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The Effects of Sediment on Fish and their Habitat

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ABSTRACT

This document provides information and comments on guidelines and criteria for the protection of aquatic resources from elevated levels of suspended and deposited sediment in fresh water.

Elevated levels of sediment and turbidities can reduce the productivity of aquatic systems. Both these correlated factors have the potential to decrease primary productivity that may have consequences to secondary productivity and the energy flow to higher trophic levels.

Examples of the lethal and sublethal effects of sediment on fish and their habitat, and factors such as temperature, particle size and angularity, and duration of exposure, that influence these effects, are presented. Levels of suspended sediment that have been determined to be acutely lethal to fish typically range from the hundreds to hundreds of thousands of $mg \cdot L^{-1}$ sediment, while sublethal effects are often manifest in the tens to hundreds of $mg \cdot L^{-1}$ sediment.

Guidelines that rely on gravimetric determinations of suspended sediment concentrations are recommended for use over those that rely solely on turbidity. However, if the relationship is known between these variables, then turbidity may be used as a surrogate for suspended sediment. The use of guidelines that incorporate the duration of exposure to sediments provide useful analytical information for predictive purposes, but caution is warranted when attempting to predict the effects of low (≤ tens of $mg \cdot L^{-1}$) levels of sediment over protracted periods of time. Guidelines that rely on the volumetric determination of “settleable solids” are not endorsed for use because of the difficulty of obtaining a meaningful and generally applicable relationship between this variable and suspended solids.

It is concluded that elevated levels of sediment (typically over background) may be harmful to fish (i.e. acutely lethal, or elicit sublethal responses that could compromise their well-being and jeopardize survival), and in addition, negatively impact on their habitat.

RÉSUMÉ

Ce document donne de l'information et des commentaires au sujet de lignes directrices et de critères pour la protection de ressources aquatiques contre des niveaux élevés de sédiments en suspension ou déposés en eau douce.

Des niveaux élevés de sédiments et de turbidité peuvent réduire la productivité des systèmes aquatiques. Ces deux facteurs reliés ont le potentiel de réduire la productivité primaire qui peut avoir des répercussions sur la productivité secondaire et l'acheminement d'énergie vers des niveaux trophiques plus élevés.

Le document donne des exemples des effets létaux et sublétaux des sédiments sur le poisson et son habitat, ainsi que des facteurs comme la température, la taille et l'angularité des particules et la durée d'exposition qui influent sur ces effets. Les niveaux de sédiments en suspension qui sont très létaux vont généralement de centaines à centaines de milliers de $mg \cdot L^{-1}$ de sédiments tandis que les effets sublétaux se manifestent souvent à des niveaux de dizaines à centaines de $mg \cdot L^{-1}$ de sédiments.

On recommande de faire appel aux lignes directrices qui reposent sur les déterminations gravimétriques de concentrations de sédiments en suspension plutôt que de ne se fier qu'à celles qui dépendent de la turbidité seulement. Toutefois, si on connaît le rapport entre ces variables, la turbidité peut être utilisée à la place des sédiments en suspension. Des lignes directrices qui incorporent la durée de l'exposition aux sédiments donnent de l'information analytique utile aux fins de prévisions, mais il faut faire preuve de prudence lorsque l'on tente de prévoir les effets de faibles niveaux (≤ dizaines de $mg \cdot L^{-1}$) de sédiments sur des périodes prolongées. Les lignes directrices qui reposent sur la détermination volumétrique des « particules sédimentables » ne sont pas recommandées en raison de la difficulté d'obtenir un rapport fondé et généralement applicable entre cette variable et les particules en suspension.

On conclut que des niveaux élevés de sédiments (surtout ambiants) peuvent nuire au poisson (c'est-à-dire très létaux ou provoquant des réactions sublétales qui pourraient compromettre son bien-être et mettre sa survie en danger), et ont en outre des répercussions négatives sur l'habitat.

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GLOSSARY OF SELECTED TERMS

The terminology and explanations presented here are based on those used in the Canadian Council of Ministers of the Environment (CCME) (1999) guideline relating to “Total Particulate Matter” and the protection of aquatic life, and from Lloyd (1985).

Suspended sediment

Matter suspended in waters may be inorganic or organic in origin. The type and concentration of this suspended matter, and dissolved material, controls the turbidity and transparency of water (CCME 1999).

The amount of suspended matter is determined through the filtration of a known volume of water. According to Caux et al. (1997) the suspended solids (the non-filterable residue – CCME (1999)) are that fraction of material retained on a 0.45 µm pore-diameter glass-fiber filter (only fine and very fine clays pass through such a filter).

For the purposes of this document, the non-filterable residue (which may contain both organic and inorganic components) will be referred to as suspended sediment. The concentration of suspended sediment in water is expressed as the quantity in 1 litre; typically it is expressed in terms of milligrams per litre ($mg \cdot L^{-1}$).

Turbidity

Turbidity is a measure of the lack of clarity or degree of transparency of water caused by inorganic and organic suspended or dissolved substances. Turbidity is an expression of the optical properties of substances that cause light to be scattered and absorbed. Turbidity may be measured through the use of nephelometry (light scattering).

A turbidity meter, which measures and relates the scattering of light in a water sample to a range of known turbidity standards, provides values that are presented as Nephelometric Turbidity Units (NTUs).

For a particular sediment suspension in water there is a strong relationship between turbidity and the concentration of material (the greater the amount of material in suspension the higher will be the turbidity).

Settleable solids

The volume of settleable solids within a water sample is determined by allowing such material in a 1-litre sample to settle during one hour to the base of an Imhoff cone. The volume of material that settles in the Imhoff cone is recorded in milliliters, and expressed in relation to the volume of the water sample that was analyzed, for example, as milliliters per litre ($mL \cdot L^{-1}$).

Geometric mean diameter (Dg)

Geometric mean diameter is a measure of streambed substrate composition that has been related to the survival of salmonid embryos, such that survival is lower when the geometric mean diameter is smaller.
The geometric mean diameter (Dg) is calculated from the following equation:

$$Dg = (d_{84} \cdot d_{16})^{0.5}$$

where d_{84} is the 84th percentile particle size, and d_{16} is the 16th percentile particle size, both of which are taken from plots of the particle size distribution of the sample (CCME 1999).

Fredle number (FN)

The Fredle number is another descriptor of the composition of streambed substrate. It is considered to be a better descriptor than the percentage of fines or the geometric mean diameter as it integrates more information on the overall particle size distribution of the substrate. The survival rate of salmonid embryos during incubation drops rapidly when the Fredle number falls below 5 (CCME 1999).

The Fredle number is calculated by dividing the geometric mean diameter of a sediment sample by the Sorting Coefficient (So). The sorting coefficient = $(d_{75} \div d_{25})^{0.5}$, can be used as a measure of the pore size of the streambed substrate (CCME 1999).

Compensation depth

This is the depth of an aquatic system at which light intensity is just sufficient to promote that level of photosynthesis equaling the respiratory, or metabolic, requirements of the phytoplankton population. It is usually considered to be the depth at which 1% of available surface light penetrates into a body of water. Net production of plant material occurs above this depth.

Euphotic volume

This is the volume of water above the compensation depth, and equals the surface area multiplied by the compensation depth.

96-h LC50

The concentration of material that proves lethal to 50% of a number of organisms being tested after an exposure of 96 hours.

INTRODUCTION

This document provides information on the potential effects that sediments may have on fish and their habitat. It is not a comprehensive review of the large body of sediment related literature. The document is intended for those who require a synopsis of the effects of sediment in aquatic systems, and at the same time it provides information on pertinent guidelines and recommendations designed to protect such waters.

Sediments occur naturally and are integral components of aquatic systems. Nearly all waters have some solid matter in suspension that may be of physical, chemical or biological origin; such quantities of material usually vary with season. This natural variation in suspended sediment concentrations occurs, typically, in response to events (e.g. rain fall, snow and glacial melting) which increase both water flows and levels which results in land erosion and sediment input to waterways. The increased energy within watercourses may move the streambed substrate and thereby increase the amount of material in suspension. Accordingly, aquatic organisms are subjected to these natural variations. They have adapted their life cycle to accommodate them and in so doing ensured the survival of the species. However, in addition to natural seasonal fluctuations of sediment levels in the aquatic environment, there are catastrophic events, such as volcanic eruptions, and certain anthropogenic activities that have the potential to add unusually large amounts of sediment to a water body, thereby markedly affecting its physical, chemical, and biological structure and integrity. Such activities as logging, road building, dredging, and placer (gold) mining, etc., may cause significant environmental changes proximal to the activity and at distances further downstream.

A perspective on the potential effects of sediment in aquatic systems may be obtained from some of the world-wide body of literature. Of particular note are those documents that present a scientific review of the literature on the effects of sediment on aquatic organisms and their habitat that were used during the formulation and recommendation of water quality criteria and guidelines. These include the European Inland Fisheries Advisory Commission (EIFAC) (1964), Hollis et al. (1964), United States Environmental Protection Agency (USEPA) (1986), Canadian Council of Resource and Environment Ministers (CCREM) (1987), Lloyd (1987), Lloyd et al. (1987), Newcombe and MacDonald (1991), Government of Canada (1993), Anderson et al. (1996), Newcombe and Jensen (1996), Caux et al. (1997), British Columbia Ministry of Environment, Lands and Parks (BCMELP) (1998), and the Canadian Council of Ministers of the Environment (CCME) (1999). These documents and pertinent recent ancillary information (e.g. Waters 1995) were used in the preparation of this report.

WATER QUALITY CRITERIA, GUIDELINES AND RECOMMENDATIONS

Although sediment and its associated effects on water clarity and turbidity is an inherent component of aquatic systems, it is apparent from the literature that there is an increased risk to the survival and well-being of aquatic organisms when levels exceed background values for a particular period of time.

