

SEDIMENT GENERATION FROM FORESTRY OPERATIONS AND ASSOCIATED EFFECTS ON AQUATIC ECOSYSTEMS

Paul G. Anderson

Alliance Pipeline Ltd., 400, 605 – 5 Avenue S.W. Calgary, Alberta, Canada, T2P 3H5
Phone (403) 716 0389 email – andersp@alliance-pipeline.com

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Abstract

Timber harvest operations have been shown to have many effects on adjacent watercourses and on the aquatic ecosystems they support. This may occur from introductions or loss of woody debris, loss of riparian vegetation, accelerated stream bank and bed erosion, the alteration of natural channel form and process, and the reduction of stream habitat diversity. However, the existing literature indicates one of the most insidious effects of logging is the elevation of sediment loads and increased sedimentation within the drainage basin.

Sediment generation from various forestry practices has been studied extensively in the past. Forestry practices which generate suspended sediments include all operations that disturb soil surfaces such as site preparations, clear-cutting, log skidding, yarding, slash burns, heavy equipment operation and road construction and maintenance. From these sources, construction, use and maintenance of logging roads located in close proximity to watercourses produce by far the highest levels of suspended sediment generation in streams.

Three aspects of logging road development and maintenance are known to elevate sediment loads in watercourses: 1) instream and near-stream construction operations; 2) reduction in retention time and associated increase in erosion in the drainage basin; and 3) mass soil movements and/or landslides associated with logging road design and placement.

This literature review examined the effects of increased sediment load and sedimentation on aquatic ecosystems emphasizing forestry operations that generate elevated sediment loads. The review included the effects of sediment on fish (behavioral, physiological and

population effects) and the effects of sedimentation on fish habitats (including spawning, rearing, food production, summer and overwintering habitats). A habitat effects relationship was presented which related the concentration and duration of specific sediment exposure events to the alteration of fish habitats. This relationship allows for post-disturbance evaluation of the potential effects on fish habitat.

Many advances have been made in the design and construction of logging roads over the past three decades. Many of these advances have been made in the design of road access systems, while other advances have been made in the development of more sophisticated mitigation techniques. These advances are described and discussed in relation to minimizing sediment generation and subsequent sedimentation in aquatic environments.

Introduction

Forest harvesting practices can elicit a number of physical changes within a watershed. These changes can set up associated responses in a wide range of physical, chemical and biological processes and can substantially alter aquatic habitats and communities. Forestry related activities can influence: stream hydrologic regime by reducing the time between peak rainfall and peak stream discharge and consequently increasing the magnitude of peak seasonal flows; water quality parameters such as temperature and sediment load; and, stream geometry by increasing erosive forces, channel migration, width to depth ratio and altering stream form and process (Meehan 1991; Salo and Cundy 1987)).

Forestry-related activities are not always harmful to freshwater fish communities. Long term and intensive study of logging on fish communities of Carnation Creek, Vancouver Island, British

Columbia, revealed the annual growth of juvenile coho salmon (*Oncorhynchus kisutch*) increased in years immediately following logging which was likely in response to increases in air and water temperatures (Hartman and Scrivener 1990; Holtby 1988; Tschaplinski and Hartman 1983). However, forestry activities have been shown to have adverse effects on resident and migratory fish communities, often related to an increase in the delivery of sediments to streams.

Although frequently viewed as synonymous, suspended sediments and sedimentation are two discrete processes that affect aquatic communities in different ways. Sediment transport in watercourses occurs in three forms, which include wash load, suspended, load and bed load. The entry of sediments to watercourses from upland sources is a natural process; however, when the rate entering a watercourse exceeds the capacity of the watercourse to transport or assimilate the sediment, stress may occur to the aquatic community and the suitability and/or productivity of aquatic habitats may be altered. Excess sediment in rivers and streams has been identified as the largest and most pervasive water pollution problem faced by aquatic systems in North America (Sweeten 1995).

This paper provides a collection of information related to biological responses to sediment synthesized from the literature and attempts to uncover relationships which are present between elevated sediment episodes and biological response. For additional information pertaining to this topic or to obtain additional references, the reader is directed to Meehan 1991, Salo and Cundy 1987, Newcombe (1994), Newcombe and Jensen (1995), Kerr (1995), Waters (1995) and Anderson *et al.* (1996).

Sources of Elevated Sediment Loads

Anthropocentric Sources of Elevated Sediments

Erosion within streams is a natural process and is affected by parameters such as stream flow, channel structure and stability, streambed composition, and disturbance within the watershed such as fire, landslide or ice scour. The disturbance of lands accelerates erosion and increases the delivery of sediment to stream systems. Any activity that is undertaken within a

watershed that disturbs land surfaces has the potential to increase sediment delivery to streams. Most of man's activities will increase erosion to some extent in forested watersheds. Man's activities which most frequently increase sediment loads in watercourses include agriculture, mining, forestry, urban development, and stream channel alteration such as dams and channelization, and instream construction associated with developments such as bridges, roads or pipeline and transmission crossings. This paper discusses the effects of sediment increase associated with episodic elevated sediment events in general, however, types of mitigation to minimize sediment load increases which are specific to the forestry industry are discussed in the Mitigation Measures section and sediment sources specific to forest harvest operations are discussed below.

Forestry Sediment Sources

Forestry operations frequently increase sediment delivery into streams. Logging operations have been shown to increase sediment production above natural sedimentation rates (Megahan and Kidd 1972). The activities which commonly result in increased sediment delivery include clear-cutting, skidding, yarding, site preparation for replanting and road construction, use and maintenance (Waters 1995).

Among forestry harvest activities, disturbance associated with logging road construction and operation produces the greatest sediment load increase (Waters 1995; Furniss *et al.* 1991; Cederholm *et al.* 1981). Roads associated with a jammer logging system in the Payette National Forest, Idaho, increased sediment production an average of approximately 750 times over the natural rate over a six-year period following construction (Megahan and Kidd 1972). The average erosion rate from roads on the jammer unit for 1.35 years preceding logging was 56.2 tons/mile²/day and the average rate for 4.8 years following logging was 51.0 tons/mile²/day (Megahan and Kidd 1972).

