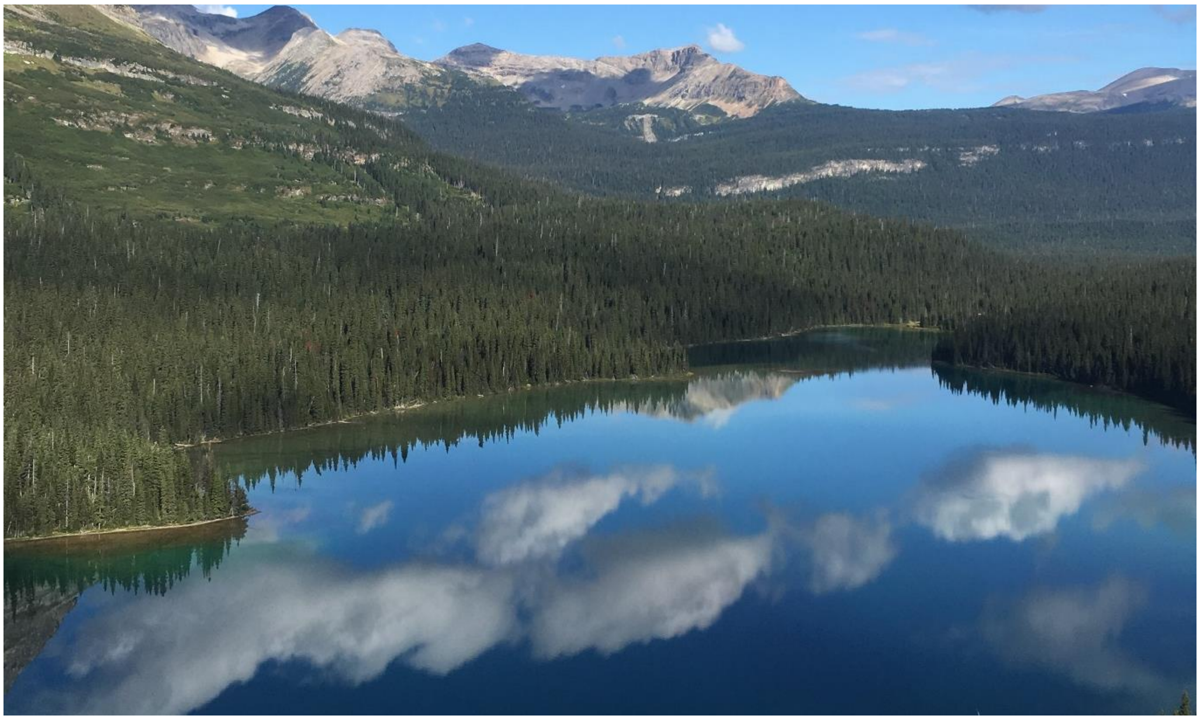
WATER QUALITY GUIDELINE SERIES

Ambient Water Quality Guidelines for Turbidity and Suspended and Benthic Sediments

Technical Appendix

Ministry of Environment and Climate Change Strategy
Water Protection & Sustainability Branch



No. WQG-18

The Water Quality Guideline Series is a collection of British Columbia (B.C.) Ministry of Environment and Climate Change Strategy water quality guidelines. Water quality guidelines are developed to protect a variety of water values and uses: aquatic life, drinking water sources, recreation, livestock watering, irrigation, and wildlife. The Water Quality Guideline Series focuses on publishing water quality guideline technical reports and guideline summaries using the best available science to aid in the management of B.C.'s water resources. For additional information on B.C.'s approved water quality parameter specific guidelines, visit:
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Document citation:
B.C. Ministry of Environment and Climate Change Strategy. 2021. Ambient Water Quality Guidelines for Turbidity and Suspended and Benthic Sediments (Reformatted from original 1997 version). Water Quality Guideline Series, WQG-18.