Research has shown that elevated levels of sediment, above background values, can be detrimental to aquatic biota; this has led to the formulation of water quality criteria, guidelines and recommendations.

Pertinent examples of guidelines to protect aquatic resources from elevated levels of sediment

The European Inland Fisheries Advisory Commission (1964), deduced that:

<25 parts per million* (ppm) of suspended solids - no evidence of harmful effects on fish and fisheries;

25 - 80 ppm - it should be possible to maintain good to moderate fisheries, however the yield would be somewhat diminished relative to waters with <25 ppm suspended solids;

80 - 400 ppm - these waters are unlikely to support good freshwater fisheries; and

>400 ppm suspended solids - at best, only poor fisheries are likely to be found.

*(parts per million approximate $mg \cdot L^{-1}$)

In addition "...there is evidence that not all species of fish are equally susceptible to suspended solids, and that not all kinds of solids are equally harmful."

EIFAC also commented that "...although several thousand parts per million solids may not kill fish during several hours or days exposure such temporary high concentrations should be prevented in rivers where good fisheries are to be maintained."

"The spawning grounds of salmon and trout require special consideration and should be kept as free as possible from finely divided solids."

These criteria were formulated after a review of the available literature, and EIFAC deduced that, "...there are at least five ways in which an excessive concentration of finely divided solid matter might be harmful to a fishery.

1) By acting directly on the fish swimming in the water in which solids are suspended, and either killing them or reducing their growth rate, resistance to disease, etc.

2) By preventing the successful development of fish eggs and larvae.

3) By modifying natural movements and migrations of fish.

4) By reducing the abundance of food available to the fish.

5) By affecting the efficiency of methods for catching fish."

Despite more research having been carried out to examine the effects of sediment on aquatic organisms, these general conclusions and findings of EIFAC (1964) are still valid.

The United States Environmental Protection Agency (1986) chose criteria which addressed solids (suspended, settleable) and turbidity together, and stipulated that settleable and suspended solids should not reduce the depth of the compensation point for photosynthetic activity by more than 10% from the seasonally established norm for aquatic life.

The Canadian Council of Resource and Environment Ministers (CCREM) (1987) provided guidelines for total suspended solids. They recommend that suspended solids should not be elevated more than 10 $mg \cdot L^{-1}$ above background levels when background is ≤ 100 $mg \cdot L^{-1}$. Suspended solids should not exceed 10% of background concentrations when background concentrations are >100 $mg \cdot L^{-1}$.

The British Columbia Ministry of Environment, Lands, and Parks (1998), and the Canadian Council of Ministers of the Environment (CCME) (1999) guidelines are the most recent documents on this topic, and they are based, in part, on the publication by Caux et al. (1997).

Listed below are the CCME (1999) recommendations for suspended sediments, turbidity, and streambed substrate (at the time of writing, these recommendations were unpublished).

Suspended sediments.

<u>Clear flow</u>: Maximum increase of 25 $mg \cdot L^{-1}$ from background levels for short-term (e.g. <24 h) exposures, and a maximum average increase of 5 $mg \cdot L^{-1}$ from background for longer-term exposures (e.g. 24 h to 30 d).

<u>High flow</u>: Maximum increase of 25 $mg \cdot L^{-1}$ from background levels at any time when background levels are between 25 $mg \cdot L^{-1}$ and 250 $mg \cdot L^{-1}$. Should not increase more than 10% of background levels when background levels are >250 $mg \cdot L^{-1}$.

Turbidity.

<u>Clear flow</u>: Maximum increase of 8 NTUs from background levels for short-term (e.g. <24 h) exposures, and a maximum average increase of 2 NTUs from background for longer-term exposures (e.g. 24 h to 30 d).

High flow or turbid waters: Maximum increase of 8 NTUs from background levels at any time when background levels are between 8 NTUs and 80 NTUs. Turbidity should not increase more than 10% of background levels when background levels are >80 NTUs.

Deposited bed load sediment.

Insufficient information to derive guidelines.

Streambed substrate.

Fine sediments: The quantity in streambed substrates should not exceed 10% of particles <2 mm, 19% of particles <3 mm, and 25% of particles <6.35 mm.

Geometric mean diameter (GMD): The GMD should not exceed 12 mm.

Fredle number: The Fredle number should not be <5 mm.

Inter-gravel dissolved oxygen: Minimum 6.5 $mg{\cdot}L^{-1}$.

Caux et al. (1997), and CCME (1999) provide details regarding the derivation of the above criteria. The ratio of the concentration of suspended solids to turbidity is considered to be about 3 to 1 (CCME 1999), but depending on the nature of the material in suspension this ratio can vary substantially from that used in the guideline (refer to Lloyd et al. 1987; Lloyd 1987). Based on an examination of a large data set by Lloyd (1987), the prediction of turbidity caused by suspended sediment varied with the stream examined: the ratio of turbidity (NTUs) to $mg{\cdot}L^{-1}$ of suspended sediment ranged from 1:1 to 1:5. Accordingly, it is critical to establish the relationship between suspended sediment and turbidity if values of the latter are to be used in guidelines for the protection of aquatic organisms. Lloyd (1987) states that measurements of turbidity can be used to identify at least threshold levels of suspended sediment concentrations for a broad range of watersheds.

As the turbidity guidelines were deduced from the suspended sediment guidelines (CCME 1999), it would be prudent to rely on the latter when assessing potential impacts from sediments.

Another cautionary note is offered in relation to the guideline of a minimum dissolved oxygen content of 6.5 $mg{\cdot}L^{-1}$ in inter-gravel substrate waters. In that both the partial pressure of oxygen and its concentration are important in ensuring that adequate dissolved oxygen enters the blood of salmonids, reliance on a variable that is influenced by other factors such as temperature (and salinity in estuarine and marine waters) is potentially problematic. Furthermore, the requirement of developing salmonid embryos for oxygen is greatest just prior to hatch, and at such times dissolved oxygen levels in excess of 6.5 $mg{\cdot}L^{-1}$ will be required (Rombough 1988).

Turbidity

Lloyd et al. (1987) carried out an extensive review of the literature on the effects of turbidity and suspended sediment on aquatic organisms and in doing so derived relationships between turbidity and suspended sediment. This review was the basis for recommending turbidity as a water quality standard for salmonid habitats in Alaska (Lloyd 1987). Lloyd concluded that "the continued application of Alaska's present water quality standard for the propagation of fish and wildlife (25 NTUs above natural conditions in streams and lakes) can be expected to provide a moderate level of protection for clear cold water habitats. A higher level of protection would require a more restrictive turbidity standard, perhaps similar to the one currently applied to drinking water in Alaska (5 NTUs above natural conditions in streams and lakes)…..Even more stricter limits may be warranted to protect extremely clear waters, due to the dramatic initial impact of turbidity on light penetration (Lloyd et al. 1987). However, such stringent limits do not appear to be necessary to protect naturally turbid systems where it may be possible to establish tiered or graded standards based on ambient water quality." The level of 25 NTUs approximates to 100 $mg \cdot L^{-1}$ and 5 NTUs to 20 $mg \cdot L^{-1}$ suspended sediment for Alaskan streams, based on the assessments of data by Lloyd et al. (1987).

Duration of exposure to sediment

Newcombe and MacDonald (1991), Anderson et al. (1996), and Newcombe and Jensen (1996) provide recent reviews of the literature on the effects of sediment on aquatic ecosystems. These authors state that aquatic biota respond to both the concentration of suspended sediments and the duration of exposure to them, and relate the two through an "index of pollution intensity (stress index)." It was Newcombe and MacDonalds' (1991) recommendation that the use of the "stress index" would provide resource managers with a method to predict the effects of "pollution episodes on aquatic biota." This concept was criticized by, for example, Servizi and Martens (1992) and Gregory et al. (1993), because of its inaccuracy as a predictive tool. The data used by Newcombe and MacDonald (1991), together with more recent information were subjected to a more rigorous analysis to determine if the "stress index"/"severity of effect" relationships with duration of exposure were applicable to different species and life cycle stages (Newcombe and Jensen 1996; Caux et al. 1997). Models that utilize a function of concentration and duration of exposure to predict potential harm (Newcombe and MacDonald 1991; Newcombe and Jensen 1996; Caux et al. 1997; BCMELP 1998; CCME 1999), reveal that an increasing duration of exposure to sediment potentially results in increasing harm to fish, life cycle stages, and to other aquatic organisms. Such trend identification is of value in predicting the potential effects of sediment on aquatic organisms, but caution must be exercised when assessing the effects of low concentrations (≤ tens of $mg \cdot L^{-1}$)

of suspended sediment over protracted periods of time. Furthermore, it is likely that there would be increased variation in the response among individual organisms and life stages of organisms to the effects of elevated but low and sublethal levels of sediment, relative to less variable responses at higher sediment levels due to a greater severity of effect and less scope for adaptation, tolerance and resistance. Because of this, judicious application of these models is warranted when assessing the potential impacts of exposure to such low levels of suspended sediment.

The models address the issue of sublethal effects and certain responses (such as avoidance, territorial and feeding responses) are implied to be of less concern if they are reversible (Anderson et al. 1996; CCME 1999). Although an avoidance response may be an initial adaptive survival strategy, displacement could be detrimental. It is possible that the consequences of fish moving from preferred habitat, to avoid increasing levels of suspended sediment, may not be beneficial if displacement is to sub-optimal habitat, and that they also become stressed and, accordingly, more vulnerable to, for example, predation (Birtwell et al. 1999).

Notwithstanding these comments, the reports by Newcombe and MacDonald (1991), Anderson et al. (1996), and Newcombe and Jensen (1996), in addition to that of Waters (1995) provide timely additions to the literature and have contributed to improved guidelines (Caux et al. 1997; BCMELP 1998; CCME 1999).

