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

Internal Report

August 20, 2003

by

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**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.

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**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

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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).

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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.

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Table 1

*RANKING OF AQUATIC STRESSORS: SUSPENDED SOLIDS & BEDDED SEDIMENT*S
[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.

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**RELATIONAL TRENDS OF FRESH WATER FISH ACTIVITY TO TURBIDITY VALUES AND TIME**
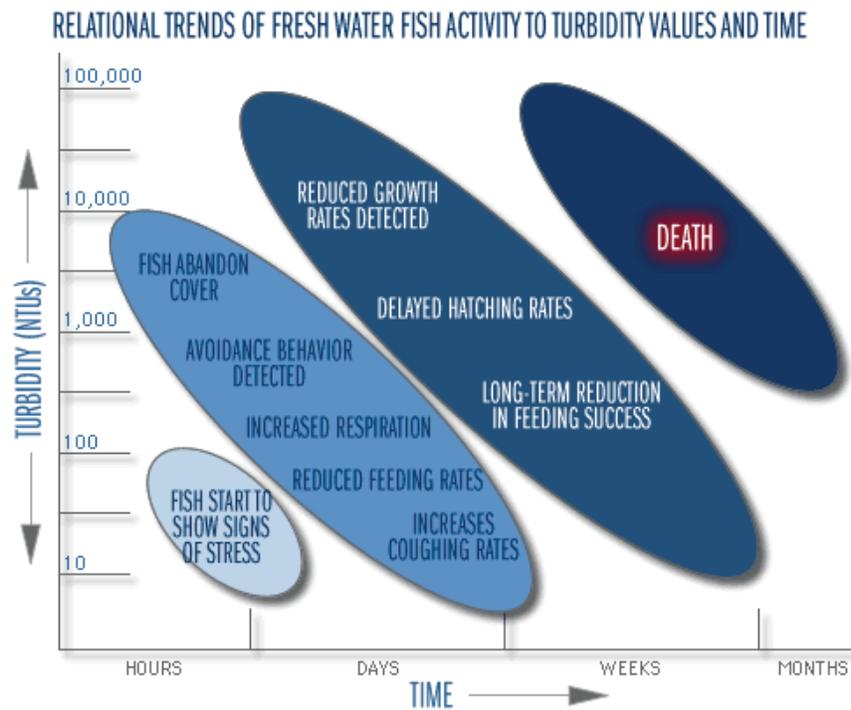
Figure 2.  Idealized model of fish response to increased suspended sediments.  Schematic source of above figure is unknown; it is a generic, un-calibrated impact assessment model based on   Newcombe, C. P., and J. O. T. Jensen. 1996. Channel suspended sediment and fisheries: a synthesis for quantitative 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 | Dura-tion 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 | |

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| | | | | | | |
|---|---|---|---|---|---|---|
| " " | 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 |

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| 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 communication 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 communication 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 |

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|---|---|---|---|---|---|---|
| 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 |

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|---|---|---|---|---|---|---|
| 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 |
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| **SALMONID EGGS AND LARVAE** | | | | | | |

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|---|---|---|---|---|---|---|
| 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 |

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|---|---|---|---|---|---|---|
| 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 |

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|---|---|---|---|---|---|---|
| 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 |

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|---|---|---|---|---|---|---|
| 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 |

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|---|---|---|---|---|---|---|
| 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 |

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|---|---|---|---|---|---|---|
| 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)

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# Appendix E.

## Summary figures from Wilber and Clarke, 2001
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# Appendix F

**Summary of state standards for suspended and bedded sediments.**

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

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See Appendix 4 of SABS Consultation Document

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