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Notes on this updated version:
This report is a reformatted version of the original technical document released in 1997. The following section of the original report have been omitted: The industrial water use guidelines have been removed as B.C. no longer develops or supports guidelines for industrial water use. The recreational water quality guidelines have been removed and are now found in B.C.'s Recreational Water Quality Guidelines. The source drinking water quality guidelines have been updated. See the recreational and source drinking water quality guidelines on the Water Quality Guidelines webpage for more information.

Cover Photograph:
Location: Lower Blue Lake, B.C.

Acknowledgements

The B.C. Ministry of Environment and Climate Change Strategy acknowledges the contributions of those who provided expertise and assistance in the preparation and review of the original document including: J. Rex (British Columbia Conservation Foundation), C. Newcombe (Habitat Protection Branch, Ministry of Environment, Lands and Parks), L.G. Swain, E.T. White, and R.W. Preston (Water Management Branch), and Jesse Brown and Mary-Lou Haynes (MacDonald Environmental Science)

PREFACE

This Provincial Water Quality Guideline Technical Appendix is an update of an earlier Appendix entitled "Water quality criteria for particulate matter" by H.J. Singleton first printed in February 1985 and reprinted without change in March 1995. The document is updated using information and data obtained between 1984 and 1995 on the subject matter. Although pre-1984 data was not reviewed in this document, these data were used, wherever pertinent, for the purpose of deriving concentration-response curves. All data used in concentration-response curve fitting were referenced appropriately. Readers wishing to obtain a review of the literature used to develop the 1985 water quality criteria for particulate matter should consult the reprinted March 1995 edition.

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SUMMARY OF RECOMMENDED WATER QUALITY GUIDELINES

Water Use	Parameter	Water Quality Guideline
Raw drinking water with treatment for particulates	Turbidity	Natural background turbidity is ≤ 50 NTU: Change from background should not exceed 5 NTU
		Natural background turbidity is > 50 NTU: Change from background should not exceed 10% of the background turbidity
Aquatic life, freshwater, marine and estuarine		Change from background of 8 NTU at any one time[1] for a duration of 24 h in all waters during clear[2] flows or in clear water; Change from background of 2 NTU at any one time[1] for a duration of 30 d in all waters during clear[2] flows or in clear water; Change from background of 5 NTU at any time when background is 8 to 50 NTU during high flows or in turbid[2] water; Change from background of 10% when background >50 NTU at any time during high flows or in turbid[2] water
	Non-filterable residue	Change from background of 25 mg/L at any one time[1] for a duration of 24 h in all waters during clear flows[2] or in clear water; Change from background of 5 mg/L at any one time[1] for a duration of 30 d in all waters during clear flows[2] or in clear water; Change from background of 10 mg/L at any time when background is 25 to 100 mg/L during high flows or in turbid[2] waters; Change from background of 10% when background >100 mg/L at any time during high flows or in turbid[2] waters
	Bedload	N.R.G.[3]
	Substrate composition	% fines not to exceed 10% <2 mm, 19% <3 mm and 28% <6.35 mm at salmonid spawning sites; Geometric mean diameter and Fredle number not less than 12 and 5 mm, respectively. (min 30-d intragravel DO[4], 6 and 8 $mg \cdot L^{-1}$, respectively)
Wildlife water	Turbidity	Change from background of 10 NTU when background ≤50 NTU; Change from background of 20% when background >50 NTU
	Non-filterable residue	Change from background of 20 mg/L when background is ≤100 mg/L; Change from background of 20% when background >100 mg/L
Livestock watering	Turbidity	Change from background of 5 NTU when backgrounds ≤50 NTU; Change from background of 10% when background >50 NTU
	Non-filterable residue	Change from background of 10 mg/L when background is ≤100 mg/L;

Water Use	Parameter	Water Quality Guideline
		Change from background of 10% when background >100 mg/L
Irrigation	Turbidity	Change from background of 10 NTU when background ≤50 NTU; Change from background of 20% when background >50 NTU
	Non-filterable residue	Change from background of 20 mg/L when background is ≤100 mg/L; Change from background of 20% when background >100 mg/L