Placer gold mining in the Yukon Territory and the risk to aquatic organisms

An assessment of the application of sediment criteria and relevant literature occurred during the revision of the Yukon Placer Authorization (Government of Canada 1993). This document describes the conditions under which placer mining can occur in the Yukon Territory and the sediment levels that may be discharged to streams.

The document recognizes the importance of placer mining in the Yukon and permits the "harmful alteration, disruption or destruction of fish habitat" in certain streams, and/or the discharge of sediment subject to compliance with the standards of allowable sediment discharge which are described in the Authorization. The compliance schedule that the placer miners must meet for the discharge of sediment into streams has been categorized according to the water's biological resources and their current and potential uses. For example, since January 1994 there was to be no increase in the concentration of sediments above background levels in salmon spawning streams, whereas for salmonid rearing streams the permissible increase is less than 200 $mg \cdot L^{-1}$.

It was recognized that there was some level of risk to aquatic organisms depending upon the sediment levels discharged and the sensitivity of the organisms in the receiving stream. After a review of available information it was determined that the impacts could be classified in relation to the levels of risk to which the fish habitat would be subjected, and that these impacts would be best assessed using increases in the concentration of suspended sediment above background levels. The levels of risk and the corresponding concentrations of sediment follow:

Sediment increase ($mg \cdot L^{-1}$)	Risk to fish and their habitat
0	No risk
<25	Very low risk
25 - 100	Low risk
100 - 200	Moderate risk
200 - 400	High risk
>400	Unacceptable risk

Based on the preceding assessments, it is concluded that suspended sediments have been shown to be harmful to aquatic systems in many areas of the world, hence the derivation and the recommendation of criteria and guidelines for the protection of aquatic organisms.

SPECIFIC EFFECTS OF SEDIMENT ON FISH AND THEIR HABITAT

Levels of sediment that are lethal to fish

Numerous reports, such as those of EIFAC (1964), Hollis et al. (1964), Lloyd (1987), Newcombe and MacDonald (1991), Waters (1995), Anderson et al. (1996), Caux et al. (1997), provide information summarizing the effects of sediment and turbidity on fish. The results of the studies identified in these reports are not easily compared owing to their use of different species, life cycle stages and test methodologies. However, it has been determined that certain concentrations of sediment kill fish directly. These concentrations typically range from the hundreds to hundreds of thousands of $mg \cdot L^{-1}$ of sediment.

The typical test that determines the lethality of a pollutant is generally carried out over 96 hours. The concentration (lethal concentration - LC) of a substance that kills 50% of the test organisms in 96 hours is referred to as the 96-h LC50; it is a basic toxicological test. By determining the 96-h LC50 values for a range of substances, their relative short-term toxicity can be compared. Such values do not indicate the effects of a more prolonged exposure to the contaminant, they do not address the onset of death in the test fish (which is of great significance to the individuals of a population), nor do they relate to effects on fish habitat. Therefore, the results of the 96-h LC50 test have limited value for predicting effects in the wild and at best they are but a coarse indicator of the short-term effects of a contaminant.

Servizi and Martens (1987), and Servizi and Gordon (1990) determined the 96-h LC50 for juvenile sockeye and chinook salmon exposed to Fraser River sediment to be 17,600 $mg \cdot L^{-1}$ and 31,000 $mg \cdot L^{-1}$ respectively. Servizi and Martens (1991) also examined the short-term lethal effects of Fraser River sediment on coho salmon in relation to temperature, season, and fish size. They determined that the 96-h LC50 values were affected by temperature over the range 1 °C to 18 °C, and that juvenile coho were most tolerant of sediment at 7 °C (96-h LC50 22,700 $mg \cdot L^{-1}$). However at 18 °C the fish were less tolerant to the sediment and the 96-h LC50 was 7,000 $mg \cdot L^{-1}$ to 8,100 $mg \cdot L^{-1}$. Of significance, was the determination of the concentration of sediment that did not kill fish during the 96-h exposure period. At 7 °C this value was 8,100 $mg \cdot L^{-1}$, and the mean concentration at which the first death occurred was 8,200 $mg \cdot L^{-1}$. Furthermore, at 1 °C and 18 °C the "no mortality" levels occurred at lower sediment concentrations, and at 18 °C this was approximately 2,000 $mg \cdot L^{-1}$. The onset of mortality at 18 °C occurred at about 3,000 $mg \cdot L^{-1}$.

The effects of equivalent elevated levels of sediment in the wild could be more harmful to fish in the short-term than these laboratory toxicity data imply, because the conditions in the laboratory do not exactly replicate conditions experienced in the wild e.g., with respect to sediment type, water velocity and potential abrasive and scouring effects, fluctuations in dissolved gases, and feeding and food supplies. Accordingly, the 96-h LC50 data may underestimate the short-term lethal and sublethal effects in the wild (refer to Martin et al. 1984).

Particle size and angularity

The effect of angularity and the size of sediment particles on fish has received little attention in the literature, but they are factors that influence the response of fish to suspended sediments (Servizi and Martens 1987). These authors determined that lethality of natural Fraser River sediments to under-yearling sockeye salmon increased with increasing particle size.

According to Lake and Hinch (1999) it has often been speculated that the shape of suspended sediment particles may affect physiological stress and hence mortality in fish. Using the same experimental apparatus that Servizi and Martens (1987, 1991) employed, Lake and Hinch (1999) determined that highly angular particles placed juvenile coho under stress at lower concentrations than did round particles. However, they did not find a difference among sediment particle shapes in causing mortality (mortality was not observed until concentrations were about 100 $g \cdot L^{-1}$). These authors suggest that suspended particle angularity may not be the main factor responsible for acute lethality in juveniles because their bioassay results yielded a value that was seven times greater than that determined by Servizi and Martens (1991) who exposed juvenile coho salmon to natural Fraser River sediment. The results of Lake and Hinch (1999) also contrast with those obtained from static laboratory bioassays which were used to examine the effects on coho salmon of exposure to angular volcanic ash (Stober et al. 1981). The latter studies yielded a 96-h LC50 value of 28,184 $mg \cdot L^{-1}$ (similar to that recorded by Servizi and Martens (1991)), but a 96-h LC50 value of 509 $mg \cdot L^{-1}$ was calculated for smolts held *in-situ* (live boxes) in the Toutle and Cowlitz Rivers which carried volcanic ash (Stober et al. 1981). Additional environmental factors were considered to lower the tolerance of the salmon to the volcanic sediment. In other experiments Newcombe and Flagg (1983) exposed juvenile sockeye salmon to volcanic ash and determined a 36-h LC50 of 6,100 $mg \cdot L^{-1}$. Servizi and Martens (1987) provide more information from published reports that describe the increased effects of angular versus round and smooth particles on fish.

Thus, although Lake and Hinch (1999) determined that sediment particle shape had little effect on test fish mortality, other studies revealed that angular particles do cause mortality at lower concentrations than those determined by these authors, and that such effects may be exacerbated under “natural” conditions. The highly angular volcanic ash (tephra) that entered the streams in Washington as a result of the Mount St. Helens eruption was considered to be a major contributing factor to the loss of fishery resources in the affected areas (Martin et al. 1984).

Sublethal effects of sediment on fish and their habitat

In their natural environment, the survival of fish depends upon many factors not the least of which are finding food, predator avoidance, immune system health and reproduction; for salmonids, sediment has the potential to affect all of these factors (refer to, for example, Lloyd 1987; Newcombe and MacDonald 1991; Waters 1995; Anderson et al. 1996; Newcombe and Jensen 1996).

Feeding and growth

For many fish the successful capture of prey is a fundamental requirement in order to obtain food, and for facultative and opportunistic sight feeding juvenile salmonids this process may be affected by variations in suspended sediments. In laboratory streams it was demonstrated by McLeay et al. (1987) that under-yearling Arctic grayling exposed to suspended sediment for six weeks had impaired feeding activity and reduced growth rates at concentrations above 100 $mg{\cdot}L^{-1}$. We carried out a similar study with under-yearling chinook salmon (Liber and Birtwell, in preparation) and, although feeding activity was impaired from above 100 $mg{\cdot}L^{-1}$ to 400 $mg{\cdot}L^{-1}$ suspended sediment in trials using live prey, there was no concomitant reduction in growth over 28 d (the fish were fed a standard commercial diet and exposed to sediment concentrations up to 3,413 $mg{\cdot}L^{-1}$). The reasons for the absence of reduced growth (despite an impairment of feeding) may relate to the different feeding strategies of this fish species relative to those of Arctic grayling, and the experimental protocols employed in the respective studies. Other studies, by Scannell (1988), revealed the importance of water clarity (turbidity) in the feeding of Arctic grayling. He determined that at 10 NTUs (63 $mg{\cdot}L^{-1}$) only 10% of the Arctic grayling's food supply would be available, and at 25 NTUs only 3% would be available. This conclusion is based on the effect that turbidity (suspended sediment) has on reducing macroinvertebrate prey density and, at the same time, decreasing the “reactive volume” of water which is immediately in front of the fish and in which it feeds.

Cover and risk of predation

There is evidence that some juvenile salmonids may benefit from the occupation of turbid waters through a reduced risk of predation (Gregory 1993). Gregory considered that whereas visual ability may be affected in turbid waters there can be increased feeding because of a reduced risk of predation (Gregory and Northcote 1993). This premise is valid for predators that rely on sight to locate prey, but may not be as tenable for those predators that locate their prey by other means and are adapted to feed in turbid environments (Vandenbyllaardt et al. 1991). However, at more elevated turbidities and suspended sediment concentrations (above 150 NTUs) the "visual ability of juvenile chinook becomes substantially impaired and foraging ability is reduced regardless of any concurrent gains" (Gregory and Northcote 1993).