Effects of Elevated Sediment Loads

Sediment Effects on Fish

In response to changes in the environment, ecosystems often undergo changes in

community composition and structure. Organisms respond to environmental change in order to avoid or minimize effects on fitness. If an organism can not compensate for a change in the environment and suffers a reduction in fitness, the environmental change is termed a stress (Brett 1958; Kohen and Bayne 1989). Therefore, a stress limits either the rate of resource acquisition or growth and reproduction so that fitness is reduced (Grime 1989).

Stress has been defined as “the sum of all the physiological responses by which an animal tries to maintain or re-establish a normal metabolism in the face of a physical or chemical force” (Selye 1950). Stress occurs when the homeostatic or stabilizing processes of the fish or organism are extended beyond the capabilities of the organism to compensate for the biotic or abiotic challenges. Anthropogenic inputs of sediments into stream and riverine environments can cause stress to aquatic systems and thereby, directly and indirectly impact upon fish behaviour and health. Increased concentrations of suspended sediments can have direct effects on fish behaviour, fish physiology and fish populations (Anderson *et al.* 1996).

Behavioural Effects

Changes in fish behaviour are some of the first effects evoked from increasing concentrations of suspended sediments. Behavioural changes are generally considered benign and transitory. They are easily reversed and do not exhibit a long-lasting impact (Newcombe 1994). Typical responses include an increased frequency of the cough reflex, avoidance of suspended sediments, reduction in feeding and temporary disruption of territoriality. The severity of the behavioural response is associated with the timing of disturbance, the level of stress (and associated energy cost) and the importance of the habitat that the fish may be being excluded from.

The avoidance of suspended sediment plumes is one of the first reactions. Bisson and Bilby (1982) observed this behaviour evoked in juvenile coho salmon at total suspended sediment (TSS) concentrations as low as 88 mg/L. Similar results were recorded by McLeay *et al.* (1987) who found that Arctic grayling (*Thymallus arcticus*) avoided concentrations greater than 100 mg/L.

Increased concentrations of suspended sediment have also been correlated with a reduction in feeding. Feeding rate may be a function of prey visibility. McLeay *et al.* (1987) states that Arctic grayling exposed to suspended sediment concentrations greater than 100 mg/L were slower to recognize the food and more frequently missed a food item when they attempted to eat it. Sigler *et al.* (1984) believes, however, that a reduced feeding is more complex than reduced ability to see prey items, as many fish species (especially benthic feeders) do not use sight to identify prey items, but still exhibit reduced levels of feeding in response to elevated sediment loads.

High concentration of suspended sediments has also been associated with the loss of territoriality and interruption of movements of salmonids. Berg and Northcote (1985) found that territorial behaviour was lost at concentrations exceeding 30 NTU. They indicate that territoriality is influential in the allocation of food and habitat resources. Disruption to territoriality can occur when turbidity limits the visual distance that individuals can see, and when the downstream drift of fish avoiding increased concentrations of suspended sediment disrupts existing territories.

Physiological Effects

Physiological changes can be measured in fish as a response to the increased stress of suspended sediments. The typical measured responses include impaired growth, histological changes to gill tissue, alterations in blood chemistry, and an overall decrease in health and resistance to parasitism and disease. The effects of sediment exposure on each of these physiological effects are discussed below. When compared to sediment exposure events that elicit behavioural responses, longer exposure periods and/or higher concentrations are generally required before physiological responses are expressed. In this respect, physiological responses are more of a chronic effect. The effects are usually a graded response to increasing sediment dose. Impacts evoked from lower doses can be transitory, while those resulting from higher doses can be more lasting and severe.

Growth

An impaired growth rate is generally one of the more sensitive physiological responses to an

increase in suspended sediment concentration. Unlike behavioural responses, impaired growth generally requires a longer exposure period before effects are manifested. Sigler *et al.* (1984) found that growth was impaired in juvenile steelhead trout (*Oncorhynchus mykiss*) and coho salmon exposed to fire clay or bentonite clay at concentrations between 84 and 120 mg/L during a 14 to 21 day exposure period. Similar concentrations of 100 mg/L or greater were found to significantly impair growth in Arctic grayling under-yearlings (McLeay *et al.* 1987), largemouth bass (*Micropterus salmoides*), bluegill (*Lepomis macrochirus*), and redear sunfish (*Lepomis gibbosus*) (Buck 1956). However, growth impairment may be related more to the metabolic demands resulting from stress caused by increased suspended sediment than from a reduction in feeding. The time required before growth impairment was measurable ranged from a low of two weeks for juvenile steelhead trout and coho salmon to a high of six weeks for Arctic grayling under-yearlings (Sigler *et al.* 1984; McLeay *et al.* 1987).

Blood Chemistry

Alteration in blood chemistry resulting from the increased stress of suspended sediments have been found associated with concentrations ranging between 500 to 1500 mg/L (Redding and Schreck 1982; Servizi and Martens 1987). The changes most commonly recorded include an increase in haematocrite, erythrocyte count, hemoglobin concentration, and elevated blood sugar levels (hyperglycemia), plus decreases in blood chloride content, and depletion of liver glycogen (Wedemeyer *et al.* 1990; Servizi and Martens 1987). These increases coincide with the release of stress hormones (i.e., cortisol and epinephrine) and traumatization of the gill, and presumably represent a compensatory response to a decrease in gill function (Newcombe 1994). In addition, Sherk *et al.* (1973) found these changes to be associated with a reduction in the swimming endurance of white perch (*Morone americana*) exposed to 650 mg/L of TSS. Most of the observed changes resulted after four to five days of exposure (Newcombe 1994). Exceptions to this, however, were noticed by Redding and Schreck (1982) who found a significant increase in haematocrit volume within steelhead trout after only nine hours of exposure to 500 mg/L of volcanic ash, clay and topsoil.

Gill Trauma

Increased concentration of suspended sediments are known to physically traumatize gill tissue. The primary mechanisms of action is through physical abrasion of tissue and particle adsorption onto the gill. The types of tissue changes observed include swelling of secondary lamella and hypertrophy (cell swelling) of epithelial cells (Sherk *et al.* 1973); hyperplasia (increase in cell number) of gill tissue (Simmons 1984); and tissue necrosis (Servizi and Martens 1987).