[1] During the course of sampling (n = 24 i.e., every hour for 24 h; n = 30, i.e., every day for 30 d), if any sample is above the guideline, it is considered an exceedance. To statistically test whether exceedances have occurred, comparisons between control (e.g., upstream) and impacted (e.g., downstream) sites are required. This can only be done if all samples (i.e., 24 samples in 24 hours and 30 samples in 30 days) are considered in the analysis. Depending on the test chosen for the site comparison, the null hypothesis (H_0) could be stated differently. For example, H_0 may be that samples from two sites (or many sites) were drawn from the same mean, or the median of the population of all possible differences is zero.

[2] In this document, the terms "clear flow period" and "turbid flow period" are used to describe the portion of the hydrograph when suspended sediment concentrations are low (i.e., <25 mg/L) and relatively elevated (i.e., ≥25 mg/L), respectively. These new terms have been proposed because the commonly utilized descriptive terminology (e.g., low flow and base flow, high flow or freshet flow, ascending or descending limb of the hydrograph) do not adequately identify the periods of low and elevated sediment transport in stream systems. In addition, many stream systems, such as those fed by lakes or reservoirs, run clear year-round. Therefore, it is possible that the water quality guidelines for suspended sediments could be incorrectly applied if standard hydrological terms were utilized in this document. The clear and turbid flow periods for individual stream systems should be defined using data on the background concentrations of suspended sediment at the site-specific level. The recommended transition value (25 mg/L) was selected by examining the hydrographs for a number of streams in British Columbia and is intended to provide an operational definition of clear flow conditions that can be applied consistently in the province.

[3] N.R.G. No recommended guideline.

[4] Ambient water quality guideline for dissolved oxygen for British Columbia (Ministry of Environment Lands and Parks 1997).

GUIDANCE ON THE APPLICATION OF WATER QUALITY GUIDELINES

Aquatic ecosystems throughout North America are affected by pollution episodes that have the potential to adversely affect fish, invertebrates, and aquatic plants. While many concerns have been raised in recent years regarding the impacts of toxic chemicals released into these systems, the mobilization of fine inorganic particles and their subsequent deposition in sensitive habitats are, arguably, the most pervasive problem facing aquatic environmental managers. Researchers in this field have provided managers with little practical guidance for making regulatory decisions. In the absence of effects-based water quality criteria for suspended sediments, regulatory decisions have generally been either arbitrary or based on background conditions at the site. Often, it has been assumed that statistically significant differences in the concentration of background suspended sediments and the current levels at a given site indicate a need for site remediation.

The pollution control strategies used during the 1970s and 1980s were based on the assumption that suspended sediments would cause little or no harm to fish and aquatic life at relatively low concentrations, regardless of the duration of exposure. For example, suspended sediment concentrations (as measured by non-filterable residues) in the order of 25 mg/L were frequently accepted, for pollution control purposes, as the thresholds for adverse biological effects (e.g., U.S. EPA 1972). The concept of exposure duration was not considered in the pollution control paradigm, thus low-level pollution episodes were officially tolerated for indefinite periods of time. A comparison of the traditional concentration-response model with a dose-response model (dose = concentration x duration) indicates that dose exposure is more strongly correlated with severity of ranked effects (Newcombe and MacDonald 1991). A Stress Index (SI) model has been developed to predict the impacts of suspended sediments on aquatic ecosystems, based on dose-exposure information (Newcombe 1986; Newcombe 1993). The SI model identified ranges of pollution intensities that are generally associated with behavioral, sublethal and lethal effects. The SI model has since changed terminology and is referred to as the severity-of-ill-effects model (SEV) (Newcombe and Jensen 1996a).

