy for Developing Water Quality Criteria for SABS include the following:

1. State-by-State Reference Condition Criteria Derivation Approach
2. Conditional Probability Approach to Establishing Thresholds
3. Toxicological Dose-Response Approach
4. Relative Bed Stability and Sedimentation Approach
5. Rosgen Geomorphological Approach
6. Water Body Use Functional Approach
7. Combinations of above approaches

The purpose of this document is to review the data available to support the development of criteria for SABS using the toxicological dose-response approach. This should help provide a basis for deciding if it will be possible to develop SABS criteria using this approach. Furthermore, an understanding of the mechanisms of action of SABS on biota complements the inferential data generated via the field data-associative approaches (Suter et al., 2002). This understanding has been referred to as "the missing link" between excess sediment and the designated use of a water body (Kuhnle and Simon, 2000; Kuhnle et al., 2001).

We propose the following steps, which correspond with the initial steps for TMDL development, for setting SABS criteria using the toxicological approach:

1) Develop a conceptual model outlining the ecological processes affected by SABS for a particular class of water body,

2) Choose the ecological processes, species or groups of species, and designated uses deemed desirable for protection, and

3) Develop numerical targets for protecting the ecological processes, species or groups of species, and designated uses deemed desirable for protection based on the correlations between SABS and biotic response.

There is a need for habitat classification in order for a program attempting to develop criteria or TMDLs for SABS to be successful because different sites have different processes involving SABS, and different tolerance levels depending on the habitat. The amount of suspended sediment tolerated in a mountain stream may be much different from that tolerated in the Mississippi River. Even within habitats there may be great variation in the effect of SABS. This need is discussed in detail in the Aquatic Stressors Framework document (U.S.EPA, 2002a), and was a continuing theme in the peer review comments on the Framework. A very general example of a conceptual model of the biological effects of SABS is presented later in this document. It may be that with a better understanding of the effects of SABS in the environment, the need for site-specificity in conceptual models for SABS will not be as great as previously thought.

The second step in the process, deciding which species or designated uses to protect, is largely a management decision, and outside the purview of this document. The simplest approach is to "protect everything", that is, to set the criteria or TMDL at a level protective of the most sensitive aquatic organisms. This is roughly equivalent to making sure that SABS do not exceed the background levels used in the reference approach. Another approach is to "protect most everything", as is done for the water quality criteria, which attempt to be protective of 95% of the genera tested (Stephan et al., 1985). An alternate approach is to choose the most sensitive of the biota which are deemed important. This requires a value judgment. The role of science is to determine which parts of the ecosystem are the most sensitive, and to develop the information that can be used to establish target levels with the desired level of protection and uncertainty associated with them.

An investigation into the science required for the third step in the process, developing effects-based target levels for protection from SABS, forms the bulk of this review. SABS have many impacts in aquatic ecosystems, and effects on biota vary considerably among habitats. However, there are dose-response models for some species in some habitats, and criteria have been developed for their protection (e.g., British Columbia Guidelines in Caux et al., 1997, Chesapeake Bay Water Clarity Guidelines in U.S.EPA, 2003b). Whether the science behind these criteria is adequate is the subject of debate. Some investigators (e.g., Newcombe and Jensen, 1996) maintain that there are empirical models presently available that can be used to predict the effects of SABS, and thus to develop effects-based guidelines. Others maintain that the models and data that are now available are not adequate for effects-based criteria development, with the possible exception of salmonid protection in streams (e.g., Wilber and Clarke, 2001) and SAV protection in the Chesapeake (U.S.EPA, 2003b).

Conceptual model of the biological effects of SABS

Organizing a broad review of the data on the effects of SABS is difficult because SABS can have effects in a wide range of habitats, including streams, rivers, lakes, estuaries, wetlands, coral reefs, and beaches. Some of these habitats are more well-studied than others. There are many studies involving streams, for example, while there are very few studies of the biological effects of SABS on beaches. Within each of these habitats live many types of animals and plants, many of which are vulnerable to effects from SABS. In this section we will present a general conceptual model. Later sections of this review will be organized by taxonomic groups: plants, benthic invertebrates, fish, and aquatic-dependent wildlife.

A conceptual model of the movement and effects of SABS is shown in Figure 1. In this model, sediments enter waterways through a wide variety of transport mechanisms, including surface water transport, bank sloughing, and atmospheric deposition. Once in the system, resuspension and deposition can "recycle" sediments, reintroducing them into the water column where they can exert water column effects, and then redepositing them where they can have further effects on the benthos. Anthropogenic activities which enhance erosional processes (e.g., forestry, mining, urban development, agriculture, dam construction) are among the most pervasive causes of sediment imbalance in aquatic systems (Waters, 1995, Nietch and Borst, 2001). Dredging activities can also lead to increased suspended sediment and deposition, both at the dredging site and the disposal site (Wilber and Clarke, 2001).

The biological effects of SABS on estuarine environments were reviewed by Wilber and Clarke (2001) and will only briefly be summarized here. Further discussion will be found later in the sections dealing with effects on the various taxonomic groups. Excessive sediments in aquatic systems contribute to increased turbidity leading to altered light regimes which can directly impact primary productivity, species distribution, behavior, feeding, reproduction, and survival of aquatic biota. Reduced light can reduce production of phytoplankton, submerged aquatic vegetation, and the zooxanthellae in corals. Reduced light and increased turbidity can also affect the feeding ability and movements of fish, especially larval fish. Larger fish may be able to reduce some of these effects by avoiding low visibility water. Wildlife may also have trouble hunting in turbid water, but like some fish they may be able to avoid some short -term turbidity events by relocating. Humans are also affected by the lack of water clarity - turbid water is generally not as aesthetically pleasing as clean for swimming or other recreational activities, or for drinking water.

Other direct effects of increased SABS include physical abrasion, and clogging of filtration and respiratory organs. The concentrations of suspended sediment required to cause these sorts of effects are generally very high, but may occur in certain situations such as near dredges (Wilber and Clarke, 2001). In extreme cases, excess SABS can cause burial and smothering of infaunal or epibenthic organisms. Most estuarine benthic organisms are adapted to living in an environment subject to periodic resuspension of sediment and can dig out from under a small amount of sediment (Maurer, 1986). Demersal eggs may be particularly vulnerable, however, as only a few millimeters of deposited sediment may prevent them from hatching (D. Nelson, personal communication).

Some of the most important indirect effects of SABS in estuarine and marine habitats relate to loss of primary and secondary production. Reductions in primary production effects primary consumers, which in turn effects secondary consumers, and on up the food chain. Eventually these effects reach even the top predators, such as eagles and humans.

The effects of SABS in streams were reviewed by Waters (1995). SABS have two major avenues of action in streams and rivers: 1) direct effects on biota and 2) direct effects on physical habitat, which results in indirect effects on biota. Examples of direct effects on biota include suppression of photosynthesis by shading primary producers; increased drifting of, and consequent predation on, benthic invertebrates; and shifts to turbidity-tolerant fish communities. Indirect effects on biota will occur as the biotic assemblages that rely upon aquatic habitat for reproduction, feeding, and cover are adversely affected by habitat loss or degradation of this habitat. A noteworthy example of indirect effects of SABS in streams and rivers is the loss of spawning habitat for salmonid fishes by an increase in embeddedness, caused by the entrapment of fine material in the gravel. Increased sedimentation can limit the amount of oxygen in the spawning beds which can reduce hatching success, or trap the fry in the sediment after hatching.

The effects of SABS in streams and rivers span the scales of biota. The biological responses to this stressor at a site are related to site-specific effects (turbidity, shading, substrate embeddedness) and to the cumulative loadings of sediments from the catchment above the site. Additionally, the effects of these biological responses at sites are cumulative for the entire catchment, such that catchment-wide assessments of impacts are possible based on the cumulative nature of the stressor. These cumulative effects might show a threshold response, a multiplicative response, or other patterns, when acting on habitats important and unimportant for the various life history stages of a species. There might also be a threshold effect in the case of an extremely mobile fish species, or one that depends upon habitat refugia that are relatively rare.

A widely applicable model of the effects of SABS might be expected to have parameters for different habitats and species which could be plugged in for specific situations. In fact there is little hard evidence in the literature that species from different habitats have different SABS requirements. This is largely because there have been very few studies that compare species from different habitats in the same study, and given the wide range of experimental designs used in the literature it is very difficult to make comparisons between studies.

One way to conduct a between-habitat comparison would be to use the models in Appendix A and D from Newcombe and Jensen (1996) and compare the models from the adult salmonids (which we might assume to be the most sensitive of the groups of adult fishes: Model 2) with the adult freshwater nonsalmonids (which might be assumed to have an intermediate sensitivity: Model 6) and the adult estuarine fishes (which might be assumed to be the least sensitive of the three: Model 5). However, if the empirical data that have been used to generate the models are compared (Newcombe and Jensen, 1996: Figures 2a, 5a and 6a), it is clear that there are not enough data to make a rigorous comparison between the models.

As stated above, another way to make a comparison between habitats is to expose organisms that live in different habitats to suspended sediments using identical experimental protocols. These types of experiments have been conducted at least twice. McFarland and Peddicord (1980) exposed a number of organisms to varying levels of kaolin in suspension. They found that the organisms restricted to muddy bottoms were very insensitive to high suspended clay concentrations. Some open water fish, fouling organisms, and sandy bottom epifauna were relatively sensitive. However, there were tolerant species identified from both groups. One particularly interesting comparison was between two members of the same genus. *Mytilus californianus*, a mussel from rocky coastal environments was more sensitive than the closely related blue mussel, *Mytilus edulis*, usually found in bays and harbors, which may be more turbid.

Cyrus and Blaber (1987a) examined, in the laboratory, the turbidity preferences of the juveniles of 10 species of fish which inhabit a large estuarine system in southeastern Africa. They compared these preferences to the field abundances of the same species in habitats with different turbidities. They found that the turbidity preferences of the fishes varied from species to species. Species which were typically found in highly turbid habitats generally preferred turbid water in the laboratory. Species which were found in clearer water in the field generally preferred clearer water in the laboratory. Both of these studies seem to indicate that the expected relationship between habitat and SABS tolerance exists. However, much more of this sort of experimental work needs to be done if this toxicological approach is to be used. In particular, it would be useful to conduct additional studies of closely related species that live in habitats with different levels of SABS.

Another reason that it is difficult to compare the effects of SABS between habitats is that most of the research on the effects of SABS has been done in streams. This is because some of the most obvious consequences of elevated SABS are in stream systems, often leading to complete loss of salmonid fisheries. Many miles of streams in the U.S. are listed as not meeting designated uses (303(b) reports), but other habitats are also impacted by increased SABS (Table 1). Of these coral reefs are among the most studied (Appendix C and Coral Reef section of this document). There have been far fewer studies done in freshwater river and lake habitats and estuaries, but several indicate that the biota in these habitats can be very sensitive to increases in SABS.

Some lake species, like the Bear Lake sculpin for example, require large cobble and boulders for spawning (Ruzycki et al, 1998). Gravel and sand embedded habitats are not suitable for spawning. An increase of the supply of sand to this system could further restrict the habitat of this endemic species. One additional problem in lake habitats is that it might take them much longer to recover from increased sedimentation than it takes for a flashy stream to recover. Spring freshets can resuspend fine materials from streams, and move them down stream, thus restoring a coarse-grained bottom. This is less likely to happen in a lake.

Effects of SABS

Summaries of the Effects of SABS

Summarizing effects data for SABS is difficult for several reasons. One reason is that there is not one agreed-upon measurement for SABS. Caux et al. (1997) provide an excellent discussion of the various methods of measuring suspended sediments. Suspended sediments contribute to turbidity and thus affect light transmission through the water column (Waters, 1995). Turbidity is an optical property of water resulting in a decrease in light transmission due to absorption and scattering. Consequently turbidity is a key water quality parameter in aquatic systems in that it has a predominant influence on the compensation point (the depth at which photosynthesis equals respiration in plants) and is therefore a critical determinant in the distribution of submerged aquatic vegetation (SAV) (Batuik, et al., 1992). The correlation of turbidity with concentrations of suspended solids (mg/L) is impractical because the size, shape, and refractive index of particulate material affect turbidity but are not directly related to the concentration of suspended solids (Caux et al., 1997), and thus the correlation is site-specific. Various measurements are used for bedded sediments as well. These include depth of deposition within a given time period, percent fines, geometric mean diameter, and Fredle number (Caux et al., 1997). (Fredle number is an index of permeability that has been found to correlate well with survival-to-emergence of salmon and trout (Lotspeich and Everest, 1981)).

Another reason summarizing effects data for SABS is difficult is that there are no standard durations for SABS effects testing. Both the duration (Newcombe and MacDonald, 1991) and frequency (Shaw and Richardson, 2001) of sediment exposures are important. For example, some species are able to recolonize between sediment events, while some other species may not be able to recover before the next event (Yount and Nimmi, 1990).

Newcombe and MacDonald (1991) recognized that the appropriate way to report data for the effects of suspended sediment on aquatic organisms was to include information on duration of exposure, as well as exposure concentration. Up until that point, the importance of duration of exposure had been largely overlooked. They summarized, in graphical and tabular form, much of the available data on the effects of SABS on fish and invertebrates. Newcombe and Jensen (1996) presented an extensive data table of the effects of SABS on fish, and went a step further developing empirical models of the effects of SABS on fish. Newcombe also developed a model for the effects of SABS on aquatic invertebrates and flora (Newcombe, 1997) and another dealing with the effects of diminished water clarity on fish (Newcombe, 2003). These models are included in Appendices A and B.