The severity of damage appears to be related to the dose of exposure, as well as the size and angularity of the particles involved. Greater damage is typically observed with larger, more angular particles (Servizi and Martens 1991). These factors could account for the large range in responses seen for different exposure rates. For example, concentrations as low as 270 mg/L are known to cause gill damage in rainbow trout (*Oncorhynchus mykiss*) (after 13 days of exposure (Herbert and Merckens 1961) and yet McLeay *et al.* (1987) found no gill damage in young-of-the-year (YOY) Arctic grayling that were exposed to concentrations as high as 1300 mg/L; the duration of exposure was, however, only four days.

Secondary effects resulting from an infestation of parasitic protozoans were found in juvenile rainbow trout that were exposed to extremely high concentrations of suspended sediments. The trout were exposed to 4887 mg/L for a period of 16 days (Goldes 1983). This author did noted that the protozoan infection and gill architecture was found to be normal 58 days after the exposure ceased.

Resistance

Increased concentrations of suspended sediments have been associated with an overall decrease in the ability to defend against disease and to tolerate chemical toxins. For example, Herbert and Merken (1961) observed rainbow trout to be more susceptible to infestations of fin rot when fish were exposed for 121 days to concentrations of 270 mg/L of diatomaceous earth. Likewise, Servizi and Martens (1991) found a correlation between the prevalence of a viral kidney infection and an increased concentration of suspended sediments in coho salmon. When concentrations of suspended

sediments exceeded 100 mg/L, the tolerance of Arctic grayling to the toxicant pentachlorophenol (PCP) decreased (McLeay *et al.* 1987). This observation by McLeay *et al.* (1987) indicates a general decrease in tolerance to increased environmental stressors.

Phagocytosis

A process that may be closely linked to reduced resistance, is phagocytosis. Newcombe and Jensen (1995) discuss the process by which fine particles are enveloped by cells within fish gill and gut tissues and are transported to internal repository tissues. The main organ of repository in fish is the spleen (Newcombe and Jensen 1995). It is hypothesized that through this process, particles could reduce resistance to other stressors by impairing fish health. In addition, particles could trigger tumour induction, especially in circumstances where contaminants were adsorbed to particles in suspension (Newcombe and Jensen 1995).

Lethal Effects

Increased concentrations of suspended sediments and increased sedimentation rates have the potential to affect fish populations. The primary mechanisms of action are through increased egg mortality, reduced egg hatch, a reduction in the successful emergence of larvae, plus the sediment-induced death of juvenile and adult fish. These mechanisms are discussed below.

Egg Mortality

The primary cause of egg death is generally from burial by settled particles. Thin coverings (a few mm) of fine particles are believed to disrupt the normal exchange of gases and metabolic wastes between the egg and water. Sedimentation rates of 0.03 to 0.14 g dry weight sediment/cm² (i.e., 1-4 mm depth of silt and clay) significantly reduced the survival of lake whitefish (*Coregonus clupeaformis*) eggs (Fudge and Bodaly 1984). The effects upon egg mortality appear to be more closely related to the sedimentation of particles and less related to the concentration of suspended sediments. Zallen (1931) observed that concentrations of 1000 to 3000 mg/L had no effect upon the survival of mountain whitefish eggs (*Prosopium*

williamsoni). Campbell (1954), however, found 100 percent mortality in rainbow trout eggs exposed to TSS concentrations of 1000 to 2500 mg/L. Differences in egg mortality effects associated with elevated sediment loads is related to the size of the sediment particles involved and rates of sediment deposition.

In addition to the concentration of suspended sediments and the size of the particles involved, the duration of exposure appears to be key a factor in determining the effects of sediments on egg survival. Slaney *et al.* (1977) noticed that hatching success for rainbow trout was reduced after 2 months of exposure to 57 mg/L. A significant reduction in the hatching success of white perch and striped bass (*Morone saxatilis*) was observed in only 7 days after exposure to about 1000 mg/L TSS (Auld and Schubel 1978). The magnitude of the effect of sediment exposure may also be influenced by the timing of sediment exposure with respect to the stage of embryo development. The dose of sediment required to induce egg mortality is greatly influenced by the physical characteristics of the stream which, in turn, affect sediment transport capabilities and the capacity to maintain sediments in suspension or otherwise to result in their deposition.

Juvenile and Adult Fish Death

Juvenile and adult fish generally appear to be more resilient to stress from suspended sediments than other life history stages. Short term increases in TSS concentrations between 11000 and 55000 mg/L appear to be the point at which salmonid mortality significantly increases (Stober *et al.* 1981; Servizi and Martens 1987; Smith 1940). McLeay *et al.* (1983) reported survival of Arctic grayling subjected to moderately high concentrations (1000 mg/L) of fine grained materials (mining silt). Lloyd (1987), in a review of existing information, reported lethal effects to fish at concentrations ranging from 500 to 6000 mg/L. Sigler *et al.* (1984) reported mortality in young of the year coho salmon and steelhead trout at 500 to 1500 mg/L. Based on the information available on sediment and acute effects to fish, it is apparent that the severity of effect caused is a function of many factors which, in addition to concentration, duration particle size and life history stage, may include temperature, physical and chemical characteristics of the particles, associated toxicants, acclimatization, other stressors and

interactions of these and other factors (Waters 1995).

Habitat Effects

Habitat Exclusion and Habitat Alteration

In addition to the direct impacts of suspended sediments on fish, increases in sediment loads can also alter fish habitat or the utilization of habitats by fish (Scullion and Milner 1979, Lisle and Lewis 1992). High sediment loads can alter fish habitats temporarily by affecting water quality, making a stream reach unsuitable for use by fish. This habitat exclusion, if timed inappropriately, could have impacts on fish populations if the affected habitat is critical to the population during the period of elevated sediment load. This principle of habitat exclusion is very important one; however, this issue is separate from the issue of direct habitat alteration that will be discussed below.

Sediment episodes can have a prolonged effect on the suitability of habitats within a stream reach through increased levels of sedimentation. In fact, sedimentation is the single most important effect associated with sediment load increases, since sediment loads can alter the gross morphology of streams as well as the composition of the stream bed and associated habitats.

Changes in Stream Bed Porosity

Larger-sized materials, such as fine to coarse sand are quick to settle onto the stream bed. This material may accumulate on the surface of the stream bed or filter down into the inter-gravel spaces. Interstitial spaces can become clogged by the downward or, to a lesser extent, by the horizontal movement of sediment (Beschta and Johnson 1979).