Although there is a need for a simple method to predict the adverse effects of suspended sediment in aquatic ecosystems (Newcombe and MacDonald 1991; Gregory et al. 1993), there is some disagreement on whether the existing information can be used to develop such a tool. Following a thorough review of the available literature methods (arbitrary and effects-based) for criteria development (e.g., SEV), the SEV model was chosen for the British Columbia aquatic life criteria development and for predicting the expected severity-of-ill-effects of suspended sediments.

The B.C. Ministry of the Environment, Lands & Parks (BC Environment) develops province-wide ambient water quality guidelines for substances and physical attributes of importance in both fresh and marine surface waters. These threshold values provide a basis for site-specific ambient objectives, for waste water discharge limits and fees and to help identify areas with degraded conditions. The criteria are not legally binding but are intended as tools to provide policy direction to those making decisions affecting water quality issues (Ministry of Environment Lands and Parks, Province of British Columbia 1995). Note, however, that objectives or guidelines can be used in a court of law to demonstrate environmental damage.

Anthropogenic activities such as forest management, road building, construction, dredging and gravel pit operations, can cause marked changes in the physical, chemical, and biological characteristics of the watercourses located nearby and those located downstream. The environmental impact is often attributable to the type of activity. For example, with timber harvest, the nature, extent and severity of the environmental effects are a function of the area cut as a percent of the given watershed, the species of tree harvested, the density of the stand harvested, and the local topography of the area affected (Verry 1986). The guidelines developed herein are the British Columbia ambient water quality guidelines for turbidity, suspended sediments, substrate composition and bedload transport in both freshwater and marine conditions. The guidelines will function as management tools, against which to assess whether a water use is adequately protected.

Water quality guidelines have been recommended to protect British Columbia’s major water-use categories which include drinking water, aquatic life, wildlife, livestock watering, irrigation, recreation and aesthetics, and industrial water supplies. Exemptions from objectives based on these guidelines may be allowed for necessary, short-term anthropogenic activities. It is recommended that such activities strictly conform to the Provincial Codes of Practice overseeing their operation when potential environmental disturbances are involved. For example, the British Columbia Forest Practices Code was enacted to establish mandatory requirements for forest management practices throughout the province. Among other things, the code specifies standards for road building, timber extractions, and related activities, which ought to minimize the impact of this land use on aquatic ecosystems. The guidelines presented in this document are intended to protect fish and other aquatic life by identifying benchmarks for environmental quality related to fine sediments. In addition, guidelines can be used to assess the impact of forest management activities.

With regards to the aquatic life water quality guidelines, the SEV model will predict the expected severity of effects once the guidelines have been exceeded. This method will assist BC Environment in their design, planning and implementation of control options. Because guidelines interpretation hinges on site-specific conditions, the guidelines are to be used as starting points on which site-specific objectives can be developed. In the case where guideline exceedances have occurred and compliance and/or prosecution measures are required, the reader should refer to Newcombe and Jensen (1996a, b, c) for further guidance.

The following gives further guidance on how to use the Provincial recommended aquatic life guidelines of 8 NTU and 25 mg/L, for 24 h or 2 NTU and 5 mg/L for 30 days during clear flow periods (see Section 2.1.1). This two-pronged approach to guideline setting for suspended sediments recognizes that exposure duration plays a key role in the toxicity response. A sampling strategy is provided as a complementary document to this document to assist field personnel in determining whether guideline exceedances occur over the short and long-term. During clear flow (see Section 2.1.1), the above guidelines are to be used. Site-specific water quality objectives consider water values that may, for example, lower the acceptable levels in very sensitive habitats with high resource use, or raise the acceptable levels for the setting of industrial abatement targets. For example, a site-specific objective for a site that has high background concentrations of suspended sediments (Fraser River) will probably require a large change in suspended sediments to cause the same SEV increase as would a small change in systems with low suspended sediment concentrations. The SEV model curves can assist a manager to determine the severity of effects this change in concentration will bring at a given time. As well, SEV model curves have been built on eight separate groups of aquatic biota. Hence, the establishment of site-specific objectives can be confined to a particular group of organisms that reflect the regional system under study.