Coho salmon smolts reduced or ceased feeding when exposed to 100 $mg{\cdot}L^{-1}$ and >300 $mg{\cdot}L^{-1}$ suspended sediment, respectively (Noggle 1978). Juvenile coho salmon also displayed surfacing behavior when exposed to concentrations of suspended sediment above 2,550 $mg{\cdot}L^{-1}$ (Servizi and Martens 1992). A similar behavioral response to elevated suspended sediment was recorded for juvenile chinook salmon exposed to 3,413 $mg{\cdot}L^{-1}$ (Liber and Birtwell, in preparation). Such surfacing behavior at elevated sediment levels may well increase the risk to avian predators. Avian predation on surfacing fish and others that were behaving abnormally has been documented in relation to the discharge of pollutants (e.g. Birtwell et al. 1983), and on visually-impaired salmon released from a hatchery (Mace 1983).

Avoidance and displacement

While a reduced risk of predation to juvenile fish may occur at naturally-occurring low levels of turbidity, there is evidence that other sublethal effects may also occur which are indicative of stress, and/or elicit responses in fish. For example, Scannell (1988) determined that Arctic grayling avoided waters with turbidities greater than 20 NTUs (86 $mg{\cdot}L^{-1}$). McLeay et al. (1987) determined that Arctic grayling were displaced downstream at suspended sediment levels >100 $mg{\cdot}L^{-1}$. Similarly, Sigler et al. (1984) found downstream displacement of steelhead trout and coho salmon fry in artificial streams receiving suspensions of clay with turbidity values as low as 25 NTUs. We determined that juvenile chinook salmon were distributed in downstream sections of artificial streams receiving suspended sediment levels >76 $mg{\cdot}L^{-1}$, and that previously unexposed fish showed a distinct (80%) preference for clear water (0 $mg{\cdot}L^{-1}$ suspended sediment), generally avoiding all suspended sediment levels >20 $mg{\cdot}L^{-1}$ (Liber and Birtwell, in preparation).

Whitman et al. (1982) report that a 7-d exposure of chinook salmon to 650 $mg{\cdot}L^{-1}$ of volcanic ash did not appear to have an effect on homing. However, the salmon had a preference for ash-free water and the upstream movements of adults were reduced in waters carrying volcanic ash. Coho salmon avoided spawning in their natal Toutle River where mean suspended sediment (volcanic ash) concentrations were 3,408 $mg{\cdot}L^{-1}$. In addition, large numbers were displaced to the Cowlitz River that had mean suspended sediment concentrations of 1,069 $mg{\cdot}L^{-1}$ (Whitman et al. 1982). Adult salmon spawned in unstable volcanic substrates with average concentrations of fines (< 0.85 mm) ranging from 11.2% to 36.0%. Survival of eggs to the hatching stage in volcanic substrate varied from 50% to 95%; successful reproduction in the sediment-impacted streams was attributed to temporary groundwater upwelling (Whitman et al. 1982).

Other indices of stress, and salmonid rearing and migration in the Fraser River

Servizi and Martens (1992) recorded indices of stress (elevated blood sugar levels, and coughing frequency) in under-yearling coho salmon exposed to Fraser River sediment >20 mg·L^{-1}, and concluded that suspended sediment levels recorded in the Fraser River could elicit sublethal responses in these fish. This opinion was shared by Scrivener et al. (1994) who considered that juvenile chinook salmon use non-natal clear streams to "seek relief" from sediment concentrations in the Fraser River thereby reducing "stress" while seeking food and refuge. Scrivener et al. (1994) stated that suspended sediment concentrations "even along the river margins (150 mg·L^{-1}) could disrupt feeding, growth and social behavior, cause some fish displacement, and increase susceptibility to disease." These findings and deductions emphasize the requirement for the maintenance of stream integrity and their protection from events (such as those that result in elevations in suspended sediment and deposition) that would reduce their value to migrating and summer and winter rearing salmonids. Current (1999) research in the Yukon Territory is examining year-round use of non-natal streams by juvenile chinook salmon and supports this view (M. Bradford, Fisheries and Oceans Canada, Vancouver, personal communication).

Servizi and Martens (1987) report the effects of suspended Fraser River sediments on sockeye salmon. They recorded a number of sublethal responses that could affect survival. Gill trauma was observed in under-yearlings exposed for 96 h to 3,148 mg·L^{-1} suspended sediment, and smolts suffered a slight impairment in osmoregulatory capacity when exposed for the same duration to 14,407 mg·L^{-1} suspended sediment. Adult sockeye salmon displayed a secondary stress response following 9-d and 15-d exposures to 1,500 mg·L^{-1} and 500 mg·L^{-1} of fine sediments: levels that have been recorded in the Fraser River during summer (freshet) spawning migrations (Servizi and Martens 1987). But for the period of peak river flows (April to July) resulting from the springtime snow melt in the watershed, suspended sediment concentrations in lower Fraser River waters are typically below 50 mg·L^{-1}, and much lower in wintertime (Milliman 1980).

Cumulative effects and fish health

The effect of suspended sediment on the health of fish has received some attention (refer to Lloyd 1987; McLeay et al. 1987; Newcombe and MacDonald 1991; Anderson et al. 1996; Newcombe and Jensen 1996), as has the combined effects with other potential stressors (refer to McLeay et al. 1987, and Servizi and Martens 1991. For example, Hebert and Merkens (1961) found that exposure to elevated levels of sediment increased the susceptibility to fin rot among rainbow trout, while McLeay et al. (1987) and Liber and Birtwell (in preparation) found that Arctic grayling and chinook salmon juveniles, respectively, were less tolerant to a contaminant after exposure to suspended sediment.

Environmental relevance

The relevance of sublethal effects due to exposure to sediment is not easy to deduce, and the focus of my current research is on this topic. In the highly competitive environment in which aquatic organisms live, the maintenance of health and performance are prerequisites for survival. It is apparent, for example, that even brief exposure of juvenile salmon to a contaminant at the sublethal level can jeopardize survival by increasing their susceptibility to predation (Kruzynski and Birtwell 1994). An increasing body of information indicates that the sublethal exposure of juvenile fish to a stressor, or combination of stressors, increases their susceptibility to predation and hence jeopardizes survival (Mesa et al. 1994; Birtwell et al. 1999) and potentially compromises physiological performance (Farrell et al. 1998; Jain et al. 1998).

The foregoing comments on the sublethal effects of sediment on fish reveal that a variety of responses may be evoked, some of which are stressful to the exposed individuals. Stressful conditions are known to reduce the adaptive responses of salmonid fish to natural environmental fluctuations and increase their susceptibility to disease (Wedemeyer et al. 1976, 1991; Wedemeyer and McLeay, 1981, as cited in McLeay et al. 1987).

REPRODUCTION AND DEPOSITED MATERIAL

Effects of sediment on the eggs of salmonids

Sediment impacts on salmonid reproduction has been investigated particularly in relation to effects on eggs and the survival of embryos and alevins. “Settleable solids” in river waters have the potential to be deposited in the stream, especially under reduced flow conditions, where they may exert a detrimental influence on salmonid egg and alevin survival in spawning beds. It is also probable that the deposition of material could reduce the quality of substrate in that salmon eggs are deposited in the streambed substrate where they are particularly susceptible to in-stream disturbances and the deposition of material. Reiser and White (1988), for example, deduced that different sizes of sediment and combinations thereof affected steelhead trout and chinook salmon egg survival, and that it is the smaller sediments (<0.84 mm) that are most detrimental to the developing eggs. During an investigation of the effects of sediment on the survival of chinook salmon eggs in the Yukon Territory, it was determined that survival ranged between 86% and 91%

when eggs were exposed to 10% fine sediments. Survival decreased to <35% when eggs were exposed to 40% fine sediments (Seakem Group Ltd. 1992). Chapman (1988) undertook a critical review of the "variables used to define the effects of fines in redds of large salmonids" and described the effects, both primary and secondary, that the intrusion of fine sediment may have on the survival and emergence of embryos and alevins within and external to the "egg pocket." Chapman (1988) comments that "survival to emergence usually relates negatively to percentages of small fines."

Inter-gravel dissolved oxygen

Aside from the physical barrier to emergence that the intrusion of sediment can create in redds, the reduction in permeability, and consequent reduction in dissolved oxygen in inter-gravel water, can reduce the survival of eggs, embryos and alevins. Chapman (1988) concluded that "any reduction in dissolved oxygen levels from saturation probably reduces survival to emergence or post-emergent survival." If this is correct, and recognizing the high oxygen requirement to meet the metabolic demands of developing alevins (Rombough 1988), the suggested guideline of the minimum dissolved oxygen level of 6.5 $mg \cdot L^{-1}$ in inter-gravel waters for the protection of aquatic life (CCME 1999) is inadequate. Perhaps some of the most relevant research that pertains to the effects of low levels of dissolved oxygen (and at different temperatures) is that reported by Rombough (1988) in relation to steelhead trout embryo and alevin development. Salmonid embryos and alevins are able to maintain a stable metabolic rate that is independent of dissolved oxygen concentrations down to a critical level after which their metabolic rate becomes directly dependent on available oxygen. These critical dissolved oxygen levels tend to increase with increasing metabolic rate during the embryonic period, with temperature and with tissue mass. Dissolved oxygen levels below the critical level have a detrimental impact. Numerous sublethal effects have been documented, including retarded development (Garside 1959), reduced growth (Silver et al. 1963; Shumway et al. 1964), premature hatch (Garside 1966), and premature emergence (Bailey et al. 1980). As concluded from Rombough's research, the critical dissolved oxygen levels to satisfy normal respiration at the different developmental stages vary widely, but are higher at higher temperatures. Maximum critical dissolved oxygen values occurred just before hatch and were 7.5 $mg \cdot L^{-1}$, 8.9 $mg \cdot L^{-1}$, 9.6 $mg \cdot L^{-1}$, and 9.7 $mg \cdot L^{-1}$ (representing 60%, 77%, 89%, and 96% of air saturated values respectively), at corresponding temperatures of 6 °C, 9 °C, 12 °C, and 15 °C. Higher dissolved oxygen levels would be required to protect the more sensitive individuals of the population tested, for the results presented here are the mean values.