Water movement through the stream bed materials is important for the benthic communities which reside there, and for the developing embryos of fish species who bury their eggs. Inter-gravel water movement is controlled by several hydraulic and physical properties of the stream and its bed. The permeability of the stream bed is determined by size composition of the substrate material, viscosity of the water (temperature dependent) and the packing of the substrate material (Stuart

1953; Cooper 1965). A small increase in the proportion of fine material can severely reduce the porosity and permeability of the gravel bed (Lisle and Lewis 1992) and the ability of alevins to receive adequate oxygen and emerge from the gravel.

Changes in Stream Morphology

In addition to altering stream bed composition, elevated sediment loads can also change channel geometry (Klein 1984). Elevated levels of sediment deposition can reduce the depth of pools and produce a net reduction in riffle areas. This accumulation of streambed deposits can reduce available habitat. For example, deposition of sediments in pools and other areas of instream cover can cause a decrease in the fish holding capacity of a stream reach (Bjornn *et al.* 1977). Smith and Saunders (1965) found that decreased brook trout (*Salvelinus fontinalis*) populations were related to infilling of available cover. Alexander and Hansen (1992) also noted that a decrease in sand bedload sediment was associated with an increase in rainbow trout and brown trout (*Salmo trutta*) populations. Changes in physical morphology of the stream can also inhibit the movement of fish or change the distribution of adult fish (Alabaster and Lloyd 1982).

Channels affected by sediment derived from Anthropogenic disturbance are also more transitory in nature. Fox (1974) found urban watersheds exhibited a 33% monthly change in geometry as compared to a 5% change in less disturbed rural drainages. Sediment material deposited within streams can be in constant motion as bedload transport slowly moves the deposited materials through the system. This material in motion can increase bed scour and bank erosion as the sediment increases the erosive force of the water, by creating a "sand blasting" effect.

Sedimentation Effects on Spawning Habitats

River spawning salmonids typically deposit their eggs in gravel beds commonly found in the upper reaches of river systems. For example, brown trout typically bury eggs in interstitial spaces of the substrata to depths of 9 to 12 cm (Scullion and Miller 1979). Alevins remain in the interstitial spaces until the start of exogenous feeding. The percolation of water through the

incubation substrate is an essential factor in determining the survival rate of incubating eggs (Lisle and Lewis 1979).

An increase in percent of fine material in the stream bed can have impacts on egg survival rates (Shaw and Maga 1943; Cordone and Kelley 1961) since it reduces streambed permeability. Lowered permeability reduces the interchange between stream flow and water movement through the redd, resulting in a reduction in the supply of dissolved oxygen to the egg and a hindrance to the removal of metabolites. Slaney *et al.* (1977) reported that rainbow trout egg survival was significantly reduced when spawning gravel contained more than 3% of fines (diameter 0.297 mm). In addition, Hall and Lantz (1969) determined that hatching success of coho salmon and cutthroat trout (*Onchorhynchus clarki*) was reduced by 40 to 80% when spawning substrates contain 20 to 50% fines (1-3 mm diameter).

Even if intergravel flow is adequate for embryo development, sand that plugs the interstitial areas near the surface of the stream bed can prevent alevins from emerging from the gravel (Koski 1966; Phillips *et al.* 1975). For example, the emergence success of westslope cutthroat trout was reduced from 76% to 4% when fine sediment was added to redds (Weaver and Fraley 1993).

Female stream spawning salmonids typically clean an area of the stream bed in which they bury their eggs. This nest building activity flushes sediments and increases the stream bed permeability. With time, sediment conditions within the redd gradually return to ambient levels (Wickett 1954; McNeil and Ahnell 1964). Under normal conditions, this slow increase in sediment intrusion is not a problem; however, increased levels of sediment within a system as a result of anthropocentric disturbance increase the rate and level of sediment intrusion and reduces the period of time in which the redd is clean. The period of time before sediment intrusion into the redd is very important with respect to the survival of salmonid larvae. Studies by Wickett (1954) suggest that sediment accumulation during early embryonic development may result in higher egg mortalities than if deposition occurs after the circulatory system of developing larvae is functional. This may be due to the higher efficiency in oxygen uptake by the embryo or

alevin with a functional circulatory system (Shaw and Maga 1942).

Ringler and Hall (1975) documented increased temperature and reduced dissolved oxygen levels of intragravel water in salmon and trout spawning beds because of clearcut logging practices. An associated reduction in resident cutthroat trout populations was attributed to this reduction in spawning habitat suitability. However, the failure to document serious reductions in coho salmon could be related to sediment clearing and removal by these larger fish during redd construction.

Sedimentation Effects on Fish Rearing Habitat

Sediment deposition also affects rearing habitat of juvenile fish since young salmonids frequently use the interstitial spaces in the stream bed for cover. Thus, a reduction in the suitability of potential rearing habitat as a result of sediment introduction is related to a reduction in the space available for occupancy (Reiser *et al.* 1985). When pools and interstitial spaces in gravel fill with sediment, the total amount of habitat available for rearing is reduced (Bjornn *et al.* 1977). Griffith and Smith (1993) found that numbers of juvenile rainbow trout and cutthroat trout decreased due to lack of available cover in heavily embedded gravel substrata. Interstitial space is particularly important during winter because juvenile fish live in these areas making them especially susceptible to impacts from increased sedimentation (Bjornn *et al.* 1977). Without these inter-gravel refugia, young fish may abandon the stream or move to less suitable areas where survival rates may be reduced.

Sedimentation Effects on Food Supply

Sedimentation can affect fish populations by altering the available food supply. Increased concentrations of suspended sediments and increased rates of sedimentation can reduce the primary productivity of the impacted area. Periphyton communities are likely the most susceptible to the scouring action of suspended particles or burial by sediments. At concentrations exceeding 115 mg/L, suspended sediments can reduce light penetration and primary productivity (Singleton 1985). A reduction in primary productivity has the potential

to appreciably decrease the food supply of macrobenthos that graze on periphyton (Newcombe and Macdonald 1992). Many macrobenthic organisms are, in turn, used as a food source by fish.