A recent review of the biological effects of suspended sediments on fish and shellfish was conducted by Wilber and Clarke (2001). Their paper synthesized the results of studies that report the dose-response relationships of estuarine aquatic organisms to suspended sediments and then related those findings to sediment conditions associated with dredging projects. Dose-response graphs were modified from Newcombe and Jensen (1996) to provide an easy reference for estimating biological responses to suspended sediments. Wilber and Clarke (2001) also provide tables that depict biological response as a function of suspended sediment exposure (sediment concentration and duration). Biological response categories reported by Wilber and Clarke (2001) include: no effect, behavioral, sub-lethal, and lethal effects. In this review (Appendix C) we have expanded the tables of Wilber and Clarke (2001) using data from other studies to include fresh water species, corals, and aquatic plants. Studies which did not include measurements of total suspended solids (TSS) were excluded from the tables. For a recent review of the effects of turbidity on fishes, see Newcombe (2003).

Effects on invertebrates

Elevated levels of SABS have been shown to have wide ranging effects on both pelagic and benthic invertebrates (Cordone and Kelly 1961; Maurer et al., 1986; Peddicord,1980; Waters, 1995; Wilber and Clarke, 2001). Effects can be classified as having a direct impact on the organism due to abrasion, clogging of filtration mechanisms thereby interfering with ingestion and respiration, and in extreme cases smothering and burial resulting in mortality. Indirect effects stem primarily from light attenuation leading to changes in feeding efficiency and behavior (i.e., drift and avoidance) and alteration of habitat stemming from changes in substrate composition, affecting the distribution of infaunal and epibenthic species (Donahue and Irvine, 2003; Waters, 1995; Zweig and Rabeni, 2001).

Increased levels of suspended sediment were shown to impair ingestion rates of freshwater mussels in laboratory studies (Aldridge et al., 1987). However, Box and Mossa (1999) reviewed the literature on the effects of sedimentation on freshwater mussels and concluded that the relative significance of human activities to sediment production, and their susbsequent effects on freshwater mussels, is difficult to evaluate. Reduced feeding activity as a response to increased levels of suspended sediments has also been reported for copepods (Tester and Turner, 1988; Sherk et al., 1976) and daphnids (Arruda et al., 1983). Invertebrate drift is directly affected by increased suspended sediment load in freshwater streams and lakes. Increases in suspended sediments (e.g., 120 mg/l) can result in increased drift, significantly altering the distribution of benthic invertebrates in streams (Herbert and Merkens, 1961).

Waters (1995) considers the effects of increased deposition of sediments on benthic invertebrates as one of the most important concerns within the sediment pollution issue, especially in regards to the dependence of freshwater fisheries on benthic productivity. Waters (1995) identifies three major relationships between benthic invertebrate communities and sediment deposition in streams: 1) correlation between abundance and substrate particle size, 2) embeddedness of substrate and loss of interstitial space, and 3) change in species composition with change in type of habitat (substrate composition).

Alteration in the quality and quantity of deposited sediments can affect the structure and function of benthic macrofaunal communities by increasing substrate embeddedness and altering substrate particle size distributions (Erman and Erman, 1984). Increased embeddedness can result in decreases in aquatic insect densities and small increases in siltation can directly affect caddisfly pupa survival. Zweig and Rabeni (2001) examined the response of benthic infauna to deposited fine sediments in four Missouri streams. Five biomonitoring metrics were significantly correlated with deposited sediments across streams. Deposited-sediment tolerance values were developed representing responses to deposited sediments for 30 taxa. Tolerance values where then used to develop the Deposited Sediment Biotic Index (DSBI). The DSBI was calculated to characterize sediment impairment in the four streams. DSBI values for each site examined were highly correlated with depth and degree of embeddedness of deposited sediment.

Several studies have examined the effects of the burial of estuarine invertebrates. Maurer et al. (1986) found that species responded differently to burial by 36-40 cm of sediment, and that some organisms were able to migrate more easily up through sandy sediment, while other organisms were able to migrate better through muddy sediment. Hinchey et al. (in review) found that species-specific response to burial by sediments varied as a function of motility, living position and inferred physiological tolerance of anoxic conditions while buried. Their study compared responses of five estuarine invertebrate species (3 infaunal and 2 epifaunal) to clean sediment burial in laboratory experiments. Hinchey et al. (in review) suggested that effective overburden stress, which incorporates both the bulk density of the sediment as well as the depth of burial (Richards et al., 1974), was a better measure of the force exerted on organisms by sediment burial than depth of sediment alone.

Effects on Corals

The increased sedimentation resulting from coastal development is a major source of coral reef degradation (Rogers, 1983, 1990; Torres, 2001). Excessive sedimentation can adversely affect the structure and function of the coral reef ecosystem by altering physical and biological processes (Rogers, 1990). High sediment loads can smother tissue resulting in bleaching in the short-term and death in the long-term (Rogers, 1983).

Cortes and Risk (1985) reported reduced growth rates in *Montastraea annularis* living in waters with average sedimentation rates between 20-1,000 mg cm^{-2} d^{-1}. Reduced growth rates and temporary bleaching in *M. annularis* were also reported by Dodge et al. (1974). In a subsequent study, Torres (2001) showed that growth rates of *M. annularis* were significantly lower and negatively related with sediment deposition rates and percentages of terrigenous sediments deposited on a coral reef on the south coast of Puerto Rico. Nemeth and Nowlis (2001) reported bleaching of coral colonies at sediment deposition rates between 10 and 14 mg $cm^{-2}d^{-1}$. Their study indicated that stress from sedimentation may lead to a decline in living coral. An indirect effect of increased suspended sediment load was an increase in turbidity, which caused a corresponding decrease in light penetration that limited the photosynthetic capacity of symbiotic zooxanthellae, and furthered the decline in coral populations.

Excessive sedimentation can affect the complex food web associated with coral reefs, killing not only corals but other reef dwelling organisms (e.g., sponges) which serve as food for commercially important fish and shellfish (Rogers, 1990). Declines in tropical reef fisheries in the Carribean and the Pacific are believed to be partially due to increased sedimentation rates (Rogers, 1985; Dahl, 1985). Increased sedimentation is also one of several factors which affect coral recruitment. Coral larvae will not settle and establish themselves in shifting sediments. Consequently, increases in sedimentation rates can alter the distribution of corals and their associated reef constituents by influencing the ability of coral larvae to settle and survive (Rogers, 1990).

Effects on Aquatic Plants

Some populations of aquatic macrophytes have experienced dramatic losses over the past two decades, a decline largely attributed to changes in underwater light climate due to increases in suspended sediment concentrations (Best et al., 2001). Turbidity limits the growth and distribution of aquatic plants by reducing available light. The large-scale declines of submerged aquatic vegetation (SAV) reported in Chesapeake Bay are believed to be directly related to increasing amounts of nutrients and sediments entering the Bay (Batiuk et al., 1992, 2000; Dennison et al., 1993). To address the unacceptable Bay-wide decline in SAV the U.S.EPA Chesapeake Bay Program office established water clarity criteria. Water clarity criteria are based on the light requirements for SAV growth and survival. The criteria take total suspended solids (particulate matter and chlorophyll a) into account, as well as epiphytic growth and salinity regime. Water clarity criteria are used in Chesapeake Bay because it is assumed that they will result in achievement of clarity/solids levels that would not impair other habitats/organisms (with the exception that the water clarity criteria may not fully protect "smothering" of bottom soft or hard bottom habitats with larger sized sediment particles from sources that "by-pass"/don't influence shallow water habitats), since the SAV represent one of the components of the Chesapeake Bay ecosystem that is most sensitive to increases in SABS. A detailed explanation of the derivation of Chesapeake Bay water clarity criteria can be found in U.S. EPA (2003b).

SAV are also subject to burial, although different species have different tolerances for sediment accretion, and different sediment entrainment qualities (Fonseca and Fisher, 1986). These different tolerances can result in changes in species composition in addition to overall loss of SAV as a result of increased siltation (Terrados et al., 1998). It is not always possible to separate out the effects of burial from the other effects of increased sediment input, e.g. reduced light penetration (Terrados et al., 1998).

Effects on fish

Of all of the taxonomic groups, fishes, particularly salmonids, have received the most attention from SABS researchers. This is because of the commercial and recreational importance of salmonids, and the obvious impact that logging and other land use activities have had on salmonid fisheries, particularly in the Pacific northwest (Waters, 1995). There are three major effects of SABS on fishes: 1) direct physiological effects of suspended sediment, such as suffocation, 2) effects due to decreases in water clarity, and 3) effects due to sediment deposition, leading to increased embeddedness or burial of eggs and larvae (Waters, 1995; Wilber and Clarke, 2001).

The conventional wisdom (at least since the publication of Newcombe and MacDonald, 1991) is that both the degree of exposure (measured as TSS or turbidity, or decreased water clarity) and the duration of the exposure are important. It follows that the longer the duration and the greater the exposure, the more severe the effects. Therefore, it is expected that the first, mild, primarily behavioral effects would be seen with low intensity, short-term exposures. As the duration of exposure and intensity of exposure increase, sublethal effects are manifested, and lethal effects begin to be expressed at more intense exposures of longer duration (Figure 2). The timing of exposure to suspended sediment is also very important, as it may affect different life-history stages in different ways. Different life-history stages of the same species may also have differing abilities to avoid exposure.

Effects of Suspended Sediment on Fish

Newcombe and Jensen (1996) summarized much of the available data on the effects of suspended sediment on fishes, and fit the data into empirical models in the form of data “triplets”, with matched biological effect, concentration and duration information. The effects were scored on a qualitative “severity of ill effect” (SEV) scale, that included responses ranging from no behavioral effects (lowest on the scale) to behavioral effects (low on the scale), to sublethal effects (higher on the scale), to lethal effects (highest on the scale). Different models were developed for different age groups of fishes: juvenile and adult salmonids together, adult salmonids, juvenile salmonids, eggs and larvae of salmonids and non-salmonids, adult estuarine non-salmonids, and adult freshwater non-salmonids. The models were presented both in visual form (as 3-dimensional response surfaces) and as linear regression equations, and were also used to interpolate and extrapolate from the empirical data. The tabular forms of the models are presented in Appendices A and D. They are taken from Newcombe (1997) and Newcombe (personal communication). Appendix A also includes an empirical model for the effects of suspended sediments on invertebrates as well as an empirical model for plants. Appendix D corrects the error in the estuarine adult fish model from Newcombe and Jensen (1996) identified by Wilber and Clarke (2001). Although the visual presentations in Newcombe and Jensen (1996) of the models look complete, it is evident from the figures of the “empirical data” (Appendix A) that there are not enough data for the various groups of organisms (with the possible exception of the salmonids) to fill in the idealized model of fish response to increased suspended sediments shown in Figure 2. This is because there are not enough data, and because of the great variability in the data.

Wilber and Clarke (2001) published another review of the data on the effects of SABS on fish, focusing on impacts of dredging on estuarine organisms. They added to the data from Newcombe and Jensen (1996) and provided a useful way of plotting the empirical data, such that all of it can be seen and compared with expected exposure concentrations (in their case, from dredging operations). These figures are presented in Appendix E. This display of data provides a powerful tool for the estimation of expected effects from a given suspended sediment exposure scenario. When looking at the figure from Wilber and Clarke (2001) describing the data for salmonid fishes (Figure 2 in Appendix E), there does appear to be enough data from studies with adult salmonids to begin to visualize the idealized pattern seen in Figure 2. However, Wilber and Clarke (2001) also plotted the adult estuarine fish data separately from the freshwater and salmonid data, to show how little data there were for the fishes, and that most of those data were from short duration tests at very high exposures (Figure 4 in Appendix E).

Effects of Decreased Water Clarity on Fish

Wilber and Clarke (2001) also summarized the effects of increased turbidity and reduced water clarity on the feeding of fishes, but did not include the data in their tables or figures, because most of them are reported in turbidity units which are difficult to convert to suspended solids concentrations (Caux et al., 1997). It is very difficult to make generalizations about these data. Some fishes are able to hunt better as suspended solids increase, at least up to a point, because of increased contrast between the prey and the surrounding water. Some larval fish, like striped bass, seem to be able to feed under extremely turbid conditions, or even complete darkness. This ability could be very important for a fish that follows the turbidity maximum for its abundant food (Chesney, 1993).

Centrarchids (e.g., smallmouth and largemouth bass), on the other hand, may be severely impacted in their ability to feed by even small increases in turbidity (J. Sweeten, personal communication). Suspended sediment has little if any effect on the nests of centrarchids due to their nesting behavior of "fanning" eggs (J. Sweeten, personal communication). However, low concentrations of suspended sediment caused reduced growth in smallmouth bass (*Micropterus dolomieui*). The inhibition concentration (IC) 25 value for a one day exposure was only 11.4 mg/L suspended bentonite (Sweeten and McCreedy, 2002). The authors concluded that even low concentrations of suspended sediment at this early life-stage may strongly affect year class strength. Other fish may be excluded from desirable habitat because of increased turbidity (Ponton and Fortier, 1992).

Despite the difficulties in putting together the data on the effects of turbidity on fishes, Newcombe (2003) has developed an impact model for clear water fishes exposed to excessively cloudy water. This is discussed in the modeling section below.

Effects of Increased Sedimentation on Fish

The effects of increased SABS resulting in increased embeddedness, on salmonids in particular, have been well documented (e.g., Waters, 1995). An increased supply of fine sediment to a stream can cause the gravel interstices of a stream bed to be filled in. This process can cause reduced hatching due to the reduction in flow through the stream bed and the resulting decrease in dissolved oxygen. It can also cause reduced larval survival because of armoring of the sediment surface which traps the larvae. Increased sedimentation in other habitats (e.g., estuaries) can cause burial of eggs (Wilber and Clarke, 2001). Even a small amount of deposited sediment can cause a problem. Winter flounder eggs, for example, will suffer reduced hatching success if buried to only one half an egg diameter (D. Nelson, NMFS, unpublished data).