1. DEFINITIONS, CHEMISTRY AND ANALYTICAL DETERMINATIONS

Turbidity, suspended sediments, bedload sediments, and substrate composition are parameters that describe the physical conditions of a water system. British Columbia water systems where these have been measured include open ocean, estuaries, lakes, rivers, streams, and groundwater. Depending on the season and location, measurements describing these parameters in a system will vary naturally and will be influenced by anthropogenic inputs. The following defines what these physical parameters are, discusses their relationship, and specifies the most current analytical method by which they are measured.

1.1 Turbidity

Turbidity is a measure of the lack of clarity or transparency of water. It is caused by biotic and abiotic suspended or dissolved substances in the water sample. The higher the concentration of these substances in water, the more turbid the water becomes. Technically, when passing light through a water sample, turbidity is an expression of the optical properties of substances that causes light to be scattered and absorbed rather than transmitted in straight lines through the sample (Wetzel 1975).

The most reliable method for determining turbidity is nephelometry (light scattering by suspended particles) which is measured by means of a turbidity meter giving Nephelometric Turbidity Units (NTU). Other methods giving Jackson Turbidity Units (JTU) or Formazin Turbidity Units (FTU) are available but have limitations or are not widely utilized. A nephelometer, much like a spectrophotometer, sends a beam of incident light through a water sample. Photo-electric cells in the instrument measure the light that is reflected at right angles to the sample. Presuming that all measures of scattered light in the sample are equal, light scattered perpendicularly will be a proportional measure of all scattered light and hence the turbidity of the sample.

Nephelometers are available to take turbidity measurements in the field. Should water samples be taken back to the laboratory, they should be stored in the dark and measures taken within a 24-h period to avoid biodegradation, pH changes, and settling of particles which will give misleading results. Environmental samples will vary within the normal range of 1 to 1000 NTU (Chapman 1992).

Transparency is the limit of visibility in the water (Wetzel 1975). In accordance with Beer-Lambert's Law, the amount of light transmitted through a water sample without diffraction, refraction, or absorption is measured with the use of a spectrophotometer. In water, a transparency measurement can be thought of as the vertical extinction of light. A familiar field apparatus that approximates water transparency is the Secchi disk. It measures the depth at which sunlight is reflected from a 20 cm diameter white disk lowered into the water column. A Secchi disk reading, which correlates closely with percentage transmission of light (Wetzel 1975), can be used to obtain complementary measurements of turbidity. It should be used preferably at midday, in relatively deep open water systems such as the ocean, estuaries, lakes and rivers.

1.2 Suspended sediments

The type and concentration of suspended matter controls the turbidity and transparency of the water. Suspended matter consists of silt, clay, fine particles of organic and inorganic matter, soluble organic compounds, plankton, and other microscopic organisms. Suspended matter is measured in the laboratory by both filterable and non-filterable residues of a water sample. Undissolved particles make up the non-filterable residues, these varying in size from approximately 10 mm to 0.1 mm in diameter, although it is usually accepted that the suspended solids are the fraction that will not pass through a 0.45 µm pore diameter glass fiber filter. For the purpose of deriving water quality guidelines, this solids fraction, containing both biotic and abiotic components, will be referred to as total suspended sediments with the unit of measure being in µg/L/. A sediment particle grade scale developed by the American Geophysical Union Subcommittee on Terminology and the settling velocities of these particles in water (Cooke et al. 1993) will be used as standard sediment terminology for guideline development (Table 1).

Table 1. Standard terminology for sediment particle size.