Increased sediment loads in streams can also have an effect on zooplankton and macrobenthos. Sediment release can effect the density, diversity and structure of resident invertebrate communities (Gammon 1970; Lenat *et al.* 1981). A number of studies have demonstrated decreases in invertebrate densities and biomass following sedimentation events (Wagener 1984; Mende 1989). Increases in sediment input may reduce the density of invertebrates by directly affecting aspects of their physiology or by altering their habitat. Suspended sediments can have an abrasive effect on invertebrates and interfere with the respiratory and feeding activities of benthic animals (Tsui and McCart 1981). Increased sediment deposition may also reduce the biomass of invertebrates by filling the interstitial spaces with sediments and by increasing invertebrate drift or covering the benthic community in a blanket of silt (Cordone and Kelley 1961; Tsui and McCart 1981). Increases in sediment deposition that affect the growth, abundance, or species composition of the periphytic (attached) algal community will also have an effect on the macroinvertebrate grazers that feed predominantly on periphyton (Newcombe and MacDonald 1991).

A change in particle-size distribution in the stream bed can alter the habitat and make it unsuitable for certain species of invertebrates. Gammon (1970) noticed that an increase in suspended sediments from 40 to 120 mg/L resulted in a 25 to 60% decrease in the density of stream macroinvertebrates. Likewise, Slaney *et al.* (1977) found that a 16 hour pulse of suspended sediments (2500 to 3000 mg/L) led to a 75% reduction of invertebrate biomass within the most affected areas.

Sedimentation can alter the structure of the benthic invertebrate community by causing a shift in the proportion from one functional group to another. For example, streams with clear water normally contain a high proportion of invertebrates in the shredder group; however, if sediment deposition is substantially increased, shifts to other groups such as grazers (Bode 1988) or collector-gatherers may occur

(Wagener 1984). Some studies indicate that increased inputs of sediments cause a shift towards chironomid-dominant benthic communities (Rosenberg and Snow 1975; Dance 1978; Lenat *et al.* 1981).

Benthic fauna possess behavioural and morphological adaptations which limit them from being displaced in a unidirectional flow environment (Hynes 1973). Invertebrate drift, however, is a continuous redistribution mechanism that occurs in most stream ecosystems. It is an important factor in the regulation of population density (Williams and Hynes 1976), in the dispersion of aggregations of young larvae (Anderson and Lehmkuhl 1967), in the abandonment of unsuitable areas (Williams and Hynes 1976), and in the recolonization of areas after disturbance (Barton 1977).

Invertebrate drift may be induced by elevated suspended sediment levels (Rosenberg and Weins 1978). Increased rates of downstream drift by macrobenthos can be induced by concentrations as low as 23 mg/L (Rosenberg and Snow 1975). Drifting affords invertebrate taxa that are sensitive to increased sediment loads the opportunity to avoid areas which become unsuitable as a result of high suspended sediment levels. Conversely, invertebrate drift is considered to be the most important component of ecosystem recovery following stream disturbances (Williams and Hynes 1976; Barton 1977; Young 1986). This is especially true in areas of swift-flowing waters (Waters 1964).

Sedimentation Effects on Overwintering Habitat

The magnitude of impact upon fish resulting from increased concentrations of suspended sediments and levels of sedimentation can vary seasonally. It has been argued that the lowered metabolic requirements during winter conditions may in some ways provide a protective influence to conditions such as gill trauma and decreased gill function (C. Newcombe, BCMELP, pers. comm.). However, the ability of the fish to compensate for the stress of suspended sediments is influenced by a number of factors including the physiological condition of the fish and its ability to respond to the stress.

Early live stages (i.e., eggs, alevins) of many salmonids are found in the stream bed during

the winter months. These stages are particularly sensitive to the effects of increased concentrations of suspended sediments and the deposition of fines sediments. The introduction of sediments during the winter, therefore, has the potential to appreciably influence these early life stages.

Bjornn *et al.* (1977) found that the number of juvenile salmon that a stream can support in winter was greatly reduced when the inter-cobble spaces were filled with fine sediment. The decreased carrying capacity was a function of both a loss of substrate cover for juvenile fish and a reduction in food as benthic invertebrate communities changed. Bjornn *et al.* (1977) suggested that the summer rearing or winter holding habitat may be more influential to the carrying capacity of a stream reach than embryo survival.

During winter fish generally experience decreased energy reserves and as such search for habitat that allows them to reduce energy expenditures (Clapp *et al.* 1990; Nickelson *et al.* 1992). Preferred habitats are species dependent; however, for most salmonids preferred habitats are located in low velocity areas such as pools and behind instream cover where focal velocities are low (Vondracek and Longanecker 1993; Griffith and Smith 1993; Modde *et al.* 1991; Heggenes and Saltveit 1990; Cunjak and Power 1986; Tschaplinski and Hartman 1983). By remaining in low velocity areas, fish are able to minimize their energy expenditures and hence reduce the rate of metabolic depletion (Cunjak and Power 1986).

Land-use activities that increase the delivery of fine materials to streams can significantly affect the overwintering survival of resident fish. A mechanism of potential impact is a depletion of critical energy reserves as a result of increased physiological stress, alterations in behaviour and/or exclusion from preferred sites of overwintering habitat. This is particularly deleterious to fish species and life stages that prefer to overwinter within the interstitial spaces of the stream bed. The net loss in energy reserves will depend on the concentration of sediment and the duration of impact. Dependent upon existing energy reserves, the fish may be able to tolerate the energy depletion attributable to an increase in the cough reflex and reduced feeding, but may not be able to tolerate the

energy depletion associated with displacement from critical habitats.

Preferred winter habitat areas of low current velocity are often predisposed to sedimentation (Cunjak 1996), and lower flows often experienced during winter may result in higher rates of sediment deposition. Bjornn *et al.* (1977) found, during sediment experiments, that the spring freshet from snow melt was rarely sufficient to transport sediment out of pools; therefore, the damage to these areas is frequently of a longer-term than sediments deposited in more erosive habitats. Due to natural factors, the availability of winter habitat is generally less than that of summer habitat and may be more influential in the determination of the stream's natural carrying capacity (Cunjak 1996; Mason 1976). A further reduction in the abundance of already limited winter habitat may significantly affect the overall fish population of a watercourse (Hartman and Scrivener 1990). Additive to this problem may be a reduction in food supply resulting from benthic drift or burying of food supplies. Elwood and Waters (1969) observed that increased sedimentation reduced the population of invertebrates and hence the capacity of the stream to support brook trout. A reduced food supply, and a greater expenditure of energy in food search and avoidance of higher concentrations of suspended sediments may significantly impact upon the fish's ability to compensate for negative physiological changes and the ability to survive the winter.