Effects on Wildlife

There are very few published reports on the effects of SABS on aquatic-dependent wildlife (i.e., birds and mammals). For the most part, aquatic-dependent wildlife are more mobile than the fish, invertebrates and plants discussed above, and therefore aquatic-dependent wildlife can avoid most of the direct effects of increased SABS. A heron or an osprey, for example, can avoid more turbid areas, and choose areas of clearer water. If and when the water clears in the area, the bird can return. If increases in SABS are wide-spread and long-term, however, they might cause a problem for aquatic-dependent wildlife that consume aquatic prey. A bear, for example, may have to abandon part of its range if there is failure of a salmon run. Loons are thought to require clear water for fishing, and may avoid nesting areas with inadequate water clarity (McIntyre, 1988).

Most of the studies of the relationship between turbidity and aquatic-dependent wildlife involve field studies with birds. Van Eeerden and Voslamber (1995) describe a mass (group) fishing behavior of cormorants, which was apparently developed as a response to an increase in the turbidity of a lake in the Netherlands. Stevens et al. (1997) found that waterbirds were most abundant on the clear and variably turbid segments of the Colorado River and least abundant on the more turbid lower segment, providing evidence that turbidity makes it difficult for birds to forage effectively. Another study in British Columbia ponds, however, found that the abundance of dabbling ducks was positively correlated with turbidity and total dissolved nitrogen, and negatively correlated with percent of forested shoreline, percent of marsh, and chloride (Savard et al., 1994). The authors had no explanation for these relationships, and felt that their results highlighted the problems associated with interpreting correlative-type studies, especially the difficulties in assessing the biological significance of the observed correlations.

Modeling the Effects of Increased SABS

The preceding discussion indicates that the effects of SABS on aquatic life are complicated, and unraveling them may be difficult. However, at least one expert in the field feels that we are well on our way to developing models that can predict the effects of SABS on fish, at least in streams. Newcombe (2000) presents a primer with information on water quality and sediment quality models for assessing the impact of excess stream channel sediment, and provides a framework for their use. The principle is simple: if the SABS problem in a stream is related to suspended sand and silt, a suspended sediment model should be used; if the problem in the stream relates to suspended clay particles, a water clarity model should be used; and if the problem relates to sediment deposition, a sediment quality model should be used. The models for the effects on fish in streams have the most data and are the most complete, but by extension they may be used in other habitats. For suspended sand and silt problems, models like those in Newcombe and Jensen (1996) should be used. A model for the effects on invertebrates is also included in Appendix A. The model for estuarine fishes has been corrected, as suggested by Wilber and Clarke (2001) (Appendix D). For excess clay a turbidity model, such as the draft model in Appendix B, can be used (Newcombe, 2003). There are three sediment quality models presented in Newcombe (2000). These models are from Crouse et al. (1981); Kondolf (1997), and Tappel and Bjornn (1983). Additional sediment models are reviewed in Caux et al. (1997). The variables most often used to assess the composition of streambed sediments are percent fines, geometric mean diameter, and Fredle number (Caux et al., 1997). All of these variables can be used to develop empirical models of salmonid hatching as a function of sediment composition.

The models described above consider the effects of SABS as a single stressor, but organisms in nature are exposed to multiple chemical and physical stressors. Most work considering the interaction of increased sediment and chemical contaminants has addressed the effects that suspended sediment can have on the bioavailability of contaminants. Increased suspended sediment can decrease the bioavailability of hydrophobic contaminants by reducing dissolved concentrations in the water column (Schrap and Opperhuzien, 1990).

A few studies, however, have examined the interaction of suspended sediment and toxicants. Herbrandson and colleagues found that increased suspended sediment load could decrease the EC50 concentration of carbofuran to *Daphnia* by a factor of five (Herbrandson et al., 2003a). They developed a model of the combined effects of suspended sediment and carbofuran. These effects were more than additive (i.e., the measured EC50s were lower than would be predicted by an additive model). They hypothesized that this reduction was due to a reduction in feeding efficiency of the test organisms in the presence of increased suspended sediment (Herbrandson et al., 2003b). The possibility of interactive effects is a real problem for SABS criteria-setting based on laboratory testing, because of the huge number of possible interactions between SABS and other stressors. Herbrandson et al. (2003a) found that there was an interaction of carbofuran and suspended sediment, even though there was no increased mortality due to very high levels of suspended sediment in the absence of carbofuran.

Current Criteria for SABS

One of the best available summaries of the current criteria for SABS is in the Technical Appendix to the *Ambient Water Quality Guidelines (Criteria) for Turbidity, Suspended and Benthic Sediments* (Caux et al., 1997), prepared for the British Columbia Ministry of Environment, Land and Parks. British Columbia has SABS criteria for a number of water uses, varying from drinking water to aquatic life use to industrial water supply. Caux et al. (1997) outline the criteria for each, and provides the scientific rationale. Caux et al. (1997) build on an earlier review of available criteria by Singleton (1985). A more recent review of current SABS criteria was done by K. Sullivan for the Office of Water (Appendix F.).

There is a wide range of criteria in current use in the United States (Appendix F). Some states use numerical criteria, some use narrative criteria, some use both, and some states have no criteria for SABS. Many states have different criteria for different stream channel substrate types. When they are differentiated, states typically have more stringent criteria for streams with hard substrates (gravel, cobble, bedrock) and less stringent criteria for streams with soft substrates (sand, silt, clay). Hawaii has a separate criteria for reefs. Cold water fisheries typically have more stringent criteria than do warm water fisheries in states that differentiate between the two. A few states use biocriteria (e.g., biotic indices), and at least one uses soil loss as a criterion. Several states provide criteria for an averaging period (e.g., 30 days) as well as an allowed daily maximum concentration.

Most states with numerical criteria use turbidity as a measure. Some use exceedances over background (e.g., “Not greater than 50 NTU over background”, or “not more than 10% above background”), while some use absolute values (e.g., “Not greater than 100 NTU”). Only a few states use suspended solids as a criterion. Suspended sediment criteria values vary from 30 mg/L up to 158 mg/L. At least one state uses transparency ($\geq$ 90% of background) as a standard. A number of states have criteria based on sediment deposited over a time period, or during a storm event. Values are typically 5 mm during an individual event (e.g., during the 24 hours following a heavy rainstorm) for streams with hard substrates bottoms and 10 mm for streams with soft bottoms. Hawaii's reef criterion is 2 mm deposited sediment after an event.

The Chesapeake Bay Program’s unique criterion is discussed in the plant section above. The criterion is based on suspended particulate matter and chlorophyll *a*, and takes into account epiphytic growth and salinity in its calculation of water clarity (U.S.EPA, 2003b).

Many states have narrative criteria for SABS in addition to, or in lieu of, numerical criteria. These criteria most frequently pertain to turbidity or appearance of the water (e.g., “free of substances that change color or turbidity”). Others refer to undesirable biological effects (e.g., “no adverse effects” or “no actions which will impair or alter the communities”). Given the wide range of measures (e.g., turbidity, TSS, color) used to measure SABS and the wide range of values within a given measure, it is difficult to evaluate what appropriate criteria should be, especially because the rationale for the criteria are not always readily available. The British Columbia SABS criteria for aquatic life are one of the few examples of criteria explicitly supported by a scientific rationale (Caux et al., 1997). Idaho's Guide to Selection of Targets for Use in Idaho TMDLs (Idaho DEQ, 2003) is another good example of a document outlining SABS criteria with explicit biological justification.

Upon first glance, the channel substrate and fisheries specificity in the criteria from across the U.S. (Appendix F), might lead one to believe that the observed variation in criteria is related to regional variation in SABS across the country. However, as is noted in other parts of this review, it does not appear that the data are sufficient to back up this contention.

“State of the Science” for SABS criteria setting

A full “state of the science review” might review all of these approaches for setting criteria for SABS and the steps needed to implement them, but that is beyond the scope of this review. Here we will focus on some of the gaps in our understanding of the effects of SABS, using the conceptual model in Figure 1 as an outline for the SABS processes affecting aquatic and aquatic-dependent life.

Our understanding of the physical processes controlling the input of sediments to aquatic systems is better developed than our understanding of the effects of these inputs. The input of sediment to streams from watershed activities is probably the most studied. Changing land use can often result in greatly increased sediment loading to streams (U.S.EPA, 1999; Leopold et al., 1964; Rosgen, 1996). Deposition due to dredged material disposal has been largely predicted using the "ADDAMS" models developed by the US Army Corps of Engineers (Schroeder and Palermo, 1995). Resuspension and deposition (both from barge overflow and resuspension) at the dredging site will be modeled with a new model, SSFATE (Suspended Sediment Fate, D. Clarke, personal communication). Deposition and resuspension from natural processes are outside of the scope of this review because the organisms in different habitats have evolved to survive in the resuspension and deposition regime native to their habitat, although a good estimate of natural resuspension and deposition helps to put anthropogenic increases in these phenomena into context. Erosion is primarily a concern in marsh habitats (Boesch,et al., 1994), and results in part from a decrease of sediment supply to some waterbodies.

Among the biological effects due to suspended sediments the most important are smothering (and abrasion), shading, and reduced feeding due to increased turbidity. Of these, the shading of SAV has been the most heavily studied, and is probably the best understood; in fact, criteria based on models of shading of SAV are available (U.S.EPA, 2003b). Models are also available for the prediction of the direct effects of suspended sediment (smothering and abrasion) on fish and invertebrates (Appendices A and D; Newcombe and Jensen, 1996), but most of the data used to support these models come from unrealistically high and short-term exposures. Also, there has been little field validation of these models. Further work with longer term and more environmentally realistic exposures will be required before the real effects of suspended sediment on fishes in the environment can be understood (Wilber and Clarke, 2001). We know less about the effects of suspended sediment on other groups of organisms, including zooplankton and aquatic-dependent wildlife.

Studies of the effects of suspended sediments on feeding have been done primarily with larval fish, and a model is available (Appendix B; Newcombe, 2003), but there has been little field validation of the model. We know less about the effects of suspended sediment on the feeding of other groups of organisms.

Wilber et al. (in review) conducted a review of the effects of burial associated with dredging as a followup to Wilber and Clarke's (2001) review of the effects of suspended sediment. They concluded that "Overall, the literature available to determine whether elevated sedimentation rates, hypothetically linked to dredging and disposal, can result in impacts to sensitive resources and other biota is scant and varies widely between habitats. Very thin veneers of sediment are known to adversely affect both settlement and recruitment of bivalve larvae. Some quantitative data are available for eggs of demersal fish, both for cover and changes to particle size of the substratum composition. Although there are documented, unambiguous, adverse effects of sedimentation on fishes, seagrasses and submerged aquatic vegetation, available data have not been and are insufficient to be transformed into target values."

Wilber et al. (in review) further concluded that "documentation of how either natural or dredging-induced sedimentation rates affect targeted biological communities is needed. There are insufficient data for all habitat types investigated to establish dose-response models (particularly with parameters appropriate to dredging) as would be required for predicting potentially harmful rates of sedimentation or establishing technically defensible guidelines for their protection. Work to date relating sedimentation to impacts on resources can generally be classified as either (1) manipulative experiments in which varying amounts of sediment are added to a targeted system, or (2) *a posteriori* determinations of causes and effects following major sedimentation events (e.g., dredging operation, storm). The latter retrospective approach suffers from confounding factors acting synergistically with or independently from sedimentation, such as elevated suspended sediment load, changes in nutrient supply, or other related environmental perturbations. Unfortunately, most reports of sedimentation impacts fall into the latter category."

The general conclusion from the analysis in this review is that, as the water clarity criteria for the protection of SAV in the Chesapeake show, the toxicological approach can be used if a species or group of species from a particular habitat is to be protected and the required dose-response data are available. Currently models of the biological effects of SABS are in use for the effects of shading on SAV, the direct effects of suspended sediments on fishes, the effects of water clarity on larval fishes, and the effects of embeddedness on the hatching of salmonids. When these models have received more field validation and are made more generalizable it may be possible to set national criteria for suspended or bedded sediment using the traditional "toxicological" dose-response approach. These criteria will presumably have to incorporate some habitat-specificity in order to be widely applicable.

Take home messages

1) Some useful models for the biological effects of SABS exist and others are under development. As the water clarity criteria for the protection of SAV in the Chesapeake show, the approach can be used if a specific species from a particular habitat is to be protected and the required dose-response data are available. Generalizations are difficult because biological response to both increased suspended sediment and increased bedded sediment varies with species and sediment characteristics.

2) After additional research it may be possible to develop national scientifically-defensible SABS criteria using the traditional "toxicological" dose-response approach. These criteria will presumably have to incorporate some habitat-specificity in order to be widely applicable.

3) Some habitats that have not been well studied (in terms of their sensitivity to SABS) deserve more study, especially those habitats with moderate and variable amounts of SABS.

4) Many states have set standards for SABS, but there is little consistency among them.

Acknowledgements

A number of people have contributed greatly to this review. C.P. Newcombe provided much insight into the problems of SABS, as well as providing the unpublished material for several of the Appendices. Dara Wilber provided valuable suggestions, and provided the figures for

Appendix E. The review also profited from discussions with Don MacDonald, Doug Clarke, and Jerry Sweeten. Thanks to Timothy Gleason, Elizabeth Hinchey, Phil Kaufman, Lesa Meng, Wayne Munns, and John Paul for the critical and constructive reviews that they provided. The conceptual model cartoon was modified from one provided by Wayne Munns. The research described in this paper has been funded by the U.S. Environmental Protection Agency. This paper has not been subjected to Agency review. Therefore, it does not necessarily reflect the views of the Agency. Mention of trade names or commercial products does not constitute endorsement or recommendation for use. This is contribution number AED-02-103.