Streambed habitat

In addition to the effects of deposited sediments on the reproductive success of salmonids, concern is also warranted over the maintenance of spaces among substrate components that are utilized by juvenile fish and other organisms. The nocturnal habit of certain species of juvenile salmonids during winter (Emmet et al. 1996; M. Bradford, Fisheries and Oceans, Canada, personal communication), allied to their use of inter-cobble substrate habitat, emphasizes the need to prevent streambeds from becoming impacted and the interstitial spaces filled with fine sediment (Cunjak 1996). Bjornn et al. (1977) found that fewer juvenile salmonids remained in artificial streams where sediment was added to pools (the interstices between large rocks in pools provided essential cover necessary to maintain large densities of fish). Similarly, Griffith and Smith (1993), during their examination of the use of "winter concealment cover by juvenile cutthroat trout and brown trout," found cobble and boulder habitat that was heavily embedded with fine sediment contained fewer juveniles of either species relative to substrates that were not so embedded.

For these reasons, and those documented by Chapman (1988), it has been considered important to restrict the deposition of sediment into salmonid spawning and rearing areas, a topic mentioned above in relation to water quality criteria (BCMELP 1998; CCME 1999).

AQUATIC PRODUCTIVITY

Effects of sediment on primary productivity and consequential effects on higher trophic levels

Lloyd et al. (1987) showed that the productivity of aquatic systems could be reduced by turbid conditions. Increases in turbidity reduced light penetration in lakes and streams (Lloyd 1987; Lloyd et al. 1987), which led to decreased plant biomass and hence reduced primary production, decreased abundance of fish food organisms (secondary production) and decreased production and abundance of fish. They also determined that the compensation depth showed a strong inverse relationship with turbidity in 14 Alaskan lakes, and these results were comparable with those for other lakes referenced in their document. The compensation depth is sensitive to small changes in turbidity, which is directly related to primary production. Because of this effect and its concomitant effect on higher trophic levels, Lloyd et al. (1987) examined changes in seasonal zooplankton densities and found that those in glacially turbid lakes were as little as 5% of those in clear-water lakes. In addition, they found that zooplankton density diminished with decreasing compensation depth. Of eight glacially turbid lakes they examined, none had populations of cladocera, a group of highly favored food organisms of juvenile salmonids. Lloyd et al. (1987) then assessed the production of fish from different lakes and determined that the yield of juvenile sockeye salmon was related to the magnitude of change in the euphotic volume, caused in some cases by increased turbidity, and the resultant decreases in primary and secondary production. Other references, cited by these authors, substantiate the relationship between turbidity and biological productivity.

In streams, as for lakes, there appears to be a close correspondence between turbidity and reduced light penetration (Van Nieuwenhuyse and LaPerriere 1986): Van Nieuwenhuyse (1983) found that light extinction was related to increased mining-induced turbidity. Based on these studies it was calculated that a turbidity of 5 NTUs (equivalent to about 15 to 20 $mg \cdot L^{-1}$) could decrease the primary productivity of shallow, clear-water streams by about 3% to 13%, and that an increase of 25 NTUs (75 $mg \cdot L^{-1}$ to 100 $mg \cdot L^{-1}$) may decrease primary production by 13% to 50%. Alabaster and

Lloyd (1982), recognized that in addition to the effects of suspended sediment on algae being related to its effect on light penetration, in conjunction with elevated flow rates, algae may be scoured from the streambed substrates.

Lloyd et al. (1987) observed that the densities and biomass of benthic invertebrates in un-mined (placer gold) streams were significantly higher than in mined streams and that in heavily mined watersheds most taxa became rare or were eliminated (Wagener and LaPerriere 1985). Although the reduced densities of invertebrates were thought to have been a result of habitat alteration caused by "settleable solids" and increased fines in bottom substrates (LaPerriere et al. 1983; Wagener and LaPerriere 1985), it was concluded that turbidity was the strongest descriptor of reduced density and biomass of macroinvertebrates in the mined streams. However, the complexity of determining the factors responsible for changes in periphyton are emphasized by the studies of Shortreed and Stockner (1983). These authors examined such changes in Carnation Creek, a coastal stream in British Columbia that was affected by logging in its watershed. They concluded that environmental factors such as increased sediment loads had little effect on periphyton production because the major limiting factor was phosphorous concentration. In contrast, changes in the benthos and fish communities (Culp 1982; Holtby and Hartman 1982) were more attributable to changes in the physical environment rather than to changes in periphyton biomass or species composition.

Effects of sediment on benthic invertebrates

As mentioned above, invertebrate populations (secondary production) depend upon primary production, and that the latter also may be adversely affected by elevated levels of sediment. Certain benthic invertebrates are grazers and depend on periphyton for food, while others may be filter feeders which could have their feeding structures clogged by sediment thereby reducing feeding efficiency and reducing growth rates (Hynes 1970). Some direct effects of sediment on aquatic invertebrates include a) physical habitat change due to the scouring of streambeds and the dislodgment of individuals, b) smothering of benthic communities, c) clogging of the interstices between substrate components which affects microhabitat, and d) abrasion of respiratory surfaces and interference of food uptake for filter-feeders (Singleton 1985, cited by CCME 1999).

Benthic invertebrate drift

One early response of benthic invertebrates to increases in sediment is that of increased drift. From the results of both comparative and experimental approaches, Culp et al. (1986) revealed that as increases in fine sediments occurred, macroinvertebrate drift rates increased and, consequently, the density and species diversity of the benthic macroinvertebrate community was reduced. Similarly, Gammon (1970), found that increases in suspended solids of $<40\ mg \cdot L^{-1}$, and $>120\ mg \cdot L^{-1}$, caused a 25%, and 60% reduction respectively, in macroinvertebrate populations below a limestone quarry.

Culp et al. (1986) examined the benthic ecology of Carnation Creek, BC, and considered that drift was a function of the microhabitat preference of the invertebrate taxa (those on the substrate responded quicker than those occupying the interstitial spaces of the substrate, which exhibited a delayed response). These authors state that the partial or complete filling of the interstices can affect benthic macroinvertebrates residing in the stony substrate through the reduction of intra-substrate current velocity and the related dissolved oxygen content, as well as restricting the particle size range and the deposition of detritus in the substrate (Rosenberg and Snow 1975). In experiments conducted to assess the response of benthic macroinvertebrate communities to additions of fine sediments to riffles, Culp et al. (1986) found no measurable impact on five of the six most numerically dominant taxa, and the only negative effect was higher drift rates and lower benthic densities of one taxonomic group. However, the physical transport of the sediment caused a disturbance that reduced total benthic densities by >50% in 24 h, and significantly influenced macroinvertebrate community composition [these changes were the result of "catastrophic drift initiated by fine sediments that slid and bounced along the surface of the stony substrate (saltation), and distinct immediate and delayed responses to diurnal drift"]. Taxa that responded immediately to the sediment movement resided predominantly at the substrate surface and were instantaneously exposed to scouring as sediments were added.

Another example of changes in benthic communities due to drift is provided by Young and Mackie (1991) who examined the responses of stream communities during the construction of a pipeline crossing in the Northwest Territories. As total sediment concentrations increased from <2 $mg \cdot L^{-1}$ to >300 $mg \cdot L^{-1}$ and peaked at above 3,000 $mg \cdot L^{-1}$, there was a concurrent increase in benthic invertebrate drift. These effects lasted up to five weeks after disturbance, but after spring ice break up and the associated increases in stream flows, there was no significant impact observed. Young and Mackie (1991) thus concluded that the impact on the benthic invertebrates of a natural perturbation was greater than the effect of the pipeline construction. This conclusion illustrates the temporal nature of certain anthropogenic activities. This event, and the seasonal cycle of increased flows and the associated dispersion of aquatic organisms, were, however, separated by time. At the time of construction the effects were an unnatural perturbation, outside of typical seasonal changes, with the potential, albeit short-term (months), to impact individual aquatic organisms.

Drift is the major means by which aquatic insects colonize natural and altered streams, and the distances drifted is variable and associated with species, substrate type, current velocity and other stream characteristics (Luedtke and Brusven 1976). Furthermore, the upstream migration of insects has been reported to be from 5% to 30% of downstream drift and could serve to assist in the colonization of, for example, rehabilitated reaches of a stream (Bishop and Hynes 1969; Brusven 1970; Elliott 1971). In experiments carried out by Luedtke and Brusven (1976), it was determined that many common riffle insects were unable to move upstream on sand substrates. The authors state that pebble and cobble may be necessary for the upstream movement even at low current velocities, and that a combination of exposure to current and to the instability of sand grains was responsible for restricting upstream movement (as occurs in streams having "heavy" sand deposition).