Sediment Load Biological Response Relationships

The Dose/Response Approach

One method that has been developed to address the issue of quantifying the adverse effects of TSS on fish is the ranked effects model first put forward by Newcombe and MacDonald (1991). This model compiled information from more than 70 studies on the effects of inorganic suspended sediments on freshwater fish (mainly salmonids) and invertebrates, and ranked the severity of impacts from 1 to 14 (rank effects). Linear regression was used to correlate ranked effects with intensity (concentration x duration) of increased suspended sediment load (Newcombe and MacDonald 1991). Since the effect of elevated TSS levels on fish is a function of both the concentration of suspended sediment and

the duration of the exposure, Newcombe and MacDonald (1991) developed a Severity Index (SI). This Index provides a standardized relative measure of exposure. It is the natural logarithm of the concentration ($\text{mg}\cdot\text{L}^{-1}$) multiplied by hours of exposure (i.e., $\text{Ln mg}\cdot\text{h}\cdot\text{L}^{-1}$). This SI provides a convenient tool for predicting effects of episodes of elevated suspended sediments of known concentration and duration.

The usefulness of the Newcombe and Macdonald (1991) concentration-duration response model has been questioned in the past (Gregory *et al.* 1993). The main concerns with the approach are the highly variable nature of the data used to develop the severity of effects model that reduces its predictive power, and the concern that the model is unrealistically simplistic (Gregory *et al.* 1993). A separate concern associated with the Newcombe and Macdonald (1991) severity of effects model is the severity index ($\text{Ln mg}\cdot\text{h}\cdot\text{L}^{-1}$) assumes a unit increase in episode duration (in hours) has a similar effect as a unit increase in concentration (in mg/L) (Anderson *et al.* 1996).

In 1994, the ranked effects model was further refined (Newcombe 1994). Using 140 articles on suspended sediment pollution, Newcombe developed a database of nearly 1200 datapoints concerning the effects of suspended sediments and associated effects upon marine and freshwater biota. With this database, regression analysis was used to relate severity of effect to the dose of TSS for specific fish species or assemblages. This approach was used to describe the dose/response relationship for the effects of suspended sediments on salmonid fishes (Newcombe and MacDonald 1991), on juvenile salmon (Newcombe 1994) and for other coldwater fishes (Newcombe 1994) using subsets of the dataset which are presented in complete form in Newcombe (1994).

In addition to the dose/response relationships presented in for salmonid fishes (Newcombe and MacDonald 1991) and coldwater fishes and underyearling trout (Newcombe 1994), Newcombe and Jensen (1995) further expand upon dose/response relationships of aquatic resources to sediment exposure. Newcombe and Jensen (1995) presented six sediment dose/response relationships for specific fish communities exposed to elevated sediment loads. One of the important additions to the analysis presented in Newcombe and Jensen (1995) was the linkage of grain size of sediment

to the nature of ill-effect associated with sediment exposure. The dose/response relationships presented in Newcombe and Jensen (1995) were organized according to four variables: taxonomic group; life stage; natural history and estimated predominant particle size range of the sediment episode. The dose/response relationships were characterized by three variables: [x], [y], and [z], where:

[x] = duration of exposure expressed as the natural log of hours;
[y] = concentration of sediment expressed as natural log of mg/L ;
[z] = severity of ill effect (SE).

Since the work of Newcombe and MacDonald (1991) and Newcombe (1994) used the Stress Index ($\text{Ln concentration} \cdot \text{duration}$), the dose/response relationships presented were of the form:

Equation 1: $z = a + bx$
where:
SE = severity of ill effects
a = intercept;
b = slope of the regression line
x = stress index value

The six dose/response relationships presented in Newcombe and Jensen (1995) do not characterize response using the stress index, and are presented in the form:

Equation 2: $z = a + b(\text{Ln } x) + c(\text{Ln } y)$

The dose/response relationships (Newcombe and MacDonald 1991; Newcombe 1994; Newcombe and Jensen 1995) provide insight into the relationship between sediment release and adverse effects on a variety of fish communities. These relationships provide increased precision for the prediction of response of a particular fish species or assemblage of species based on a given dose of suspended sediment.

The database used to determine these relationships relied heavily on the physiological response of fish to increases in sediment load; therefore, the relationships presented may not be directly applicable to the prediction of physical alteration to fish habitats due to sediment load increases and increased sedimentation nor the long term impacts from reduced growth or the possible exclusion from certain habitats.

Developing the Habitat Effects Database

Anderson *et al.* (1996) attempted to develop effects relationships for fish habitat response to increased sediment. The first step in developing more specific dose/response criteria for habitat effects was to search the literature for studies reporting TSS concentrations and their effects on fish habitat. Several fundamental criteria had to be satisfied for data from a report to be included in the expanded database that was developed. The report had to give at least: 1) a concentration of TSS; 2) a duration of exposure to 3) one or more identified species of fish; and 4) a description of the effect. The severity of effect, and occasionally other data, sometimes had to be inferred from the qualitative descriptions. A total of 18 reports, containing some 53 new documentations of TSS effects, were retrieved in the literature search (Anderson *et al.* 1996).

Dose/Response Relationships of Freshwater Habitats to Sediment Releases

Severity of ill-effects rankings on a scale from 0-14 were assigned to each documented effect following the severity-of-ill-effects scale published by MacDonald and Newcombe (1993), Newcombe (1994), and Newcombe and Jensen (1995). The nature of the observed habitat effect was assigned one of the following class effect rankings:

- SE = 3 Measured change in habitat preference;
- SE = 7 Moderate habitat degradation - measured by a change in the invertebrate community;
- SE = 10 Moderately severe habitat degradation - as defined by measurable reductions in the productivity of habitat for extended periods (months) or over a large area (kms);
- SE = 12 Severe habitat degradation - as measured by long-term (years) alterations in the ability of existing habitats to support fish or invertebrates; or ,
- SE = 14 Catastrophic or total destruction of habitat in the receiving environment.