References (Note: some of these references apply only to Appendix C)

Aldridge, D.W., B.S. Payne, and A.C. Miller. 1987. The effects of intermittent exposure to suspended solids and turbulence on three species of freshwater mussels. Environmental Pollution 45 :17-28.

Arruda, J.A., G.R. Marzolf and R.T. Faulk. 1983. Role of suspended sediments in the nutrition of zooplankton in turbid reservoirs. Ecology. 64:1225-1235.

Batiuk, R. A., P. Bergstrom, M. Kemp, E. Koch, L. Murray, J. C. Stevenson, R. Bartleson, V. Carter, N. B. Rybicki, J. M. Landwehr, C. Gallegos, L. Karrh, M. Naylor, D. Wilcox, K. A. Moore, S. Ailstock and M. Teichberg. 2000. Chesapeake Bay submerged aquatic vegetation water quality and habitat-based requirements and restoration targets: A second technical synthesis. CBP/TRS 245/00 EPA 903-R-00-014. U.S. EPA Chesapeake Bay Program, Annapolis, Maryland.

Batiuk, R. A., R. Orth, K. Moore, J. C. Stevenson, W. Dennison, L. Staver, V. Carter, N. B. Rybicki, R. Hickman, S. Kollar and S. Bieber. 1992. Chesapeake Bay submerged aquatic vegetation water quality and habitat-based requirements and restoration targets: A technical synthesis. CBP/TRS 83/92. U.S. EPA Chesapeake Bay Program, Annapolis, Maryland.

Best, E.P.H.,C.P. Buzzelli, S.M. Bartell, R.L. Wetzel,W.A. Boyd, R.D. Doyle, and K.R. Campbell. 2001. Modeling submersed macrophyte growth in relation to underwater light climate:modeling approaches and application potential. Hydrobiologia. 444:43-70.

Boesch, D.F., M.J. Josselyn, A.J. Mehta, J.T. Morris, W.K. Nuttle, C.A. Simenstad, and D.J.P. Swift. 1994. Scientific assessment of coastal wetland loss, restoration and management in Louisiana. Journal of Coastal Research Special Issue 20:101-103.

Box, J.B. and J. Mossa. 1999. Sediment, land use, and freshwater mussels: prospects and problems. Journal of the American Benthological Society. 18:99-117.

Bricelj, V.,R., Malouf, and C. De Quillfeldt. 1984. Growth of juvenile *Mercenaria mercenaria* and the effect of resuspended bottom sediments. Marine Biology. 84: 155-164.

Cardwell, R.D., C.E. Wolke, M.I. Carr, and E.W. Sanborn. 1976. Sediment and elutriate toxicity to oyster larvae. Pp 687-718 in P.A. Krenkel, J.Harrison, and J.C. Burdick III. Eds. Proceedings of the speciality conference on dredging and its environmental effects. American Society of Civil Engineers, Mobile, AL.

Caux, P. -Y., D.R.J. Moore, and D. MacDonald. 1997. Ambient water quality guidelines (criteria) for turbidity, suspended and benthic sediments. Technical Appendix. Prepared for BC Ministry of Environment, Land and Parks. April, 1997.

Chesney E.J. 1993. A model of survival and growth of striped bass larvae *Morone saxatilis* in the Potomac River, 1987. Marine Ecology Progress Series. 92 :15-25.

Cordone, A.J. and D.W. Kelly. 1961. The influence of inorganic sediment on the aquatic life of streams. California Fish and game. 47:189-228.

Cortes, J., and M. Risk. 1985. A reef under siltation stress: Cahuita, Costa Rica. Bulletin Marine Science. 36:339-356.

Crouse, M.R., C.A. Callahan, K.W. Malueg, and S.E. Dominguez. 1981. Effects of fine sediments on growth of juvenile coho salmon in laboratory streams. Transactions of the American Fisheries Society. 110:281-286.

Cyrus D P. and S.J.M. Blaber. 1987a. The Influence of Turbidity on Juvenile Marine Fishes in Estuaries Part 1. Field Studies at Lake St. Lucia South Africa on the Southeastern Coast of Africa. J Exp Mar Biol Ecol 109: 53-70.

Cyrus, D. P. and S. J. M. Blaber. 1987b. The influence of turbidity on juvenile marine fishes in estuaries. Part 2. Laboratory studies, comparisons with field data and conclusions. Journal of Experimental Marine Biology and Ecology. 109: 71-91.

Daborn, G.R. and M. Brylinsky. 1981. Zooplankton diversity and species associations in the inner Bay of Fundy. Estuaries. 4: 253.

Dahl, A.L. 1985. Status and conservation of South Pacific coraL reefs. Proc. 5th Int. Coral Reef Congr. 6:509-513.

Davis, H.C., and H. Hidu. 1969. Effects of turbidity-producing material on eggs and larvae of three genera of bivalve mollusks. The Veliger. 11:316-323.

Dennison, W. C., R. J. Orth, K. A. Moore, J. C. Stevenson, V. Carter, S. Kollar, P. W. Bergstrom and R. A. Batiuk. 1993. Assessing water quality with submersed aquatic vegetation habitat requirements as barometers of Chesapeake Bay health. Bioscience 43:86-94.

Dodge, R. E., Aller, R.C. & Thomson, J. (1974). Coral growth related to resuspension of bottom sediments. Nature. 247: 574-577.

Donahue, I and K. Irvine. 2003. Effects of sediment particle size composition on survivorship of benthic invertebrates from Lake Tanganyika, Africa. Archive fuer Hydrobiolgie. 157:131-144.

Erman, D.C. and N.A. Erman. 1984. The response of stream invertebrates to substrate size and heterogeneity. Hydrobiologia. 108:75-82.

Fonseca, M. S., and J. S. Fisher. 1986. A comparison of canopy friction and sediment movement between four species of seagrass with reference to their ecology and restoration. Marine Ecology Progress Series. 29:15-22.

Herbert, D.W. M. and J.C. Merkens, 1961. The effect of suspended mineral solids on the survival of trout. International Journal of Air and Water Pollution. 5:46-55.

Herbrandson, C., S.P. Bradbury, and D.L. Swackhamer. 2003a. Influence of suspended solids on acute toxicity of carbofuran to *Daphnia magna* I. Interactive effects. Aquatic Toxicology. 63:333-342.

Herbrandson, C., S.P. Bradbury, and D.L. Swackhamer. 2003b. Influence of suspended solids on acute toxicity of carbofuran to *Daphnia magna* II. An evaluation of potential interactive mechanisms. Aquatic Toxicology. 63:343-355.

Hinchey, E. K., L.C. Schaffner, C.C. Hoar, B.W. Vogt, and L.P. Batte. In review. Responses of estuarine benthic invertebrates to sediment burial: the importance of mobility and lifestyle.

Idaho DEQ, 2003. Guide to Selection of Targets for Use in Idaho TMDLs. M. Rowe, D. Essig and B. Jessup. June 2003.

Kondolf, G.M. 1997. Application of the pebble count: notes on purpose, method, and variants. Journal of the American Water resources Bulletin. 30: 509-529.

Kuhnle, R.A., and A. Simon. 2000. Evaluation of sediment transport data for clean sediment TMDLs. Proceedings of the Seventh Federal Interagency Sedimentation Conference, March 25 to 29, 2001, Reno, NV. Pp. VII-49 - VII-56.

Kuhnle, R.A., A. Simon, and S.S Knight. 2001. Developing linkages between sediment load and biological impairment for clean sediment TMDLs. Wetlands Engineering & River Restoration Conference, 2001. ASCE, August 27-31, 2001, Reno Nevada.

Leopold, L.B., M.G. Wolman, and J.P Miller. 1964. Fluvial processes in geomorphology. W.H. Freeman and Co., San Francisco.

Lin, H.G., P. Thuet, and J. Trilles. 1992. Effects of turbidity on survival, osmoregulation and gill Na^{+}-K^{+} ATPase in juvenile shrimp, *Penaeus japonicus*. Marine Ecology Progress Series. 90:31-37.

Lotspeich, F. E., and F. H. Everest. 1981. A new method for reporting and interpreting textural composition of spawning gravel. U.S. Forest Service Research Note PNW-369.

Maurer, D. R.T. Keck, J.C. Tinsman, W.A. Leatham, C. Wethe, C. Lord, and T.M. Church. 1986. Vertical migration and mortality of marine benthos in dredged material: a synthesis. International Revue ges Hydrobiology. 71:50-63.

McIntyre, J. 1988. The common loon: spirit of northern lakes. U Minn Press.

McFarland, V.A. and R.K. Peddicord. 1980. Lethality of a suspended clay to a diverse selection of marine and estuarine macrofauna. Archives of Environmental Contamination and Toxicology. 9:733-741.

Naiman, R.J., and R.E. Baily. 1998. River ecology and management. Springer-Verlag, New York, NY.

Nemeth, R. S. and S. Nowlis. 2001. "Monitoring the Effects of Land Development on the Near-Shore Reef Environment. Bulletin of Marine Science Volume 69:759-775.

Newcombe, C.P. 1997. Channel suspended sediment and fisheries: a concise guide. British Columbia Ministry of the Environment, Lands and Parks, Habitat Protection Branch, Victoria, British Columbia, 37 pages. January, 1997.

Newcombe, C.P. 2000. Excess stream channel sediment: six impact assessment models for fisheries streams - a primer. British Columbia Ministry of the Environment, Lands and Parks, Habitat Protection Branch, Victoria, British Columbia. September, 2000.

Newcombe, C. P. 2003. Impact assessment model for clear water fishes exposed to excessively cloudy water. Journal of the American Water Resources Association. 39:529-544.

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

Newcombe, C.P., and D.D. MacDonald. 1991. Effects of suspended sediments on aquatic ecosystems. North American Journal of Fisheries Management. 11:72-82.

Nietch, C.T. and M. Borst. 2001. Suspended Solids and Sediments Risk Management Research for Aquatic Ecosystem Restoration and Protection. Internal report. U.S. EPA Office of Research and Development, National Risk Management Research Laboratory, Water Supply and Water Resources Division, Urban Watershed Management Branch.

Nimmo, D.R., T.L. Hamaker, E. Mathews, and W.T. Young. 1982. The long-term effects of suspended particulates on survival and reproduction of the mysid shrimp, *Mysidopsis bahia*, in the laboratory. Pp. 413-422, in G.F. Mayer, ed. Ecological stress and the New York Bight. Estuarine Research Federation, Columbia, SC.

Peddicord, R.K.. 1980. Direct effects of suspended sediments on aquatic organisms. Pp. 501-536 in R. A. Baker, Editor. Contaminants and Sediments. Volume I. Fate and Transport case studies, modeling, toxicity. Ann Arbor Science Publishers, Ann Arbor MI.

Ponton, D., and L. Fortier. 1992. Vertical distribution and foraging of marine fish larvae under the ice cover of southeastern Hudson Bay. Marine Ecology Progress Series. 81:215-227.

Reid, L.M., and T. Dunne. 1996. Rapid evaluation of sediment budgets. Catena Verlag. Reiskirchen, Germany.

Richards, A.F., T.J.Hirst, and J.M.Parks, 1974. Bulk density-water content relationship in marine silts and clays. Journal of Sedimentary Petrology 44, 1004-1009.

Robinson, W.E., W.E. Wehling, and M.P. Morse. 1984. The effect of suspended clay on feeding and digestive efficiency of the surf clam *Spisula solidissima*. Journal of Experimental Marine Biology and Ecology. 74:1-12.

Rogers, C. 1983. Sublethal and lethal effects of sediments applied to common Caribbean reef corals in the field. Marine Pollution Bulletin. 14:378-382.

Rogers, C.S. 1985. Degradation of Caribbean and Western Atlantic coral reefs and decline of associated fisheries. Proc. 5th int. Coral Reef Congress. 6: 491-496.

Rogers, C. 1990. Responses of coral reefs and reef organisms to sedimentation. Mar. Ecology Progress. Series. 62:185-202.

Rosgen, D. 1996. Applied river morphology. Wildland Hydrology Books, Pagosa Springs, CO.

Ruzycki, J.R., W.A. Wurtsbaugh, and C. Lay. 1998. Reproductive ecology and early life history of a lacustrine sculpin, *Cottus extensus* (Teleostei, cottidae). Environmental Biology of Fishes. 53 :117-127.

Savard J.-P., S.W. Boyd, and J.G.E. Smith. 1994. Waterfowl-wetland relationships in the Aspen Parkland of British Columbia: comparison of analytical methods. Hydrobiologia 279/280:309-325.

Schrap, S.M., and A. Opperhuizen. 1990. Relationship between bioavailability and hydrophobicity: reduction of the uptake of organic chemicals by fish due to the sorption on particles. Environmental Toxicology and Chemistry. 9:715- 724.

Schroeder, P.R. and M.R. Palermo. 1995. The Automated Dredging and Disposal Alternatives Management System (ADDAMS). Prepared by U.S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi. Technical Notes EEDP-06-12. http://www.wes.army.mil/el/elmodels/addainfo.html.

Sellner, K.G. and M.H. Bundy. 1986. Preliminary results of experiments to determine the effects of suspended sediments on the estuarine copepod *Eurytemora affinis*. 16Th Annual Symposium of the Estuarine and Brackish-Water Sciences Association, Plymouth

(UK). Uncles, R.J. (Ed) Dynamics of Turbid Coastal Environments. Contintental Shelf Research. 7:1435-1438.