Sediment particle size		Size-class of sediment particle	Velocity of settling particle (mm/sec)
Millimeters	Microns		
2,000-4,000		**boulders** very large	
1,000-2,000		large	
500-1,000		medium	
250-500		small	
130-250		**cobbles** large	
64-130		small	
32-64		**gravel** very coarse	
16-32		coarse	
8-16		medium fine	
4-8		very fine	
2-4	1,000-2,000	**sand** very coarse	100-200
1-2	500-1,000	coarse	53-100
0.5-1	250-500	medium	26-53
	125-250	fine	11-26
	62-125	very fine	3-11
	31-62	**silt** coarse	1-3
	16-31	medium	0.18-0.66
	8-16	fine	0.044-0.18
	4-8	very fine	0.011-0.044
	2-4	**clay** coarse	<0.011
	1-2	medium	
	0.5-1	fine	
	0.24-0.5	very fine	

Relationships between turbidity and suspended sediments are site-specific as turbidity is affected by factors such as the concentration, size, shape, and refractive index of suspended sediments, (Singleton 1985; Lloyd et al. 1987; Allen 1979) and the water color (Gippel 1995). Relationships vary from stream to stream and between seasons in the same stream. Comparison of 573 paired non-filterable residue and turbidity measurements taken from several watercourses in the Kootenay Region of British Columbia indicated an average ratio of approximately 3:1; with 86% of the paired measurements ranging from a 1:1 to a 4:1 non-filterable residue/turbidity ratio (Ministry of Environment 1978). A similar relationship comparing monthly means of non-filterable residue and turbidity in the lower Fraser River was also found (Ministry of Environment 1980a). A plot of turbidity and suspended sediment concentration for miscellaneous disturbed drainages within the Elk River basin is shown in Figure 1.