Substrate embeddedness

The degree to which the major stream substrate particles are surrounded by fine material (the degree of embeddedness) was found to have a strong correlation with macroinvertebrate assemblage richness and composition (Richards and Host 1994, cited by CCME 1999). Many of the organisms that are favored as food items (e.g. mayflies, caddisflies and stoneflies) by stream-dwelling fish prefer relatively coarse streambed substrates and are harmed by intrusions of fine sediments (Everest et al. 1986), while others (e.g. midges) are considered to be more tolerant (Nuttall 1972). However, in a study of the diet of under-yearling Arctic grayling in a watershed receiving elevated sediments generated from placer gold mining operations, Birtwell et al. (1984) found that chironomid larvae were predominant food items and that they were most numerous in drift samples from the creek subjected to mining and sediment input. Bjornn et al. (1977), found that in artificial stream channels benthic insect density in fully sedimented riffles (>66% embeddedness) was half of that in un-sedimented riffles, but the abundance of drifting insects was not significantly smaller. In a natural stream riffle cleaned of sediment, benthic insects were 1.5 times more abundant and mayflies and stoneflies were four and eight times more abundant respectively. These authors recorded an increase in abundance of benthic invertebrates at lower levels of substrate embeddedness. Additional examples are available in the literature that reveal the negative effect of sediment deposition and embeddedness on benthic invertebrates (refer to Golder Associates 1999).

Examples of anthropogenic activities and the associated impacts of elevated levels of sediment

Increased water flows and the draining of impoundments

The draining of impoundments produces negative effects on aquatic communities by, in part, elevating sediment levels. For example, Langer (Fisheries and Oceans Canada, Vancouver, unpublished data; personal communication) reports that the flushing of British Columbia's Shuswap Falls dam in 1970, resulted in the elevation of suspended sediments. The "stressing" of whitefish, trout, and chinook salmon, and the clogging of gills was evident at suspended sediment levels of 10,000 mg·L^{-1}.

In 1991, water was spilled from the Terzaghi Dam, in BC, into the Bridge River. Suspended sediment levels rose from 6 mg·L^{-1} to 15,900 mg·L^{-1} within a matter of hours 10 km downstream from the dam. Similarly, at a location 30 km further downstream from the release of water, suspended sediment levels rose from 46 mg·L^{-1} to 3,590 mg·L^{-1}. The effect of these increases in flow resulted in significant removal of salmon spawning substrate, habitat change and erosion. Unfortunately, the effects on the aquatic organisms were not quantified. Internationally, there are similar observations. Hesse and Newcombe (1982) report that in Nebraska, the elevation in suspended sediment was generally up to 10,000 mg·L^{-1}, but reached a maximum value of 21,875 mg·L^{-1} during the flushing of the Spencer Dam. During the flushing, fish kills were reported on four occasions between 1975 and 1979, and 30 species of fish were affected and 22,471 killed. Palmer and O'Keefe (1990) reported that the draining of the Elandsdrift Dam, South Africa, resulted in suspended sediment levels of 2,829 mg·L^{-1}, killing fish for 3.5 km downstream, and depleting benthic invertebrate population densities for over 200 km (the latter recovering within two months). Doeg and Koehn (1994) report the effects of the draining of the Upper Yarra Dam, Australia, which elevated suspended sediment levels to 4,610 mg·L^{-1}, reduced fish abundance from 90% to 59% 2.7 km from the dam, and reduced benthic macroinvertebrates, whose recovery took between 45 days and seven months.

These examples illustrate the potential for the release of water to elevate suspended sediment levels and with it the concomitant impact on different trophic levels. However, caution should be exercised over concluding the cause of the biological changes that were observed. Garrick et al. (1990) mention that factors in addition to suspended sediment (e.g. reduced levels of dissolved oxygen, increases in ammonia) are often associated with the draining of dams, and that these factors may cause cumulative impacts or act synergistically.

Placer gold mining

In a study on the effects of sediment from placer mining operations on a number of streams in the Yukon Territory, it was deduced that taxonomic diversity, density, and biomass of benthic macroinvertebrates was reduced (Seakem Group Ltd. 1992). Densities of benthic invertebrates were reduced by factors of up to 118 to 204 times at sites where mean concentrations of suspended sediment exceeded a threshold in the range of 50 mg·L^{-1} to 175 mg·L^{-1} relative to less impacted streams. Seakem Group Ltd. (1992) concluded that in areas with high suspended sediment concentrations, the suspended sediment threshold for major effects appeared to be in the range of 25 mg·L^{-1} to 100 mg·L^{-1}; the lowest concentration to have an observable effect was <75 mg·L^{-1}. This result supports the findings of other researchers who have examined the effects of elevated levels of sediment from placer mining operations on aquatic organisms (Mathers et al. 1981; Laperriere et al. 1983; Soroka and McKenzie-Grieve 1984; Wagener and LaPerriere 1985; Weber 1986).

The effect of changes to stream integrity that were recorded by Seakem Group Ltd. (1992) for benthic invertebrates was also found in relation to fish. Un-mined creeks with turbidities of about 23 NTUs supported a standing stock of fish 40 times that of placer-mined streams with turbidities of 440 to 465 NTUs (about 500 mg·L^{-1}). Overall, streams that had seasonal suspended sediment concentrations in excess of 50 mg·L^{-1} did not support "significant" numbers of under-yearling Arctic grayling or chinook salmon.

From the results of their field studies, Seakem Group Ltd. (1992) concluded the threshold for direct and indirect effects of sediment from placer mining on Arctic grayling and chinook salmon was <75 mg·L^{-1} to 130 mg·L^{-1}. These authors also stated that "settleable solids" standards for this industry were inappropriate because the corresponding suspended sediment levels exceeded the thresholds for adverse effects on fish and their food resources.

SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

It is concluded that elevated levels of sediment (typically over background) may be harmful to fish (e.g. acutely lethal, or elicit sublethal responses that compromise their well-being and jeopardize survival), and in addition, negatively impact on their habitat. Lethal levels of sediment, determined through laboratory experimentation over different durations of exposure, typically range from hundreds to hundreds of thousands of $mg \cdot L^{-1}$ suspended sediment, whereas sublethal effects are typically manifest in the tens to hundreds of $mg \cdot L^{-1}$ suspended sediment. Some species of aquatic organisms are more tolerant of suspended and deposited sediment than others (Lloyd 1987; Newcombe and MacDonald 1991; Caux et al. 1997), and this variation must be recognized when assessing potential effects.

Although elevated levels of suspended sediment elicit adverse responses in individual aquatic organisms, it is difficult to extrapolate effects to the population or ecosystem levels. However, the biological productivity of turbid systems has been shown to be less than that of non-turbid systems (Lloyd et al. 1987). Anthropogenic activities, such as some placer mining operations, have resulted in lowered densities of aquatic organisms in watersheds through the elevation of suspended and deposited sediments (refer to Lloyd et al. 1987; Seakem Group Ltd. 1992).

Criteria, guidelines and recommendations, though having been formulated by different agencies, all tend to be mutually supportive. At the same time they have application limitations, especially relating to the protection of aquatic organisms from the effects of sediment concentrations in the ≤ tens of $mg \cdot L^{-1}$. Application of the criteria must be done while recognizing potential impacts on aquatic organisms at both the lethal and the sublethal level. Particle size and nature of the sediment must be considered as well. Bioassay information that reveals the lethal effect of sediment over a short period of time (such as 96 h), provides only a coarse indication of the effects of elevated levels of sediment in the wild. Accordingly, and when available, the more appropriate criteria which incorporate sublethal and lethal effects knowledge should be used.

Criteria documented in this report that are based on suspended sediment levels are appropriate and endorsed for use.

Recent guidelines have related elevated sediment levels to the natural hydrological regimes in streams and the associated variation in suspended sediment concentrations (CCREM 1987; BCMELP 1998; CCME 1999). In addition, the use of risk criteria in relation to the elevation of sediment concentrations above background (Government of Canada 1993) have merit and are supportive of, and based on, earlier published criteria.

Models that utilize sediment concentration and duration-of-exposure to predict harm (Newcombe and MacDonald 1991; Newcombe and Jensen 1996; Caux et al. 1997; BCMELP 1998; CCME 1999) reveal significant trends in increasing harm to fish and other aquatic organisms with increasing duration of exposure. Such trend identification is of value in predicting the potential effects of sediment on aquatic organisms, but caution must be exercised when assessing the effects of low concentrations (≤ tens of $mg \cdot L^{-1}$) of suspended sediment over protracted periods of time. Furthermore, it is likely that there would be increased variation in the response among individuals and life stages of organisms to the effects of elevated, but lower and sublethal levels of sediments, relative to less variable responses at higher sediment levels, due to a greater severity of effect and less scope for adaptation, tolerance and resistance. Because of this, judicious application of these models is warranted when assessing the potential impacts of exposure to low levels of suspended sediment.

Criteria that rely solely on the use of turbidity to protect aquatic organisms from elevated levels of suspended solids (USEPA 1986; Lloyd 1987) are not generally recommended for use because the typical site-specific and highly significant relationship that exists between turbidity (NTUs) and suspended solids is not universally applicable. Turbidity determinations integrate the effects of suspended and dissolved material on the penetration of light in waters which, in turn, affects biological productivity (Lloyd et al. 1987). Depending upon the nature of the sediment in suspension and of the dissolved material, the relationship between turbidity and suspended sediment will vary, usually by a factor less than 10. If, however, the relationship between suspended sediment concentrations and turbidity is known for a particular area (watershed, stream, reach etc.), then turbidity *per se* may be used as a surrogate for suspended sediment and the appropriate criteria for the latter applied. Turbidity is, therefore, a useful, but approximate, indicator for suspended sediment.

Because of the slow settling time for fine silts and clays and some organic particles, the volumetric determination of "settleable solids" may not accurately reflect the amount of sediment that a water sample may contain. Also, the practical detection limit (0.2 $mL \cdot L^{-1}$) is high, and most natural stream waters have less than this value for most of the year (Seakem Group Ltd. 1992). Furthermore, strong relationships between "settleable solids" and suspended sediment tend to be difficult to obtain, thus making this determination a much less useful measure of solids in a sample and, accordingly, of lower value for assessing effects on biota than the use of suspended sediment and turbidity upon which much research has been carried out.