Shaw, E. A., and J. S. Richardson. 2001. Direct and indirect effects of sediment pulse duration on stream invertebrate assemblages and rainbow trout (*Oncorhynchus mykiss*) growth and survival. Canadian Journal of Fisheries and Aquatic Sciences. 58:2213-2221.

Sherk, J.A., J.M. O'Connor, and D.A. Neuman. 1976. Effects of suspended solids on selected estuarine plankton. Corps of Engineers, Ft.Belvoir, Va, Coastal Engineering Research Center Report No. MR76-1. Jan. 1976.

Singleton, H.J. 1985. Water Quality Criteria for Particulate Matter: Technical Appendix. Ministry of the Environment, Lands, and Parks, Victoria, B.C. pp. 1-82.

Stephan, C.E., D.I. Mount, D.J. Hansen, J.H. Gentile, G.A. Chapman, and W.A. Brungs. 1985. Benchmarks for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. National Technical Information Service, Springfield, VA.

Stevens, L. E., K.A. Buck, B.T. Brown, and N.C. Kline. 1997. Dam and geomorphological influences on Colorado River waterbird distribution, Grand Canyon, Arizona, USA.. Regulated Rivers Research & Management 13:151-169. (Abstract only read)

Suter, G. II, S.B. Norton, and S.M. Cormier. 2002. A methodology for inferring the causes of observed impairments in aquatic ecosystems. Environmental Toxicology and Chemistry. 21:1101-111.

Sweeten, J. and C. McCreedy. 2002. Suspended stream sediment: an environmental indicator of warmwater streams.   319 nonpoint source pollution report, ARN 98-175. Asherwood Environmental Science Center, Wabash, IN.

Tanner, C.C., J.S. Clayton, and R.D.S. Wells. 1993. Effects of suspended solids on the establishment and growth of *Egeria densa*. Aquatic Botany. 45:299-310.

Tappel, P.D., and T.C. Bjornn. 1983. A new method of relating size of spawning gravel to salmonid embryo survival. North American Journal of Fisheries Management. 3:123-133.

Terrados, J., C. M. Duarte, M. D.Fortes, J.Borum, N. S. R.Agawin, S.Bach, U.Thampanya, L.Kamp-Nielsen, W. J.Kenworthy, O.Geertz-Hansen, and J. Vermaat. 1998. Changes in community structure and biomass of seagrass communities along gradients of siltation in SE Asia. Estuarine and Coastal Shelf Science. 46: 757-768.

Tester, P.A. and J.T Turner. 1988. Zooplankton feeding ecology: Feeding rates of the copepods *Acartia tonsa*, *Centropages velificatus*, and *Eucalanus pileatus* in relation to the suspended sediments in the plume of the Mississippi River. Topics in Marine Biology. J.D. Ross (ed) Scient. Mar. (2-3): 231-237.

Thirb, H.H. and K.Benson-Evans. 1985. The effect of suspended solids on the growth of apical tips of gametophyte plants of Lemanea and on carpospore germination and subsequent colonization. Archives of Hydrobiology. 103:409-417.

Torres, J. 2001. Impacts of sedimentation on the growth rates of *Montastrea Annularis* in Southwest Puerto Rico. Bulletin of Marine Science. 69: 631-637.

Turner, E.J., and D.C. Miller. 1991. Behavior and growth of *Mercenaria mercenaria* during simulated storm events. Marine Biology 111:55-64

U.S.EPA. 1999. Protocol for developing sediment TMDLs. EPA-B-99-004. Office of Water. October, 1999.

U.S.EPA. 2000. The Quality of Our Nations Waters. A Summary of the National Water Quality Inventory: 1998 Report to Congress. Office of Water, Washington DC. 841-S-00-001.

U.S.EPA. 2002a. Aquatic Stressors, Framework and Implementation Plan for Effects Research. NHEERL, ORD, U.S.EPA, Research Triangle Park, NC. September 20, 2002.

U.S.EPA 2002b. Strategic Planning and Research Coordination (SPRC) action plan for aquatic ecology and water quality. Office of Research and Development/ Office of Water. October 31, 2002.

U.S.EPA 2002c. Aquatic Stressors: Framework and implementation plan for effects research. National Health and Environmental Effects Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Research Triangle Park, NC 27711. September 20, 2002.

U.S.EPA. 2003a. National Water Quality Report to Congress (305(b) report). http://www.epa.gov/OWOW/305b/

U.S.EPA. 2003b. Ambient water quality criteria for dissolved oxygen, water clarity and Chlorophyll A for Chesapeake Bay and tidal tributaries. U.S.EPA, Chesapeake Bay Program Office, Annapolis MD. April, 2003.

Van Eerden M. R and B. Voslamber. 1995. Mass fishing by Cormorants *Phalacrocorax carbo sinensis* at lake Ijsselmeer, The Netherlands: A recent and successful adaptation to a turbid environment. Ardea. 83:199-212.

Wakeman, T., R. Peddicord, and J.Sustar. 1975. Effects of suspended solids associated with dredging operations on estuarine organisms. Pp. 431-436 in D.M. Bolle, editor. Proceedings of Oceans 75. San Diego CA.

Waters, T.F. 1995. Sediment in streams- Sources, biological effects and control. American Fisheries Society Monograph 7. American Fisheries Society, Bethesda, MD.

Wilber, D.H. and D.G. Clarke. 2001. Biological effects of suspended sediments: a review of suspended sediment impacts on fish and shellfish with relation to dredging activities in estuaries. North American Journal of Fisheries Management. l21:855-875.

Wilber, D.H., D.G. Clarke, and W. Brostoff. In review. Sedimentation: Potential biological effects from dredging operations in estuarine and marine environments. Draft Technical Note E-x. U.S. Army Engineer Research and Development Center, Vicksburg, MS.

Wilson, K.P., J.P. Shannon, and D.W. Blinn. 1999. Effects of suspended sediment on the biomass and cell morphology of *Cladophora glomerta (*Chlorophyta) in the Colorado River, Arizona. Journal of Phycology. 35:35-41.

Yount, J.D. and G.J. Niemi. 1990. Recovery of lotic communities and ecosystems from disturbance - A narrative review of case studies. Environmental Management. 14:547-569.

Zweig, L.D. and C. F. Rabeni. 2001. Biomonitoring for deposited sediment using benthic invertebrates: a test on 4 Missouri streams. Journal of the North American Benthological Society. 20:643-657.

List of Tables

List of Figures

Table 1

RANKING OF AQUATIC STRESSORS: SUSPENDED SOLIDS & BEDDED SEDIMENTS

[Comparisons of 305(b) National Water Quality Inventory Reports (U.S.EPA, 2003a)]

RIVERS & STREAMS:

1994 (7 P/S* Listed)	1996 (8 P/S Listed)	1998 (8 P/S Listed)	2000 (8 P/S Listed)
Siltation (2)**	Siltation (1)	Siltation (1)	Siltation (2)
Suspended Solids (7)	Suspended Solids (7)		

NOTES: *P/S = Pollutants/Stressors
** As an example, Siltation was ranked No. 2 out of the seven (7) Pollutants/Stressors found on the Table for Rivers & Streams in the 1994 Report.

LAKES, PONDS, & RESERVOIRS:

1994 (7 P/S Listed)	1996 (7 P/S Listed)	1998 (7 P/S Listed)	2000 (7 P/S Listed)
Siltation (2)	Siltation (3)	Siltation (3)	Siltation (3)
Suspended Solids (5)	Suspended Solids (6)	Suspended Solids (5)	

WETLANDS:

1994 (9 P/S Listed)	1996 (8 P/S Listed)	1998 (7 P/S Listed)	2000 (6 P/S Listed)
Sediment (1)	Sedimentation & Siltation (1)	Sedimentation & Siltation (1)	Sedimentation & Siltation (1)

ESTUARIES:

1994 (7 P/S Listed)	1996 (7 P/S Listed)	1998 (7 P/S Listed)	2000 (7 P/S Listed)

Note that Siltation, Suspended Solids, Sediment, and Turbidity were not found on these Lists for 1994, 1996, 1998, and the 2000 305(b) Reports.

OCEAN SHORELINE WATERS:

1994 (7 P/S Listed)	1996 (8 P/S Listed)	1998 (7 P/S Listed)	2000 (7 P/S Listed)
Turbidity (4)	Turbidity (2)	Turbidity (2)	Turbidity (3)
Siltation (5)	Suspended Solids (5)	Suspended Solids (4)	Suspended Solids (4)
		Siltation (5)	

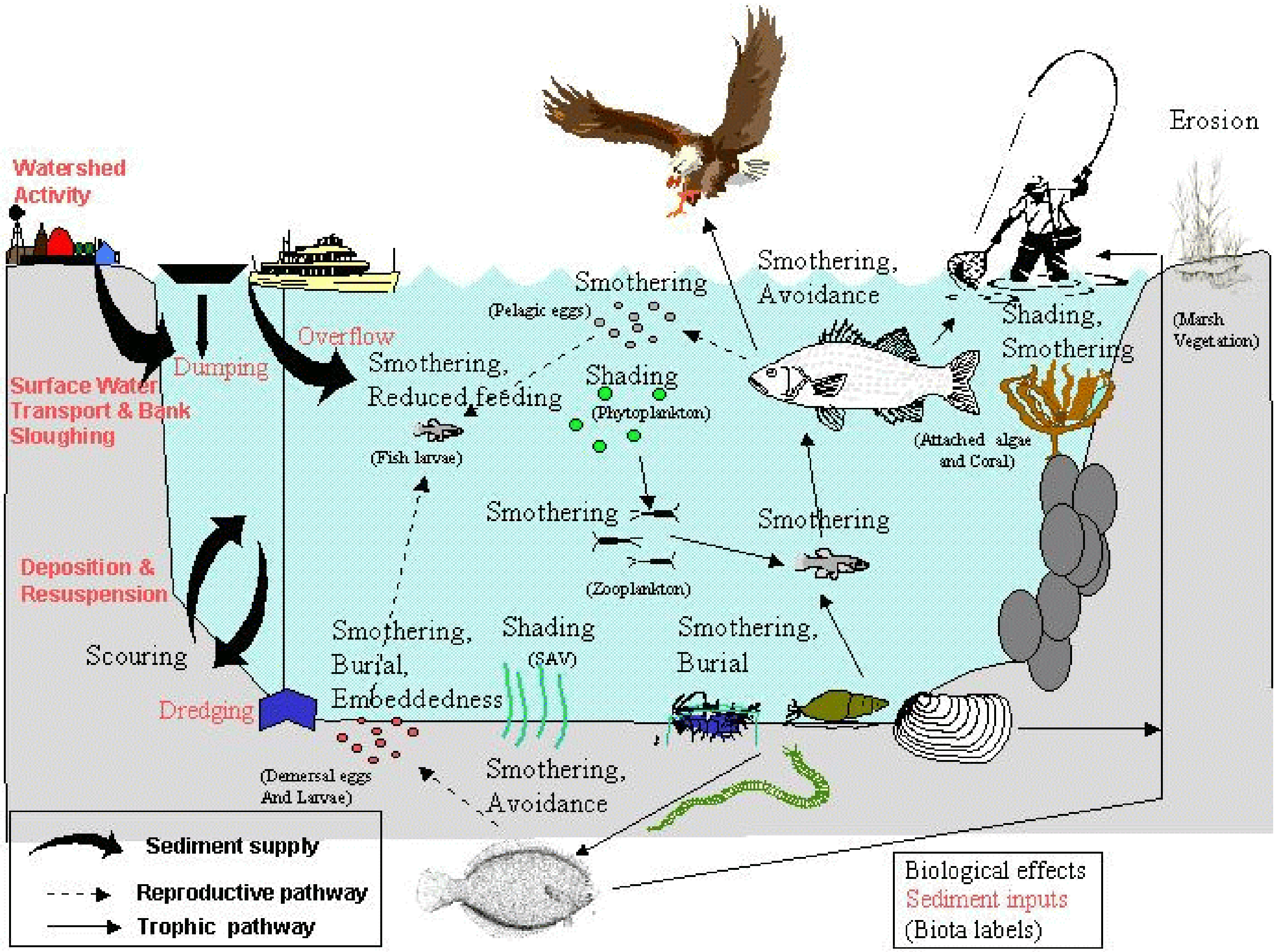

Figure 1: Conceptual model of biological effects of suspended and bedded sediments in estuaries.

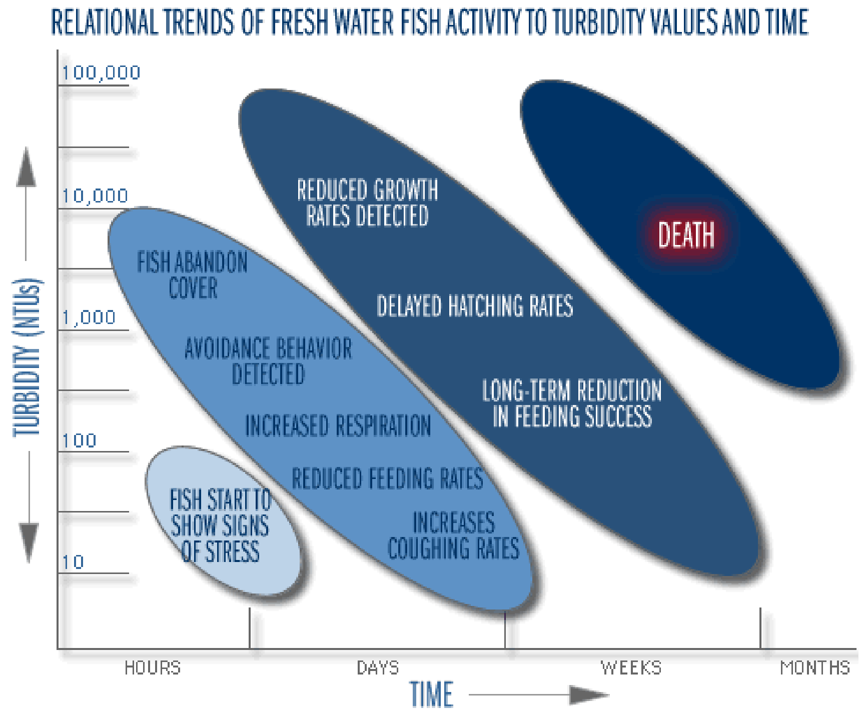

Figure 2. Idealized model of fish response to increased suspended sediments. Schematic source of above figure is unknown; it is a generic, un-calibrated impact assessment model based on Newcombe, C. P., and J. O. T. Jensen. 1996. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management. 16: 693-727. Reprinted, with permission, from: http://wow.nrri.umn.edu/wow/under/parameters/turbidity.html.