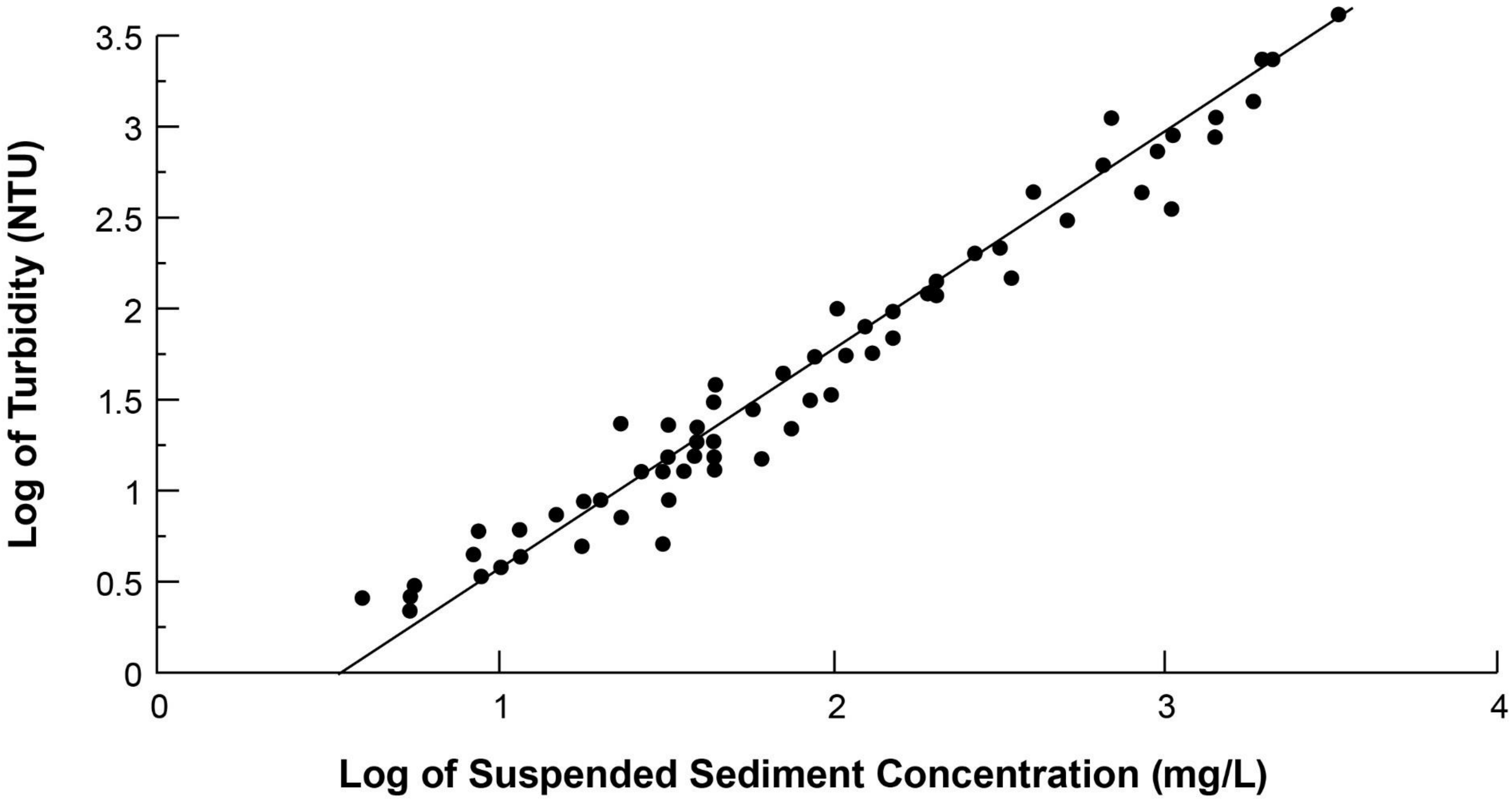

Figure 1. Turbidity (NTU) vs. suspended sediment concentration (mg/L) within the Elk River basin.

At sites where the relationship between suspended sediment concentration and turbidity is known and characterized, turbidity can be used as a surrogate to predict suspended sediment concentrations. An example of a regression developed for streams in interior Alaska (Lloyd et al. 1987) is:

$$\log_{10} T = 0.0425 + 0.9679 \log_{10} SSC$$

where T is the turbidity (NTU) and SSC the suspended sediment concentration (mg/L). As turbidity measurements change along a downstream gradient from a sediment source (Lloyd 1987), turbidity and suspended sediment relationships only apply to specified stream reaches.

1.3 Bedload sediments

Two modes of sediment transport have been identified in stream systems: bedload transport and suspended load transport. Bedload sediment refers to that portion of the total sediment load that is carried by the streambed. Particles in this phase move by sliding, rolling, or saltating on the streambed. Bedload generally consists of coarse sand or larger sized particles (i.e., >1 mm in diameter; Leopold and Wolman 1964). Fine to coarse sand (0.125 to 1 mm in diameter) is also frequently transported as bedload but can also be carried as part of the suspended load at higher water velocities.

The transport of bedload sediment requires greater hydraulic energy than does the transport of suspended sediment. Even more hydraulic energy is required to disrupt the armour layer of gravel streams sufficiently to mobilize the sediments stored in streambed substrates (Sidle 1988). For this reason, the transport of significant quantities of bedload sediment may be limited to only a few days a year in many streams, typically during the peak of freshet when stream flows are the highest (Parker and Andrews 1985). It is during these periods that long-term changes in channel morphology and the composition of streambed substrates can occur.

1.4 Deposited sediments

The term deposited sediments refers to that portion of the sediment load that settles out of the flow and becomes associated with the streambed substrate. The factors influencing the deposition of inorganic sediments in stream systems include the characteristics of the material (particle sizes and volumes), hydraulic forces (stream size, discharge, and velocity), and the occurrence of roughness elements (Parker and Andrews 1985). Roughness elements, such as large organic debris (i.e., large trees and root systems), boulders, and bedrock outcroppings, are important because they enhance streambed stability, alter flow patterns and velocities, and create sediment storage sites in stream channels (Swanson and Lienkaemper 1978).