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REFERENCES

Alabaster J.S., and R. Lloyd. 1982. Finely divided solids. pp 1-20 In: J.S. Alabaster and R. Lloyd (eds.) Water quality criteria for freshwater fish. Butterworth Publishing Co., London.

Anderson, P. B. Taylor, and G. Balch. 1996. Quantifying the effects of sediment release on fish and their habitats. Can. Manuscr. Rep. Fish. Aquat. Sci. 2346. 110p.

Bailey, J.E., S.D. Rice, J.J. Pella, and S.G. Taylor. 1980. Effects of seeding density of pink salmon, *Oncorhynchus gorbuscha*, eggs on water chemistry, fry characteristics, and fry survival in gravel incubators. Fish. Bull. 78:649-658.

British Columbia Ministry of Environment, Lands, and Parks (BCMELP). 1998. Ambient water quality guidelines (criteria) for turbidity, suspended and benthic sediments: overview. Water Management Branch, Environment and Resource Division, British Columbia Ministry of Environment, Lands and Parks. Victoria, BC.

Birtwell, I.K., G. Hartman, B. Anderson, D.J. McLeay, and J.G. Malik. 1984. A brief investigation of Arctic grayling (*Thymallus arcticus*) and aquatic invertebrates in the Minto Creek drainage, Mayo, Yukon Territory. Can. Tech. Rept. Fish. Aquat. Sci. 1287.

Birtwell, I.K., G.L. Greer, M.D. Nassichuk, and I.H. Rogers. 1983. Studies on the impact of municipal sewage discharged onto an intertidal area within the Fraser River estuary, British Columbia. Can. Tech. Rep. Fish. Aquat. Sci. 1170. 55p.

Birtwell, I.K., J.S. Korstrom, M. Komatsu, B.J. Fink, L.I. Richmond, and R.P. Fink. 1999. The susceptibility of juvenile chum salmon (*Oncorhynchus keta*) to predation following sublethal exposure to elevated temperature and dissolved gas supersaturation in seawater. Can. Tech. Rep. Fish. Aquat. Sci. (In preparation).

Bishop, J.E., and H.B.N. Hynes. 1969. Upstream movements of benthic invertebrates in the Speed River, Ontario. J. Fish. Res. Board Can. 26:279-298.

Bjornn, T.C., M.A. Brusven, M.P. Molnau, J.H. Milligan, R.A. Klamt, E. Chacho, and C. Schaye. 1977. Transport of granitic sediment in streams and its effects on insects and fish. College of Forestry, Wildlife and Range Sciences, University of Idaho. Publication 17. 53p.

Brusven, M.A. 1970. Drift periodicity and upstream dispersion of stream insects. J. Entomol. Soc. BC. 67:48-59.

Canadian Council of Resource and Environment Ministers (CCREM). 1987. Total suspended solids. In: Canadian Environmental Quality Guidelines, Environment Canada, Ottawa, ON.

Canadian Council of Ministers of the Environment (CCME). 1999. Canadian water quality guidelines for the protection of aquatic life: total particulate matter. In: Canadian Environmental Quality Guidelines, 1999, Environment Canada, Winnipeg, Manitoba. (In preparation).

Caux, P.-Y., D.R.J. Moore, and D. MacDonald. 1997. Ambient water quality guidelines (criteria) for turbidity, suspended and benthic sediments: technical appendix. Prepared for the British Columbia Ministry of Environment, Lands and Parks. Water Quality Branch, Environment and Resource Management Division, Victoria, BC. 82p.

Chapman, D.W. 1988. Critical review of variables used to define effects of fines in redds of large salmonids. Trans. Amer. Fish. Soc. 117:1-21.

Culp, J. M. 1982. The ecological effects of logging on the macroinvertebrates of a salmon stream in British Columbia. Ph.D. thesis. University of Calgary, AB. 157p.

Culp, J.M., F.J. Wrona, and R.W. Davies. 1986. Response of stream benthos and drift to fine sediment deposition versus transport. Can. J. Zool. 64:1345-1351.

Cunjak, R.A. 1996. Winter habitat of selected stream fishes and potential impact from land-use activities. Can. J. Fish. Aquat. Sci. 53 (Supplement 1):267-282.

Doeg, T.G., and J.D. Koehn. 1994. Effects of draining and desilting a small weir on downstream fish and macroinvertebrates. Regulated Rivers: Research and Management. 9:263-277.

Elliot, J.M. 1971. Upstream movements of benthic invertebrates in a Lake District stream. J. Anim. Ecol. 40:235-252.

Emmett, B., G.M. Kruzynski, and I.K. Birtwell. 1996. The use of pulp mill effluent mixing zone during winter by juvenile chinook salmon in the Fraser River at Prince George, British Columbia. Can. Manuscr. Rep. Fish. Aquat. Sci. 2341: 38p.

European Inland Fisheries Advisory Commission (EIFAC). 1964. Water quality criteria for European freshwater fish. Report on Finely Divided Solids and Inland Fisheries. European Inland Fisheries Advisory Commission. Food and Agriculture Organization of the United Nations. Rome. EIFAC/1. 21p.

Everest, F.H., R.L. Beschta, J.C. Srivener, K.V. Koski, J.R. Sedell, and C.J. Cederholm. 1986. Fine sediment and salmonid production: a paradox. USDA Forest Service, Corvallis, OR. 104p.

Farrell, A.P., A.K. Gamperl, and I.K. Birtwell. 1998. Prolonged swimming, recovery and repeat swimming performance of mature sockeye salmon *Oncorhynchus nerka* exposed to moderate hypoxia and pentachlorophenol. J. Exp. Biol. 201:2183-2193.

Gammon, J.R. 1970. The effect of inorganic sediment on stream biota. Water Pollution Control Research Series. Environmental Protection Agency, Water Quality Office, Washington, DC. 142p.

Garrick, J., B. Migeon, and E. Vindimian. 1990. Lethal effects of draining on brown trout. A predictive model based on field and laboratory studies. Wat. Res. 24:59-65.

Garside, E.T. 1959. Some effects of oxygen in relation to temperature on the development of lake trout embryos. Can. J. Zool. 37:689-698.

Garside, E.T. 1966. Effects of oxygen in relation to temperature on the development of embryos of brook trout and rainbow trout. J. Fish. Res. Board Can. 23:1121-1134.

Golder Associates Ltd. 1999. Development of tools to predict the effect of sediment deposition on fish habitat in lotic systems. Prepared for Fisheries and Oceans Canada. Calgary, AB. 168p.

Government of Canada. 1993. The Yukon Placer Authorization and supporting documents, applicable to placer mining in the Yukon Territory. Ottawa, ON. 36p.

Gregory, R.S. 1993. Effect of turbidity on the predator avoidance behavior of juvenile chinook salmon (*Oncorhynchus tshawytscha*). Can. J. Fish. Aquat. Sci. 50:241-246.

Gregory, R.S., and T.G. Northcote. 1993. Surface, planktonic, and benthic foraging by juvenile chinook salmon (*Oncorhynchus tshawytscha*) in turbid laboratory conditions. Can. J. Fish. Aquat. Sci. 50:233-240.

Gregory, R.S., J.A. Servizi, and D.W. Martens. 1993. The effects of suspended sediment on salmonids: a critical examination of the Stress Index. North Amer. J. Fish. Management. 13:868-873.

Griffith, J.S., and R.W. Smith. 1993. Use of winter concealment cover by juvenile cut-throat trout and brown trout in the South Fork of the Snake River, Idaho. North Amer. J. Fish. Management. 13:823-830.

Hebert, D.W.M., and J.C. Merkens. 1961. The effect of suspended mineral solids on the survival of trout. Int. J. Air Water Pollut. 5:56-75.

Hesse, L.W., and B.A. Newcombe. 1982. Effects of flushing Spencer Hydro on water quality, fish, and insect fauna in the Niobrara River, Nebraska. North Amer. J. Fish. Management. 2:45-52.

Hollis, E.H., J.G. Boone, C.R. DeRose, and G.J. Murphy. 1964. A literature review of the effects of turbidity and siltation on aquatic life. Department of Chesapeake Bay Affairs, Annapolis, Maryland. 26p.

Holtby, L.B., and G.F. Hartman. 1982. The population dynamics of coho salmon (*Oncorhynchus kisutch*) in a west coast rain forest stream subjected to logging. pp 308-347, In: G. Hartman (ed.) Proceedings of the Carnation Creek workshop, a 10-year review. Pacific Biological Station, Nanaimo, BC.

Hynes, H.B.N. 1970. The ecology of running waters. Liverpool University Press. Liverpool, England.

Jain, K.E., I.K. Birtwell, and A.P. Farrell. 1998. Repeat swimming performance of mature sockeye salmon following a brief recovery period: a sensitive measure of fish health and water quality. Can. J. Zool. 76:1-9.

Kruzynski, G. M., and I.K. Birtwell. 1994. A predation bioassay to quantify the ecological significance of sublethal responses of chinook salmon to the antisapstain fungicide TCMTB. Can. J. Fish. Aquat. Sci. 51:1780-1790.

Lake, R.G., and S.G. Hinch. 1999. Acute effects of suspended sediment angularity on juvenile coho salmon (*Oncorhynchus kisutch*). Can. J. Fish. Aquat. Sci. 56:862-867.

LaPerriere, J.D., D.M. Bjerklie, R.C. Simmons, E.E. Van Nieuwenhuyse, S.M. Wagener, and J.B. Reynolds. 1983. Effects of gold placer mining on interior Alaskan stream ecosystems. pp (12)12-34 In: J.W. Aldrich (ed.) Managing water resources for Alaska's development, proceedings. University of Alaska, Institute of Water Resources Report IWR-105, Fairbanks, AK.