Emphasis was placed on sediment release events (i.e., effects of sediment pollution events rather than chronic erosion and sediment load problems). The database excluded any datapoints for which the extent of habitat modification could not be ascertained from the primary manuscripts. The database was reduced to 35 entries (Anderson *et al.* 1996) and was used in the analysis described below to develop a relationship between sediment dose and habitat effects.

The dose/response approach of Newcombe and MacDonald (1991) and Newcombe (1994) defines dose as the product of TSS concentration (C in mg/L) and duration of exposure (T in hours). This definition of dose is strictly empirical and reflects the observation that the product of concentration and duration bears a closer correlation with ranked effects than concentration alone. The inherent assumption is that brief exposures to high doses of TSS are equivalent in effect to prolonged exposure of much lower doses. Since severity of effect is determined based on a linear relationship with dose ($\ln C \cdot T$) of the form $SE = a + b (\ln C \cdot T)$, the biological receptor response (SE) is assumed to respond to an effective dose in which concentration and duration are equally as important (i.e., the Effective Dose = $C^n T$; where $n = 1$).

However, it has been proven in much of the literature related to the response of biological receptors to toxic agents, that the relationship between concentration and duration is often more complex. That is, a high concentration for a very short time can cause a higher or lower response than can a low concentration for a longer time (Zelt 1995). In essence, by assuming a linear response (as measured by SE) to dose (as a function of $\ln C \cdot T$), it is assumed that a unit increase in concentration (in mg/L) is equal to a unit increase in time (in hours). This assumption may or may not be a valid one. In an effort to address the potential for non-linearity in the relationship between concentration and duration in determining the effective dose of sediment (i.e., Effective Dose = $C^n T$; where $n \neq 1$), Newcombe and Jensen (1995) used multiple regression analysis to develop severity of effect relationships based on concentration and duration:

$$\text{Equation 3: } z = a + b(\ln x) + c(\ln y)$$

This approach, in effect, allows for different factors (slopes) to be assigned separately to the variables of concentration and duration.

In order to explore the relationship between concentration and duration in influencing habitat change, Anderson *et al.* (1996) also used multiple regression analysis to analyze the habitat effects database. This analysis identified a relationship between sediment exposure and habitat effects that can be described by the equation:

$$\text{Equation 4: } z = 0.637 + 0.740\text{Ln}(X) + 0.864\text{Ln}(Y);$$

$$r^2(\text{adj}) = 0.627; n=35; p<0.001.$$

Statistics for the multiple regression relationship presented are summarized in Table 1. The “T” statistic for each slope in the regression (Table 2) is an expression of the importance of each variable with respect to the relationship derived. The higher score attributed to duration indicates its importance in determination of habitat effects. This indicates that concentration and duration affect the extent of habitat alteration in dissimilar ways or in other words, that the effective dose of sediment is a function of a non-linear relationship between the two predictive variables (i.e., Effective Dose = $C^n \cdot T$; where $n \neq 1$).

Table 1
Statistics for the Multiple Regression Relationship (Equation 4)

| Variable | Coefficient t | Std. Error | Std Coef | Tolerance | T | P (2Tailed) |
|-------------|---------------|------------|----------|-----------|-------|-------------|
| Constant | 0.0637 | 1.293 | 0.000 | | 0.493 | 0.625 |
| Ln Con. | 0.864 | 0.176 | 0.520 | 0.973 | 4.903 | 0.000 |
| Ln Duration | 0.740 | 0.111 | 0.706 | 0.973 | 6.652 | 0.000 |

Discussion

Confounding Factors

The dose/response relationships that have been developed make generalizations about the anticipated level of effects to the aquatic environment that may result from elevated sediment levels. Since these are generalizations, the actual effects that are realized by a sediment release episode may be more or less severe based on a number of confounding factors.

The potential for adverse effects on fish and their habitats associated with sediment release is a function of increasing particle size (Newcombe 1996). More information relating dose-response relationships between specific fish guilds or habitat types as a function of particle size range is required in order to develop a better understanding of this confounding factor.

The angularity or mineralogy of suspended particles may play a important role in the potential for physiological or toxicity effects

(Newcombe and Jensen 1995). The angularity of a particle may be of particular importance with respect to gill abrasion of fish within the receiving environment, and may also influence the rate of infiltration of particles into the stream bed. Meanwhile, the mineralogy of the particle may be important since the particle itself may have some potential chemical activity at the cellular level (Newcombe and Jensen 1995). In addition, the potential for contaminants adsorbed to sediment particles is also a concern, since contaminated sediments could have more dramatic effects than those which might be caused by the increase of sediment load alone.

The amount of material that intrudes into the gravel bed has been shown to be highly dependent on the grain-size distribution of the transported sediment as well as that of the gravel bed. If the suspended sediment load is composed of very fine material, the gravel pores tend to fill from the bottom to the top of the pavement layer. If the suspended particles are larger in size, angular or platelet in shape, a film can develop within the substrata which will tend to limit the intrusion of additional sediments into the interstitial spaces of the stream bed (Beschta and Jackson 1979). Beschta and Jackson (1979) concluded that the finer the suspended

sediment, the greater the potential was to fill interstitial voids.

The shape of the stream bed substrata may also affect sediment deposition. Under low flow conditions, rounded stream bed substrata tend to accumulate more sediment than angular substrata, whereas, during high flows, the reverse is true (Meehan and Swanston 1977). This may be due to the reduced turbulence levels at the gravel bed in rounded stream beds during low flows, while at higher discharges a flow separation zone can develop behind angular materials causing greater sediment deposition (Reiser *et al.* 1985).

The temperature of the water can have an impact on the severity of the effects caused by a sediment release event. The oxygen holding capacity of water and the metabolic and respiratory rates of fish are influenced by water temperature. Consequently, the effects of sediment exposure may be greater in seasonably warm waters than in seasonably cold water (Newcombe and Jensen 1995). However, during winter conditions aquatic organisms may be especially vulnerable to additional stressors. Since stress has been defined as the sum of all physiological responses, the severity of effects that are caused by sediment release will be a function of the level of stress at the time of, or before, the period of elevated sediment load.