Appendix A

“Channel Suspended Sediment and Fisheries: A Concise Guide to Impacts”

By

Charles P. Newcombe
Ministry of Environment, Land, and Parks
Victoria, British Columbia

Not included, available upon request

Appendix B

Model of the effects of turbidity on fishes

(C.P. Newcombe, Personal Communication)

Reprinted with permission of the author

This is similar to the model in Newcombe (2003)

Not included, available upon request

Appendix C

Available data on the effects of suspended sediments on biota. Data take from the original literature (unless otherwise noted) or Newcombe and Jensen (1996: “N&J”)

Key:

Life Stage: A = Adult, J = Juvenile, L = Larval
Concentration: Material is listed if known: k = kaolin, ns = natural sediment
Source: Original data consulted unless otherwise noted. N&J = Newcombe and Jensen, 1996.
Duration: Duration is in hours unless otherwise noted. d = days. f = field studies.

SPECIES	Life Stage	Concentration in mg/l	Duration in Hours	EFFECT (Response)	REFERENCE	Source
MOLLUSCA						
Eastern oyster *Crassostrea virginica*	L	400	12 d	10% mortality	Davis and Hidu 1969	
“ ”	L	500	12 d	18% mortality	“ ”	
	L	750	12 d	reduced growth		
	L	750	12 d	30% mortality		
	L	1000	12 d	40% mortality		
	L	1500	12 d	58 % mortality		
	L	2000	12 d	75% mortality		
	L	3000	12 d	99 % mortality		
Pacific Oyster *Cassostrea gigas*	L	$1200	2 d	abnormal shell development	Cardwell et. al. 1976	
	L	$800	2 d	50% mortality		
Hard Clam *Mercenaria mercenaria*	L	$750	10 d	10% mortality	Davis and Hidu 1969	
	L	3000	10 d	15% mortality		
	L	4000	11 d	30% mortality		
Eastern Oyster *Crassostrea virginica*	A	$1000	2 d	reduced pumping	Loosanoff, 1962	
Soft Shell Clam *Mya arenaria*	A	100	35 d	reduced growth	Grant and Murphy 1985	
Hard Clam *Mercenaria mercenaria*	A	27	14 d	reduced growth	Murphy, 1985	
“ ”	A	100	2 d	reduced growth	Turner and Miller, 1991	
“ ”	J	44	21 d	reduced growth	Bricelj et al.,1984	
Coast Mussels *Mytilus californiamus*	A	8100	17 d	10% mortality	Peddicord, 1980	
“ ”	J	15500	16 d	20-14% mortality	“ ”	
“ ”	A	80000	11 d	50% mortality	“ ”	
“ ”	A	85000	9 d	50% mortality	“ ”	
Blue Mussel *Mytilus edulis*	A	15000	8 d	0-20% mortality	Peddicord, 1976	

" "	J	100000	5 d	10% mortality	McFarland and Peddicors, 1980	
	A	60000	10 d	10% mortality	Wakeman et al., 1975	
Surf Clam *Spisula solidissima*	A	500	21 d	reduced growth	Robinson et al.,1984	
Bay Scallop *Argopecten irradians*	A	500	7 d	increased respiration	Morre, 1978	
" "	A	1000	7 d	increased respiration	" " "	
CRUSTACEA						
Sand Shrimp *Crangon nirgomaculata*		16000	8 d	10%mortality	Mc Farland and Peddicord 1980	
" "		50000	8 d	50% mortality	" "	
Grass Shrimp *Paleomon macrodactylus*		24000 (k)	10 d	10% mortality	" "	
" "		77000 (k)	8 d	20% mortality	" "	
Dungeness Crab *Cancer magister*		9200 (ns)	8 d	5% mortality	Peddicord and McFarland, 1976	
" "		11700 (ns)	7 d	20% mortality	" "	
" " juvenile	J	15900 (ns)	9 d	15% mortality	" "	
" " "	J	18900 (ns)	4 d	20% mortality	" "	
" " adult	A	10000 (k)	8 d	10% mortality	McFarland and Peddicord, 1980	
" " "	A	32000 (k)	8 d	50% mortality	" "	
Kuruma Prawn *Penaeus japonicus*	J	180 (ns)	21 d	10% mortality	Lin et al., 1992	
" "	J	370 (ns)	21 d	32% mortality	" "	
Black-tailed Sand Shrimp *Cragnon nigrocauda*		11900 (ns)	5 d	10% mortality	Peddicord, 1990	
" "		4300 (ns)	3 d	5% mortality	" "	
" "		9000 (b)	10 d	10% mortality	Wakeman et al. 1975	
Mysid Shrimp *Mysidopsis bahia*		230 (ns)	28 d	40 % mortality	Nimmo et al. 1982	

" "		1020 (ns)	28 d	60-80% mortality	" "	
Copepod *Eurytmora affinis*		>350 (ns)	f	reduced population growth	Sellner and Bundy, 1986	
Copepod		>100 (ns)	f	reduced vertical migration	Daborn and Brylinsky, 1981	
Copepod *Acartia tonsa*		>95 (ns)	f	reduced feeding	Tester and Turner, 1988	
Copepod *A.tonsa, E. affinis*		>250	f	reduced feeding	Sherk et al., 1976	
Daphnids		50-100 (ns)	<18 d	reduced feeding	Arruda et al., 1983	
Benthic Algae		2.0-4.2	f	decrease in biomass, growth	Wilson et al., 1999	
Freshwater Mussels		600-750 (ns)	f	decreased filter clearance	Aldridge et al., 1987	
Red Algae *Lemanea*		5000 (ns)	21 d	reduced primary production	Thirb and Benson-Evans 1985	
" " *Egeria*		30-40 (ns)	40 d	reduced growth	Tanner et al. 1993	
Oyster *Crassostrea virginica*		100	f	reduced pumping	Sherk et al. 1975	
FISH						
Adult salmonids and rainbow smelt						
Grayling (Arctic)	A	100	0.10	Fish avoided turbid water	Suchanek et al. (1984a, 1984b)	N & J
	A	100	1,008	Fish had decreased resistance to environmental stress	McLeay et al. (1984)	N & J
	A	100	1,008	Impaired feeding		N & J
Salmon	A	25	4	Feeding activity reduced	Phillips (1970)	N & J
	A	16.5	24	Feeding behavior apparently educed	Townsend (1983); Ott (1984)	N & J
	A	1,650	240	Loss of habitat caused by excessive sediment transport	Coats et al. (1985)	N & J
Salmon (Atlantic)	A	2,500	24	Increased risk of predation	Gibson (1933)	N & J

Salmon (chinook)	A	650	168	No histological signs of damage to olfactory epithelium	Brannon et al. (1981)	N & J
Salmon (chinook)	A	350	0.17	Home water preference disrupted	Whitman et al. (1982)	N & J
Salmon (chinook)	A	650	168	Homing behavior normal, but fewer test fish returned	Whitman et al. (1982)	N & J
Salmon (chinook)	A	39,300	24	No mortality (VA, <5-100 um; median, <15 um)	Newcomb and Flagg (1983)	N & J
Salmon (chinook)	A	82,400	6	Mortality rate 60% (VA, <5-100 um)	Newcomb and Flagg (1983)	N & J
Salmon (chinook)	A	207,000	1	Mortality rate 100% (VA, <5-100 um)	Newcomb and Flagg (1983)	N & J
Salmon (Pacific)	A	525	588	No mortality (other end points not investigated)	Griffin (1938)	N & J
Salmon (sockeye)	A	500	96	Plasma glucose levels increased 39%	Servizi and Martens (1987)	N & J
Salmon (sockeye)	A	1,500	96	Plasma glucose levels increased 150%	Servizi and Martens (1987)	N & J
Salmon (sockeye)	A	39,300	24	No mortality (VA, <5-100 um; median, <15 um)	Newcomb and Flagg (1983)	N & J
Salmon (sockeye)	A	82,400	6	Mortality rate 60% (VA, <5-100 um; median, <15 um)	Newcomb and Flagg (1983)	N & J
Salmon (sockeye)	A	207,000	1	Mortality rate 100% (VA)	Newcomb and Flagg (1983)	N & J
Smelt (rainbow)	A	3.5	168	Increased vulnerability to predation	Swenson (1978)	N & J
Steelhead	A	500	3	Signs of sublethal stress (VA)	Redding and Schreck (1982)	N & J
Steelhead	A	1,650	240	Loss of habitat caused by excessive sediment transport	Coats et al. (1985)	N & J
Steelhead	A	500	9	Blood cell count and blood chemistry change	Redding and Schreck (1982)	N & J
Trout	A	16.5	24	Feeding behavior apparently reduced	Townsend (1983); Ott (1984)	N & J
Trout	A	75	168	Reduced quality of rearing habitat	Slaney et al. (1977b)	N & J

Trout	A	270	312	Gill tissue damaged	Herbert and Merkens (1961)	N & J
Trout	A	525	588	No mortality (other end points not investigated)	Griffin (1938)	N & J
Trout	A	300	720	Decrease in population size	Peters (1967)	N & J
Trout (brook)	A	4.5	168	Fish more active and less dependent on cover	Gradall and Swenson (1982)	N & J
Trout (brown)	A	1,040	17,520	Gill lamellae thickened (VFSS)	Herbert et al. (1961)	N & J
Trout (brown)	A	1,210	17,520	Some gill lamellae became fused (VFSS)	Herbert et al. (1961)	N& J
Trout (brown)	A	18	720	Abundance reduced	Peters (1967)	N & J
Trout (brown)	A	100	720	Population reduced	Scullion and Edwards (1980)	N & J
Trout (brown)	A	1,040	8.760	Population one-seventh of expected size (River Fal)	Herbert et al. (1961)	N & J
Trout (brown)	A	5,838	8,760	Fish numbers one-seventh of expected size (River Par)	Herbert et al. (1961)	N & J
Trout (cutthroat)	A	35	2	Feeding ceased; fish sought cover	Cordone and Kelly (1961)	N & J
Trout (lake)	A	3.5	168	Fish avoided turbid areas	Swenson (1978)	N & J
Trout (rainbow)	A	66	1	Avoidance behavior manifested part of the time	Lawrence and Scherer (1974)	N & J
Trout (rainbow)	A	665	1	Fish attracted to turbidity	Lawrence and Scherer (1974)	N & J
Trout (rainbow)	A	100	0.10	Fish avoided turbid water (avoidance behavior)	Suchanek et al. (1984a, 1984b)	N & J
Trout (rainbow)	A	100	0.25	Rate of coughing increased (FSS)	Hughes (1975)	N & J
Trout (rainbow)	A	250	0.25	Rate of coughing increased (FSS)	Hughes (1975)	N & J
Trout (rainbow)	A	810	504	Gills of fish that survived had thickened epithelium	Herbert and Merkens (1961)	N & J

Trout (rainbow)	A	17,500	168	Fish survived: gill epithelium proliferated and thickened	Slanina (1962)	N & J
Trout (rainbow)	A	50	960	Rate of weight gain reduced (CWS)	Herbert and Richards (1963)	N & J
Trout (rainbow)	A	50	960	Rate of weight gain reduced (WF)	Herbert and Richards (1963)	N & J
Trout (rainbow)	A	810	504	Some fish died	Herbert and Merkens (1961)	N & J
Trout (rainbow)	A	270	3,240	Survival rate reduced	Herbert and Merkens (1961)	N & J
Trout (rainbow)	A	200	24	Test fish began to die on the first day (WF)	Herbert and Richards (1963)	N & J
Trout (rainbow)	A	80,000	24	No mortality	D. Herbert, personal communicatio n to Alabaster and Lloyd (1980)	N & J
Trout (rainbow)	A	18	720	Abundance reduced	Peters (1967)	N & J
Trout (rainbow)	A	59	2,232	Habitat damage; reduced porosity of gravel	Slaney et al. (1977b)	N & J
Trout (rainbow)	A	4,250	588	Mortality rate 50% (CS)	Herbert and Wakeford (1962)	N & J
Trout (rainbow)	A	49,838	96	Mortality rate 50% (DM)	Lawrence and Scherer (1974)	N & J
Trout (rainbow)	A	3,500	1,488	Catastrophic reduction in population size	Herbert and Merkens (1961)	N & J
Trout (rainbow)	A	160,000	24	Mortality rate 100%	D. Herbert, personal communicatio n to Alabaster and Lloyd (1980)	N & J
Trout (sea)	A	210	24	Fish abandoned traditional spawning habitat	Hamilton (1961)	N & J
Whitefish (lake)	A	0.66	1	Swimming behavior changed	Lawrence and Scherer (1974)	N & J