Lloyd, D.S. 1985. Turbidity in freshwater habitats of Alaska: a review of published and unpublished literature relevant to the use of turbidity as a water quality standard. Rep. 85-1. Alaska Dept., Fish and Game. Juneau, AK. 101p.

Lloyd, D.S. 1987. Turbidity as a water quality standard for salmonid habitats in Alaska. North Amer. J. Fish. Management. 7:34-45.

Lloyd, D.S., J.P. Koenings, and J.D. LaPerriere. 1987. Effects of turbidity in fresh waters of Alaska. North Amer. J. Fish. Management. 7:18-33.

Luedtke, R.J., and M.A. Bruvsen. 1976. Effects of sand sedimentation on colonization of stream insects. J. Fish. Res. Board Can. 33(9):1881-1886.

Mace, P.M. 1983. Bird predation on juvenile salmonids in the Big Qualicum Estuary, Vancouver Island. Can. Tech. Rep. Fish. Aquat. Sci. 1176:79 p.

Martin, D.J., L.J. Wasserman, R.P. Jones, and E.O. Salo. 1984. Effects of Mount St. Helens eruption on salmon populations and habitat in the Toutle River. University of Washington, WA. Fish. Res. Inst. FRI-UW-8412 147 p.

Mathers, J.S., N.O. West, and B. Burns. 1981. Aquatic and wildlife resources of seven Yukon streams subject to placer mining. Can. Dept., Fisheries and Oceans, Dept., Indian and Northern Affairs, and Dept., of Environment, Whitehorse, Yukon. 183p.

McLeay, D.J., I.K. Birtwell, G.F. Hartman, and G.L. Ennis. 1987. Responses of Arctic grayling (*Thymallus arcticus*) to acute and prolonged exposure to Yukon placer mining sediment. Can. J. Fish. Aquat. Sci. 44: 658-673.

Mesa, M.G., T.P. Poe, D.M. Gadomski, and J.H. Petersen. 1994. Are all prey created equal? A review and synthesis of differential predation on prey in substandard condition. J. Fish. Biol. 45(Suppl. A): 81-96.

Milliman, J.D. 1980. Sedimentation in the Fraser River and its estuary, south-western British Columbia (Canada). Estuarine and Coastal Marine Science. 10:609-633.

Newcombe, T.W., and T.A. Flagg. 1983. Some effects of Mt. St. Helens volcanic ash on juvenile salmon smolts. Mar. Fish. Review. 45(2): 8-12.

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

Newcombe, C.P., and D.D. MacDonald. 1991. Effects of suspended sediment on aquatic ecosystems. North Amer. J. Fish. Management. 11:72-82.

Noggle, C.C. 1978. Behavior, physiological and lethal effects of suspended sediment on juvenile salmonids. MSc. thesis, University of Washington, Seattle, WA. 87p.

Nuttall, P.M. 1972. The effects of sand deposition upon the macro-invertebrate fauna of the River Camel, Cornwall. Freshwater Biol. 2:81-186.

Palmer, R.W., and J.H. O'Keeffe. 1990. Downstream effects of a small impoundment on a turbid river. Arch. Hydrobiol. 119:457-473.

Reiser, D.W., and R.G. White. 1988. Effects of two sediment size-classes on survival of steelhead and chinook salmon eggs. North Amer. J. Fish. Management. 8:432-437.

Richards, C., and G. Host. 1994. Examining land use influences on stream habitats and macroinvertebrates: a GIS approach. Water Resour. Bull. 30(4):729-738.

Rombough, P.J. 1988. Growth, aerobic metabolism, and dissolved oxygen requirements of embryos and alevins of steelhead, *Salmo gairdneri*. Can. J. Zool. 66:651-660.

Rosenberg, D.M., and N.B. Snow. 1975. Ecological studies of aquatic organisms in the McKenzie and Porcupine River drainages in relation to sedimentation. Tech. Rep. Fish. Mar. Serv. (Can). 547.

Scannell, P.O. 1988. Effects of elevated sediment levels from placer mining on survival and behavior of immature arctic grayling. MSc. thesis. University of Alaska Fairbanks, AK. 93p.

Scrivener, J.C., T.G. Brown, and B.C. Anderson. 1994. Juvenile chinook salmon (*Oncorhynchus tshawytscha*) utilisation of Hawks Creek, a small and non-natal tributary of the upper Fraser River. Can. J. Fish. Aquat. Sci. 51:1139-1146.

Seakem Group Ltd. 1992. Yukon Placer Mining Study. Volume 1 Executive Summary. Prepared for the Yukon Placer Mining Implementation Review Committee. Sidney, British Columbia. 17p.

Servizi, J.A., and R.W. Gordon. 1990. Acute lethal toxicity of ammonia and suspended sediment mixtures to chinook salmon (*Oncorhynchus tshawytscha*). Bull. Environ. Contam. Toxicol. 44:650-656.

Servizi, J.A., and D.W. Martens. 1987. Some effects of suspended Fraser River sediments on sockeye salmon (*Oncorhynchus nerka*). pp. 254-264 In: H.D. Smith, L. Margolis, and C.C. Wood (eds.) Sockeye salmon (*Oncorhynchus nerka*) population biology and future management. Can. Spec. Publ. Fish. Aquat. Sci. 96.

Servizi, J.A., and D.W. Martens. 1991. Effect of temperature, season, and fish size on acute lethality of suspended sediments to coho salmon (*Oncorhynchus kisutch*). Can. J. Fish. Aquat. Sci. 48:493-497.

Servizi, J.A., and D.W. Martens. 1992. Sublethal responses of coho salmon (*Oncorhynchus kisutch*) to suspended sediment. Can. J. Fish. Aquat. Sci. 49:1389-1395.

Shortreed, K.S., and J.G. Stockner 1983. Periphyton biomass and species composition in a coastal rainforest stream in BC: effects of environmental changes caused by logging. Can. J. Fish. Aquat. Sci. 40:1887-1895.

Shumway, D.L., C.E. Warren, and P. Doudoroff. 1964. Influence of oxygen concentration and water movement on the growth of steelhead trout and coho salmon embryos. Trans. Amer. Fish. Soc. 93:342-356.

Sigler, J.W., T.C. Bjornn, and F.H. Everest. 1984. Effect of chronic turbidity on density and growth of steelheads and coho salmon. Trans. Amer. Fish. Soc. 113:142-150.

Silver, S.J., C.E. Warren, and P. Doudoroff. 1963. Dissolved oxygen requirements of developing steelhead trout and chinook salmon embryos at different water velocities. Trans. Amer. Fish. Soc. 92:327-343.

Singleton, H.J. 1985. Water quality criteria for particulate matter: technical appendix. British Columbia Ministry of Environment, Lands, and Parks. Victoria, BC.

Soroka, I.K., and G. McKenzie-Grieve. 1984. A biological and water quality assessment at a placer mine on Barlow Creek, Yukon Territory. Regional Program Report 84-16. Environment Canada, Environmental Protection Service, Whitehorse, Yukon. 102p.

Stober, Q.J., B.D. Ross, C.L. Melby, P.A. Dinnel, T.H. Jagielo, and E.O. Salo. 1981. Effects of suspended volcanic sediment on coho and chinook salmon in the Toutle and Cowlitz Rivers. Technical Completion Report. University of Washington, WA. Fish. Res. Inst. FRI-UW-8124. 147p.

United States Environmental Protection Agency. (USEPA) 1986. Quality criteria for water. U.S. Department of Commerce, National Technical Information Service, Springfield, VA. 22161.

Vandenbyllaardt, L., F.J. Ward, C.R. Braekevelt, and D.B. McIntyre. 1991. Relationships between turbidity, piscivory, and development of the retina in juvenile walleyes. Trans. Amer. Fish. Soc. 120:382-390.

Van Nieuwenhuyse, E.E. 1983. The effects of placer mining on the primary productivity of interior Alaska streams. MSc. thesis, University of Alaska, Fairbanks, AK.

Van Nieuwenhuyse, E.E., and J.D. LaPerriere. 1986. Effects of placer gold mining on primary production in subarctic streams of Alaska. Water Resourc. Bull. 22:91-99.

Wagener, S.M., and J.D. LaPerriere. 1985. Effects of placer mining on the invertebrate communities of interior Alaska streams. Freshwater Invertebrate Biology. 4:208-214.

Waters, T.F. 1995. Sediment in streams: sources, biological effects and control. Amer. Fish. Soc. Bethesda, Maryland. 251p.

Weber, P.K. 1986. Downstream effects of placer mining in Birch Creek basin, Alaska. Technical Report 85-2. Alaska Dept., Fish and Game, Juneau, AK. 21p.

Wedemeyer, G.A., and D.J. McLeay. 1981. Methods for determining the tolerance of fish to environmental stressors. pp. 247-275 In: A.D. Pickering (ed.) Stress and Fish. Academic Press, London.

Wedemeyer, G.A., B.A. Barton, and D.J. McLeay. 1991. Stress and acclimation. pp. 451-489. In: C.B. Schreck and P.B. Moyle (eds.) Methods for fish biology. American Fisheries Society, Bethesda, Maryland.

Wedemeyer, G.A., F.P. Meyer, and L. Smith. 1976. Environmental stress and fish diseases. pp. 1-192. In: S.F. Snieszko, and H.R. Axelrod (eds.) Book 5. Diseases of fishes. T.H.F. Publications, Inc., Jersey City, New York.

Whitman R.P., T.P. Quinn, and E.L. Brannon. 1982. Influence of suspended volcanic ash on homing behavior of adult chinook salmon. Trans. Amer. Fish. Soc. 111:63-69.

Young, R.J., and G.L. Mackie. 1991. Effect of oil pipeline construction on the benthic invertebrate community structure of Hodgson Creek, Northwest Territories. Can. J. Zool. 69:2154-2160.