Factors Influencing the Risk of Habitat Alteration

Many attributes may influence the extent of sediment-induced aquatic habitat alteration. The principal factor which influences the extent of habitat alteration is the increase in sediment load associated with watershed disturbance. In addition, the sensitivity of the exposed habitats and the length of time habitats are likely to be impaired are also important factors which influence the level of habitat alteration.

Considerations regarding the sensitivity of the receiving environment include the susceptibility to alteration of the habitats within the receiving environment, and the timing of the elevated sediment loads. In addition to overall sensitivity of the watercourse and the sensitivity of the habitats it supports, a related consideration is the species and life stages which may be present during times of instream activity associated with forest harvesting or road

construction. Certain life stages are especially sensitive to increases in sediment load (such as developing eggs and larvae, or overwintering fish – particularly juvenile salmonid in streams); as a result, the presence of these life stages during instream construction would increase the sensitivity of the watercourse to disturbance.

An additional related consideration regarding construction timing sensitivity is the flow conditions within the watercourse during periods of elevated sediment loads. Watercourse discharge influences the concentration of suspended sediments, transport and deposition of materials as well as the extent of habitat present and the ability of resident biota to avoid areas of elevated sediment.

The duration of habitat impairment is one of the most critical considerations in relating the extent of habitat alteration to aquatic community impact since habitat alteration will only affect the aquatic community if the altered habitat would have been used during the period of impairment. Therefore, the level of concern associated with habitat alteration increases as the duration of impairment increases. The duration of impairment is considered to be the length of time before deposited sediments are flushed from the watercourse into a less sensitive area such as a lake and are normally viewed as the number of life history stages which are impacted during the period in which the habitat is in an altered state.

As a result, evaluation of the extent of habitat alteration associated with elevated sediment loads must consider the level of sediment load increase (concentration and duration), the nature of the habitats and communities affected and the duration of likely impairment caused. The severity of effects approach which have been developed by Newcombe and others are not easily applied to the prediction of habitat change and attributing justifiable numbers to abstract concepts such as system sensitivity is difficult at best. Resource managers need to apply expert judgement to ensure that models and assumptions are not applied blindly and that model results do not violate the most important management tool, which is common sense.

Mitigation Measures

Although most disturbances within a watershed inevitably increase the amount of erosion, the delivery of sediments to streams resulting from

disturbances can be largely circumvented by proper design and planning. As discussed previously, logging road construction and operation can dramatically increase sediment loads in streams. The amount of disturbance caused by road construction and maintenance depends upon its design standard, gradient, total distance of road and intensity of use. For example, Megahan and Kidd (1972) indicated that proper siting and construction of roads could eliminate much of the sediment loading associated with mass erosion in steep terrain.

The density of logging road distribution can be a major factor in determining the associated increase in sediment loads in streams (Waters 1995). Cederholm *et al.* (1981) documented the greatest accumulation of fine sediments in streambeds associated with road areas that exceeded 2.5% of the total basin area. In addition, the length of logging roads also influences sediment delivery to streams. Cederholm *et al.* (1981) calculated total road lengths of 2.5 km of road per km² of watershed basin produced sediment more than four times natural rates. As a result, sediment mitigation measures have concentrated on the minimizing sediment associated with logging roads.

The logging system used is a critical factor in the determination of road density. For example, the use of jammer logging systems in the past has required a dense network of roads since these logging systems require a maximum road spacing of approximately 150 m. In areas of steep terrain, this approach to logging may disturb soil on 25 percent of the total logging area. High lead logging is a method that reduces the level of disturbance since a reduced road network is required to support this method. In steep topography areas in Idaho, high lead logging has reduced disturbance to less than 10% of the logging area. On the same types of slopes, jammer logging roads spaced 60-120 m apart disturbed 25-30% of the total area (Rice *et*

al. 1974). Skyline logging systems permit an even wider road spacing of 500 m or more, depending on local conditions and topography. This reduces the area disturbed by 75% and may provide a greater buffer area for sediment filtration between the road and the stream channel (Megahan and Kidd 1972). Skyline, balloon and helicopter systems have been developed to permit the logging of steep topography with a minimum amount of road disturbances (Rice *et al.* 1974). Binkley (1965) estimated that skyline yarding would save from 2.2 to 2.9 kilometres of road over that required for high lead in a 1600 ha drainage area to be logged.

Control measures to minimize sediment delivery to streams associated with accelerated erosion within watersheds disturbed by forest harvest operations are of paramount importance in minimizing elevated sediment loads in streams. Since the best mitigation measure to minimize sediment loads in streams is to minimize the amount of erosion in the watershed, design features for access planning can be implemented which will minimize erosion associated with logging roads. Waters (1995) summarizes design features for the reduction of erosion from logging roads. This summary table is reproduced as Table 2.

Disturbance within watersheds inevitably increases sediment loads within adjacent watercourses. These increases in sediment load can have substantial effect on fish and on their habitat. Through the proper planning and design of forest harvest systems, the level of sediments delivered to streams can be minimized allowing for the existence of both forestry and fish.

Table 2
Logging Road Mitigation Measures
(Taken from Water 1995)

| Design Feature | Method | Purpose |
|---------------------|--|--|
| road placement | avoid streams and steep slopes | reduce erosion & mass soil movement |
| road length | few, short roads | reduce total area of exposed roadbed |
| road width | narrow as practicable | reduce area of disturbance |
| road grade | 5-15%, not flat, minimum 3% for drainage | avoid rapid run-off |
| road surface | gravel, crushed rock roadbed | reduce roadbed erosion |
| cut slopes | vertical or near vertical cut | reduce excavation and erosion of slope |
| fill slopes | avoid road drainage and woody debris in fill | stabilize fill slopes |
| road drainage | outslope drainage on shallow slopes, inside drainage on steep grades | disperse drainage |
| inside drainage | ditch inside road | carry run-off along road |
| cross-drainage | underground pipe or log construction | drain inside ditches or waterways |
| culvert | | |
| water bar | low hump, 30 ° angle downslope | disperse run-off from roadbed |
| broadbased dip | wide drainage dip or bench | disperse run-off from roadbed |
| stream crossing | minimize number, use appropriate method | reduce direct aquatic impact |
| vegetation planting | seed grass, plant trees | reduce exposed surface |
| daylighting | cut canopy to permit sunlight penetration | promote drying of roadbed |
| abandonment | close access, remove crossings, install dips and water bars | avoid subsequent use and maintenance |

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