Whitefish (lake)	A	16,613	96	Mortality rate 50% (DM)	Lawrence and Scherer (1974)	N & J
Whitefish (mountain)	A	10,000	24	Fish died; silt-clogged gills	Langer (1980)	N & J
JUVENILE SALMONIDS						
Grayling (Arctic)	U	20	24	Fish avoided parts of the stream	Birtwell et al. (1984)	N & J
Grayling (Arctic)	U	10,000	96	Fish swam near the surface	McLeay et al. (1987)	N & J
Grayling (Arctic)	J	86	0.42	78% of fish avoided turbid water (NTU, <20)	Scannell (1988)	N & J
Grayling (Arctic)	U	100	1	Catch rate reduced (unfamiliar prey: drosophila)	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	100	1	Catch rate reduced (unfamiliar prey: tubificids)	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	300	1	Catch rate reduced (unfamiliar prey: drosophila)	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	1,000	1	Feeding rate reduced (unfamiliar prey: tubificids)	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	1,000	1	Feeding rate reduced (unfamiliar prey: drosophila)	McLeay et al. (1987)	N & J
Grayling (Arctic)	YY	3,810	144	Food intake severely limited	Simmons (1982)	N & J
Grayling (Arctic)	U	100	12	Reduced ability to tolerate high temperatures	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	100	756	Fish moved out of the test	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	1,000	1,008	Fish had frequent misstrikes while feeding	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	1,000	1,008	Fish responded very slowly to prey	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	300	1,008	Rate of feeding reduced	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	1,000	840	Rate of feeding reduced	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	1,000	1,008	Fish failed to consume all prey	McLeay et al. (1987)	N & J

Grayling (Arctic)	U	300	840	Serious impairment of feeding	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	300	1,008	Respiration rate increased (FSS)	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	300	1,008	Fish less tolerant of pentachlorophenol	McLeay et al. (1987)	N & J
Grayling (Arctic)	YY	3,810	144	Mucus and sediment accumulated in the gill lamellae	Simmons (1982)	N & J
Grayling (Arctic)	YY	3,810	144	Fish displayed many signs of poor condition	Simmons (1982)	N & J
Grayling (Arctic)	YY	1,250	48	Moderate damage to gill tissue	Simmons (1982)	N & J
Grayling (Arctic)	YY	1,388	96	Hyperplasia and hypertrophy of gill tissue	Simmons (1982)	N & J
Grayling (Arctic)	U	100	1,008	Growth rate reduced	McLeay et al. (1984)	N & J
Grayling (Arctic)	U	100	840	Fish responded less rapidly to drifting food	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	300	1,008	Weight gain reduced	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	1,000	1,008	Weight gained reduced by 33%	McLeay et al. (1987)	N & J
Grayling (Arctic)	U	300	756	Fish displaced from their habitat	McLeay et al. (1987))	N & J
Grayling (Arctic)	U	100,000	168	No changes in gill histology (not an end point)	McLeay et al. (1983)	N & J
Salmon (chinook)	S	943	72	Tolerance to stress reduced (VA)	Stober et al. (1981)	N & J
Salmon (chinook)	J	6	1,440	Growth rate reduced (LNFH)	MacKinley et al. (1987)	N & J
Salmon (chinook)	J	1,400	36	Mortality rate 50%	Newcomb and Flagg (1983)	N & J
Salmon (chinook)	J	9,400	36	Mortality rate 50%	Newcomb and Flagg (1983)	N & J
Salmon (chinook)	S	488	96	Mortality rate 50%	Stober et al. (1981)	N & J
Salmon (chinook)	S	11,000	96	Mortality rate 50%	Stober et al. (1981)	N & J
Salmon (chinook)	S	19,364	96	Mortality rate 50%	Stober et al. (1981)	N & J
Salmon (chinook)	J	39,400	36	Mortality rate 90% (VA)	Newcomb and Flagg (1983)	N & J
Salmon (chum)	J	28,000	96	Mortality rate 50%	Smith (1940)	N & J

Salmon (chum)	J	55,000	96	Mortality rate 50% (winter)	Smith (1940)	N & J
Salmon (coho)	J	53.5	0.02	Alarm reaction	Berg (1983)	N & J
Salmon (coho)	J	88	0.02	Alarm reaction	Bisson and Bilby (1982)	N & J
Salmon (coho)	U	20	0.05	Cough frequency not increased	Servizi and Martens (1992)	N & J
Salmon (coho)	J	53.5	12	Changes in territorial behavior	Berg and Northcote (1985)	N & J
Salmon (coho)	J	88	0.08	Avoidance behavior	Bisson and Bilby (1982)	N & J
Salmon (coho)	J	6,000	1	Avoidance behavior	Noggle (1978)	N & J
Salmon (coho)	U	300	0.17	Avoidance behavior within minutes	Servizi and Martens (1992)	N & J
Salmon (coho)	J	25	1	Feeding rate decreased	Noggle (1978)	N & J
Salmon (coho)	J	100	1	Feeding rate decreased to 55% of maximum	Noggle (1978)	N & J
Salmon (coho)	J	250	1	Feeding rate decreased to 10% of maximum	Noggle (1978)	N & J
Salmon (coho)	J	300	1	Feeding ceased	Noggle (1978)	N & J
Salmon (coho)	U	2,460	0.05	Coughing behavior manifest within minutes	Servizi and Martens (1992)	N & J
Salmon (coho)	J	53.5	12	Increased physiological stress	Berg and Northcote (1985)	N & J
Salmon (coho)	U	2,460	1	Cough frequency greatly increased	Servizi and Martens (1992)	N & J
Salmon (coho)	U	240	24	Cough frequency increased more than 5-fold	Servizi and Martens (1992)	N & J
Salmon (coho)	U	530	96	Blood glucose levels increased	Servizi and Martens (1992)	N & J
Salmon (coho)	J	1,547	96	Gill damage	Noggle (1978)	N & J
Salmon (coho)	U	2,460	24	Fatigue of the cough reflex	Servizi and Martens (1992)	N & J

Salmon (coho)	U	3,000	48	High level sublethal stress; avoidance	Servizi and Martens (1992)	N & J
Salmon (coho)	J	102	336	Growth rate reduced (FC, BC)	Sigler et al. (1984)	N & J
Salmon (coho)	U	8,000	96	Mortality rate 1%	Servizi and Martens (1991)	N & J
Salmon (coho)	J	1,200	96	Mortality rate 50%	Noggle (1978)	N & J
Salmon (coho)	J	35,000	96	Mortality rate 50%	Noggle (1978)	N & J
Salmon (coho)	U	22,700	96	Mortality rate 50%	Servizi and Martens (1991)	N & J
Salmon (coho)	F*	8,100	96	Mortality rate 50%	Servizi and Martens (1991)	N & J
Salmon (coho)	PS	18,672	96	Mortality rate 50%	Stober et al. (1981)	N & J
Salmon (coho)	S	509	96	Mortality rate 50%	Stober et al. (1981)	N & J
Salmon (coho)	S	1,217	96	Mortality rate 50% (VA)	Stober et al. (1981)	N & J
Salmon (coho)	S	28,184	96	Mortality rate 50% (VA)	Stober et al. (1981)	N & J
Salmon (coho)	S	29,580	96	Mortality rate 50%	Stober et al. (1981)	N & J
Salmon (sockeye)	S	1,261	96	Body moisture content reduced	Servizi and Martens (1987)	N & J
Salmon (sockeye)	S	7,447	96	Plasma chloride levels increased slightly	Servizi and Martens (1987)	N & J
Salmon (sockeye)	U	1,465	96	Hypertrophy and necrosis of gill tissue (CSS)	Servizi and Martens (1987)	N & J
Salmon (sockeye)	U	3,143	96	Hypertrophy and necrosis of gill tissue (FSS)	Servizi and Martens (1987)	N & J
Salmon (sockeye)	U	9,851	96	Hypertrophy and necrosis of gill tissue (MCSS)	Servizi and Martens (1987)	N & J
Salmon (sockeye)	U	17,560	96	Hypertrophy and necrosis of gill tissue (FSS)	Servizi and Martens (1987)	N & J

Salmon (sockeye)	U	23,790	96	Hypertrophy and necrosis of gill tissue (FSS)	Servizi and Martens (1987)	N & J
Salmon (sockeye)	U	2,688	96	Hypertrophy and necrosis of gill tissue (MCSS)	Servizi and Martens (1987)	N & J
Salmon (sockeye)	U	2,100	96	No fish died (MFSS)	Servizi and Martens (1987)	N & J
Salmon (sockeye)	U	9,000	96	No mortality	Servizi and Martens (1987)	N & J
Salmon (sockeye)	U	13,900	96	Mortality rate 10% (FSS)	Servizi and Martens (1987)	N & J
Salmon (sockeye)	U	9,850	96	Gill hyperplasia, hypertrophy, separation, necrosis (MFSS)	Servizi and Martens (1987)	N & J
Salmon (sockeye)	J	1,400	36	Mortality rate 50%	Newcomb and Flagg (1983)	N & J
Salmon (sockeye)	J	9,400	36	Mortality rate 50%	Newcomb and Flagg (1983)	N & J
Salmon (sockeye)	U	1,700	96	Mortality rate 50% (CSS)	Servizi and Martens (1987)	N & J
Salmon (sockeye)	U	4,850	96	Mortality rate 50% (MCSS)	Servizi and Martens (1987)	N & J
Salmon (sockeye)	U	8,200	96	Mortality rate 50% (MFSS)	Servizi and Martens (1987)	N & J
Salmon (sockeye)	U	17,560	96	Mortality rate 50% (FSS)	Servizi and Martens (1987)	N & J
Salmon (sockeye)	J	39,400	36	Mortality rate 90% (VA)	Newcomb and Flagg (1983)	N & J
Salmon (sockeye)	U	13,000	96	Mortality rate 90% (MFSS)	Servizi and Martens (1987)	N & J
Salmon (sockeye)	U	23,900	96	Mortality rate 90% (FSS)	Servizi and Martens (1987)	N & J
Steelhead	J	102	336	Growth rate reduced (FC, BC)	Sigler et al. (1984)	N & J
Trout (brook)	FF	12	5,880	Growth rates declined	Sykora et al. (1972)	N & J
Trout (brook)	FF	24	5,208	Growth rate reduced (LNFH)	Sykora et al. (1972)	N & J

Trout (brook)	FF*	100	1,176	Test fish weighed 16% of controls (LNFH)	Sykora et al. (1972)	N & J
Trout (brook)	FF	50	1,848	Growth rates declined (LNFH)	Sykora et al. (1972)	N & J
Trout (rainbow)	FF	1,750	480	Mortality rate 57% (controls 5%)	Campbell (1954)	N & J
Trout (rainbow)	J	4,887	384	Hyperplasia of gill tissue	Goldes (1983)	N & J
Trout (rainbow)	J	4,887	384	Parasitic infection of gill tissue	Goldes (1983)	N & J
Trout (rainbow)	J	171	96	Particles penetrated cells of branchial epithelium	Goldes (1983)	N & J
Trout (rainbow)	Y	90	456	Mortality rates 0-20% (DE)	Herbert and Merkens (1961)	N & J
Trout (rainbow)	Y	90	456	Mortality rates 0-15% (KC)	Herbert and Merkens (1961)	N & J
Trout (rainbow)	Y	270	456	Mortality rates 10-35% (KC)	Herbert and Merkens (1961)	N & J
Trout (rainbow)	Y	810	456	Mortality rates 35-85% (DE)	Herbert and Merkens (1961)	N & J
Trout (rainbow)	Y	810	456	Mortality rates 5-80% (KC)	Herbert and Merkens (1961)	N & J
Trout (rainbow)	Y	270	456	Mortality rates 25-80% (DE)	Herbert and Merkens (1961)	N & J
Trout (rainbow)	Y	7,433	672	Mortality rate 40% (CS)	Herbert and Wakeford (1962)	N & J
Trout (rainbow)	Y	4,250	672	Mortality rate 50%	Herbert and Wakeford (1962)	N & J
Trout (rainbow)	Y	2,120	672	Mortality rate 100%	Herbert and Wakeford (1962)	N & J
Trout (rainbow)	J	4,315	57	Mortality rate ~ 100% (CSS)	Newcombe et al. (1995)	N & J
SALMONID EGGS AND LARVAE						

Grayling (Arctic)	SF	25	24	Mortality rate 5.7%	J. LaPerriere (personal com- munication)	N & J
Grayling (Arctic)	SF	22.5	48	Mortality rate 14.0%	J. LaPerriere (personal com- munication)	N & J
Grayling (Arctic)	SF	65	24	Mortality rate 15.0%	J. LaPerriere (personal com- munication)	N & J
Grayling (Arctic)	SF	21.7	72	Mortality rate 14.7%	J. LaPerriere (personal com- munication)	N & J
Grayling (Arctic)	SF	20	96	Mortality rate 13.4%	J. LaPerriere (personal com- munication)	N & J
Grayling (Arctic)	SF	142.5	48	Mortality rate 26%	J. LaPerriere (personal com- munication)	N & J
Grayling (Arctic)	SF	185	72	Mortality rate 41.3%	J. LaPerriere (personal com- munication)	N & J
Grayling (Arctic)	SF	230	96	Mortality rate of 47%	J. LaPerriere (personal com- munication)	N & J
Salmon	E	117	960	Mortality; deterioration of spawning gravel	Cederholm et al. (1981)	N & J
Salmon (chum)	E	97	2,808	Mortality rate 77% (controls, 6%)	Langer (1980)	N & J
Salmon (coho)	E	157	1,728	Mortality rate 100% (controls, 16.2%)	Shaw and Maga (1943)	N & J
Steelhead	E	37	1,488	Hatching success 42% (controls, 63%)	Slaney et al. (1977b)	N & J
Trout	E	117	960	Mortality; deterioration of spawning gravel	Cederholm et al. (1981)	N & J
Trout (rainbow)	EE	1,750	144	Mortality rate greater than controls (controls, 6%)	Campbell (1954)	N & J
Trout (rainbow)	E	6.6	1,152	Mortality rate 40%	Slaney et al. (1977b)	N & J
Trout (rainbow)	E	57	1,488	Mortality rate 47% (controls, 32%)	Slaney et al. (1977b)	N & J

Trout (rainbow)	E	120	384	Mortality rates 60-70% (controls, 38.6%)	Erman and Lignon (1988)	N & J
Trout (rainbow)	E	20.8	1,152	Mortality rate 72%	Slaney et al. (1977a)	N & J
Trout (rainbow)	E	46.6	1,152	Mortality rate 100%	Slaney et al. (1977b)	N & J
Trout (rainbow)	E	101	1,440	Mortality rate 98% (controls, 14.6%)	Turnpenny and Williams (1980)	N & J
NONSALMONID EGGS AND LARVAE						
Bass (striped)	L	200	0.42	Feeding rate reduced 40%	Breitburg (1988)	N & J
Bass (striped)	E	800	24	Development rate slowed significantly	Morgan et al. (1983)	N & J
Bass (striped)	E	100	24	Hatching delayed	Schubel and Wang (1973)	N & J
Bass (striped)	E	1,000	168	Reduced hatching success	Auld and Schubel (1978)	N & J
Bass (striped)	L	1,000	68	Mortality rate 35% (controls, 16%)	Auld and Schubel (1978)	N & J
Bass (striped)	L	500	72	Mortality rate 42% (controls, 17%)	Auld and Schubel (1978)	N & J
Bass (striped)	L	485	24	Mortality rate 50%	Morgan et al. (1973)	N & J
Herring	L	10	3	Depth preference changed	Johnson and Wildish (1982)	N & J
Herring (lake)	L	16	24	Depth preference changed	Swenson and Matson (1976)	N & J
Herring (Pacific)	L	2,000	2	Feeding rate reduced	Boehlert and Morgan (1985)	N & J
Herring (Pacific)	L	1,000	24	Mechanical damage to epidermis	Boehlert (1984)	N & J
Herring (Pacific)	L	4,000	24	Epidermis punctured; microridges less distinct	Boehlert (1984)	N & J
Perch (white)	E	800	24	Egg development slowed significantly	Morgan et al. (1983)	N & J
Perch (white)	E	100	24	Hatching delayed	Schubel and Wang (1973)	N & J

Perch (white)	E	1,000	168	Reduced hatching success	Auld and Schubel (1978)	N & J
Perch (white)	L	155	48	Mortality rate 50%	Morgan et al. (1973)	N & J
Perch (white)	L	373	24	Mortality rate 50%	Morgan et al. (1973)	N & J
Perch (white)	L	280	48	Mortality rate 50%	Morgan et al. (1973)	N & J
Perch (yellow)	L	500	96	Mortality rate 37% (controls, 7%)	Auld and Schubel (1978)	N & J
Perch (yellow)	L	1,000	96	Mortality rate 38% (controls, 7%)	Auld and Schubel (1978)	N & J
Shad (American)	L	100	96	Mortality rate 18% (controls, 5%)	Auld and Schubel (1978)	N & J
Shad (American)	L	500	96	Mortality rate 36% (controls, 4%)	Auld and Schubel (1978)	N & J
Shad (American)	L	1,000	96	Mortality rate 34% (controls, 5%)	Auld and Schubel (1978)	N & J
ADULT NONSALMONIDS						
Anchovy (bay)	A	231	24	Mortality rate 10% (FE)	Sherk et al. (1975)	N & J
Anchovy (bay)	A	471	24	Mortality rate 50% (FE)	Sherk et al. (1975)	N & J
Anchovy (bay)	A	960	24	Mortality rate 90%	Sherk et al. (1975)	N & J
Bass (striped	A	1,500	336	Haematocrit increased (FE)	Sherk et al. (1975)	N & J
Bass (striped)	A	1,500	336	Plasma osmolality increased (FE)	Sherk et al. (1975)	N & J
Cunner	A	28,000	24	Mortality rate 50% (20.0-25.0° C)	Rogers (1969)	N & J
Cunner	A	133,000	12	Mortality rate 50% (15°C)	Rogers (1969)	N & J
Cunner	A	100,000	24	Mortality rate 50% (15°C)	Rogers (1969)	N & J
Cunner	A	72,000	48	Mortality rate 50% (15°C)	Rogers (1969)	N & J
Fish	A	3,000	240	Fish died	Kemp (1949)	N & J

Herring (Atlantic)	A	20	3	Reduced feeding rate	Johnson and Wildish (1982)	N & J
Hogchoker	A	1,240	24	Energy utilization increased	Sherk et al. (1975)	N & J
Hogchoker	A	1,240	120	Erythrocyte count increased	Sherk et al. (1975)	N & J
Hogchoker	A	1,240	120	Haematocrit increased	Sherk et al. (1975)	N & J
Killifish (striped)	A	960	120	Haematocrit increased	Sherk et al. (1975)	N & J
Killifish (striped)	A	3,277	24	Mortality rate 10% (FE)	Sherk et al. (1975)	N & J
Killifish (striped)	A	9,720	24	Mortality rate 10%	Sherk et al. (1975)	N & J
Killifish (striped)	A	3,819	24	Mortality rate 50%	Sherk et al. (1975)	N & J
Killifish (striped)	A	12,820	24	Mortality rate 50%	Sherk et al. (1975)	N & J
Killifish (striped)	A	16,930	24	Mortality rate 90%	Sherk et al. (1975)	N & J
Killifish (striped)	A	6,136	24	Mortality rate 90%	Sherk et al. (1975)	N & J
Menhaden (Atlantic)	A	154	24	Mortality rate 10% (FE)	Sherk et al. (1975)	N & J
Menhaden (Atlantic)	A	247	24	Mortality rate 50% (FE)	Sherk et al. (1975)	N & J
Menhaden (Atlantic)	A	396	24	Mortality rate 90% (FE)	Sherk et al. (1975)	N & J
Minnow (sheepshead)	A	200,000	24	Mortality rate 10% (15°C)	Rogers (1969)	N & J
Minnow (sheepshead)	A	300,000	24	Mortality rate 30% (10°C)	Rogers (1969)	N & J
Minnow (sheepshead)	A	100,000	24	Mortality rate 90% (19°C)	Rogers (1969)	N & J
Mummichog	A	300,000	24	No mortality (15°C)	Rogers (1969)	N & J
Mummichog	A	2,447	24	Mortality rate 10% (FE)	Sherk et al. (1975)	N & J
Mummichog	A	3,900	24	Mortality rate 50% (FE)	Sherk et al. (1975)	N & J
Mummichog	A	6,217	24	Mortality rate 90%	Sherk et al. (1975)	N & J
Perch (white)	A	650	120	Haematocrit increased	Sherk et al. (1975)	N & J

Perch (white)	A	650	120	Erythrocyte count increased	Sherk et al. (1975)	N & J
Perch (white)	A	650	120	Hemoglobin concentration increased	Sherk et al. (1975)	N & J
Perch (white)	A	305	120	Gill tissue may have been damaged	Sherk et al. (1975)	N & J
Perch (white)	A	650	120	Histological damage to gill tissue	Sherk et al. (1975)	N & J
Perch (white)	A	305	24	Mortality rate 10% (FE)	Sherk et al. (1975)	N & J
Perch (white)	A	985	24	Mortality rate 50%	Sherk et al. (1975)	N & J
Perch (white)	A	3,181	24	Mortality rate 90% (FE)	Sherk et al. (1975)	N & J
Rasbora (harlequin)	A	40,000	24	Fish died (BC)	Alabaster and Lloyd (1980)	N & J
Rasbora (harlequin)	A	6,000	168	No mortality	Alabaster and Lloyd (1980)	N & J
Shad (American)	A	150	0.25	Change in preferred swimming depth	Dadswell et al. (1983)	N & J
Silverside (Atlantic)	A	58	24	Mortality rate 10% (FE)	Sherk et al. (1975)	N & J
Silverside (Atlantic)	A	250	24	Mortality rate 50% (FE)	Sherk et al. (1975)	N & J
Silverside (Atlantic)	A	1,000	24	Mortality rate 90% (FE)	Sherk et al. (1975)	N & J
Spot	A	114	48	Mortality rate 10% (FE)	Sherk et al. (1975)	N & J
Spot	A	1,309	24	Mortality rate 10% (FE)	Sherk et al. (1975)	N & J
Spot	A	6,875	24	Mortality rate 10%	Sherk et al. (1975)	N & J
Spot	A	189	48	Mortality rate 50% (FE)	Sherk et al. (1975)	N & J
Spot	A	2,034	24	Mortality rate 50%	Sherk et al. (1975)	N & J
Spot	A	8,800	24	Mortality rate 50%	Sherk et al. (1975)	N & J
Spot	A	317	48	Mortality rate 90% (FE)	Sherk et al. (1975)	N & J
Spot	A	11,263	24	Mortality rate 90%	Sherk et al. (1975)	N & J
Stickleback (four spine)	A	100	24	Mortality rate <1% (IA)	Rogers (1969)	N & J

Stickleback (four spine)	A	10,000	24	No mortality (KS: 10-12°C)	Rogers (1969)	N & J
Stickleback (four spine)	A	300	24	Mortality rate ~50% (IA)	Rogers (1969)	N & J
Stickleback (four spine)	A	18,000	24	Mortality rate 50% (15.0-16.0°C)	Rogers (1969)	N & J
Stickleback (four spine)	A	50,000	24	Mortality rate 50% (KS)	Rogers (1969)	N & J
Stickleback (four spine)	A	53,000	24	Mortality rate 50% (10-12°C)	Rogers (1969)	N & J
Stickleback (four spine)	A	330,000	24	Mortality rate 50% (9.0-9.5°C)	Rogers (1969)	N & J
Stickleback (four spine)	A	500	24	Mortality rate 100%	Rogers (1969)	N & J
Stickleback (four spine)	A	200,000	24	Mortality rate 95% (KS)	Rogers (1969)	N & J
Stickleback (three spine)	A	28,000	96	No mortality in test designed to identify lethal threshold	LeGore and DesVoigne (1973)	N & J
Toadfish (oyster)	A	3,360	1	Oxygen consumption more variable in prestressed fish	Neumann et al. (1975)	N & J
Toadfish (oyster)	A	14,600	72	Fish largely unaffected, but developed latent ill effects	Neumann et al. (1975)	N & J
Toadfish (oyster)	A	11,090	72	Latent ill effects manifested in subsequent test at low SS	Neumann et al. (1975)	N & J
ADULT NONSALMONIDS						
Bass (largemouth)	A	62.5	720	Weight gain reduced ~ 50%	Buck (1956)	N & J
Bass (largemouth)	A	144.5	720	Growth retarded	Buck (1956)	N & J
Bass (largemouth)	A	144.5	720	Fish unable to reproduce	Buck (1956)	N & J
Bluegill	A	423	0.05	Rate of feeding reduced	Gardner (1981)	N & J
Bluegill	A	15	1	Reduced capacity to locate prey	Vinyard and O'Brien (1976)	N & J
Bluegill	A	144.5	720	Growth retarded	Buck (1956)	N & J
Bluegill	A	62.5	720	Weight gain reduced ~ 50%	Buck (1956)	N & J
Bluegill	A	144.5	720	Fish unable to reproduce	Buck (1956)	N & J

Carp (common)	A	25,000	336	Some mortality (MC)	Wallen (1951)	N & J
Darters	A	2,045	8,760	Darters absent	Vaughan (1979); Vaughan et al. (1982)	N & J
Fish	A	120	384	Density of fish reduced	Erman and Lignon (1988)	N & J
Fish	A	620	48	Fish kills downstream from sediment source	Hesse and Newcomb (1982)	N & J
Fish	A	900	720	Fish absent or markedly reduced in abundance	Herbert and Richards (1963)	N & J
Fish	A	2,045	8,760	Habitat destruction; fish populations smaller than expected	Vaughan (1979); Vaughan et al. (1982)	N & J
Fish (warm water)	A	100,000	252	Some fish died; most survived	Wallen (1951)	N & J
Fish (warm water)	A	200,000	1,125	Fish died; opercular cavities and gill filaments clogged	Wallen (1951)	N & J
Fish (warm water)	A	22	8,760	Fish populations destroyed	Menzel et al. (1984)	N & J
Goldfish	A	25,000	336	Some mortality (MC)	Wallen (1951)	N & J
Sunfish (green)	A	9,600	1	Rate of ventilation increased	Horkel and Pearson (1976)	N & J
Sunfish (red ear)	A	62.5	720	Weight gain reduced ~ 50% compared to controls	Buck (1956)	N & J
Sunfish (red ear)	A	144.5	720	Growth retarded	Buck (1956)	N & J
Sunfish (red ear)	A	144.5	720	Fish unable to reproduce	Buck (1956)	N & J

Appendix D

Revised model of the effects of suspended sediments on estuarine fishes

(C.P. Newcombe, Personal Communication)

Reprinted with permission of the author

Not included, available upon request

Appendix E.

Summary figures from Wilber and Clarke, 2001
Reprinted with permission of the authors

Not included, available upon request

Appendix F

Summary of state standards for suspended and bedded sediments.

Prepared by Kathleen Sullivan for the U.S.EPA, Office of Water.

Reprinted with permission of the U.S.EPA Office of Water (Chris Zabawa and Bill Swietlik)

See Appendix 4 of SABS Consultation Document

Available upon request