Local hydraulic conditions dictate which process, sediment transport or sediment deposition, is dominant in any stream reach and period (Norton 1986). Due to differences in the hydraulic conditions (i.e., resulting from differences in gradient, instream debris, etc.), there tends to be substantial variability in sedimentation and resuspension rates between reaches in any stream system (Platts et al. 1979). These differences are reflected in the composition of streambed substrates within each stream reach (Peterson 1978; Reimchen and Douglas 1978).

Various methods have been developed to evaluate sedimentation rates and streambed substrate composition in flowing waters. These include sediment traps, bedload samplers, freeze-core samplers, and core samplers (Yuzyk 1986). Evaluations of streambed substrate composition typically involve the collection and analysis of substrate samples to determine the particle size distribution (i.e., the percentage of the sample that falls within various size classes) (McNeil and Ahnell 1964). The results are often expressed as the percent of the sample that is finer than a given particle size (e.g., 6.35 mm) or as a statistic that summarizes the information on the particle size distribution (e.g., geometric mean diameter and Fredle number) (MacDonald and McDonald 1987).

2. OCCURRENCE

Suspended sediments, the deposition of sediments onto streambeds, and bedload movement all have natural and anthropogenic sources. Throughout time, extensive demand on land-use in British Columbia has made it difficult to attribute an impact to a specific source or activity. The following summarizes what is known to cause natural and anthropogenic sedimentation inputs into B.C. rivers.

2.1 Natural

Natural erosion processes of unstable geological formations are the most common source of suspended sediment to a waterbody. Sediment in British Columbia comes from the erosion of Quaternary sediments from lacustrine deposits in river valleys. In the northeast, sediments originate from Mesozoic and Cenozoic sediments in valley walls (Church et al. 1988). The rates and magnitude of erosion are dependent upon climate, geology, exposure, slope, soil type, and vegetation cover. Sediments may remain in storage in the channel bed and banks until critical velocities are exceeded, mobilizing the bedload. Movement of streambed materials in turn generates additional smaller particles through abrasion (see discussion below).

Natural levels of suspended sediments vary widely from waterbody to waterbody and can have large daily and seasonal variations in British Columbia (Singleton 1985).

2.1.1 *Background levels*

There are operational and pre-operational background levels. In the former, ambient levels of suspended sediments and turbidity and the state of the substrate composition and bedload transport are monitored at a control site upstream or outside the influence of sediment inputs or disturbances. These levels provide a relative measure with which one can assess change and may not necessarily represent the levels of a pristine or a pre-operational system. Pre-operational background levels provide a fixed measure of the historical background levels. The choice of background level will depend on the site-specific objective. For example, in the Fraser River basin, extensive use of the land of the basin has had profound effects on the natural erosion processes characterizing this basin and a comparison to pre-operational background levels may be of little value. In smaller rivers and streams, however, where anthropogenic development has not overtly changed the geomorphology of the basin, a community may wish to maintain the pristine condition of the system. As well, some waterbodies, such as transboundary waters, may need to be protected under transboundary acts and defining a fixed measure may become a necessity.

For suspended sediments and turbidity, in most lotic systems, background levels are to be monitored in clear flow periods. Figure 2 illustrates the flow hydrograph of the North Fork Flathead River near Columbia Falls. According to the figure, clear flow periods are from June to April of each year from 1975 - 1978. Clear flow must not be confused with low flow periods which gives a much smaller window of opportunity for sampling background levels than clear flow periods. Clear flow periods are determined on a site-specific basis. Even though the majority of sediment load in streams is transported during spring freshets and storm events, these high flow periods have been excluded from the determination of background levels due to the extreme variability found in relationships between suspended sediment concentrations and discharge flows (MacDonald et al. 1991). An operational definition has been given to clear and turbid flows. In this document, the terms “clear flow period” and “turbid flow period” are used to describe the portion of the hydrograph when suspended sediment concentrations are low (i.e., <25 mg/L) and relatively elevated (i.e., ≥25 mg/L), respectively. These terms have been proposed because the

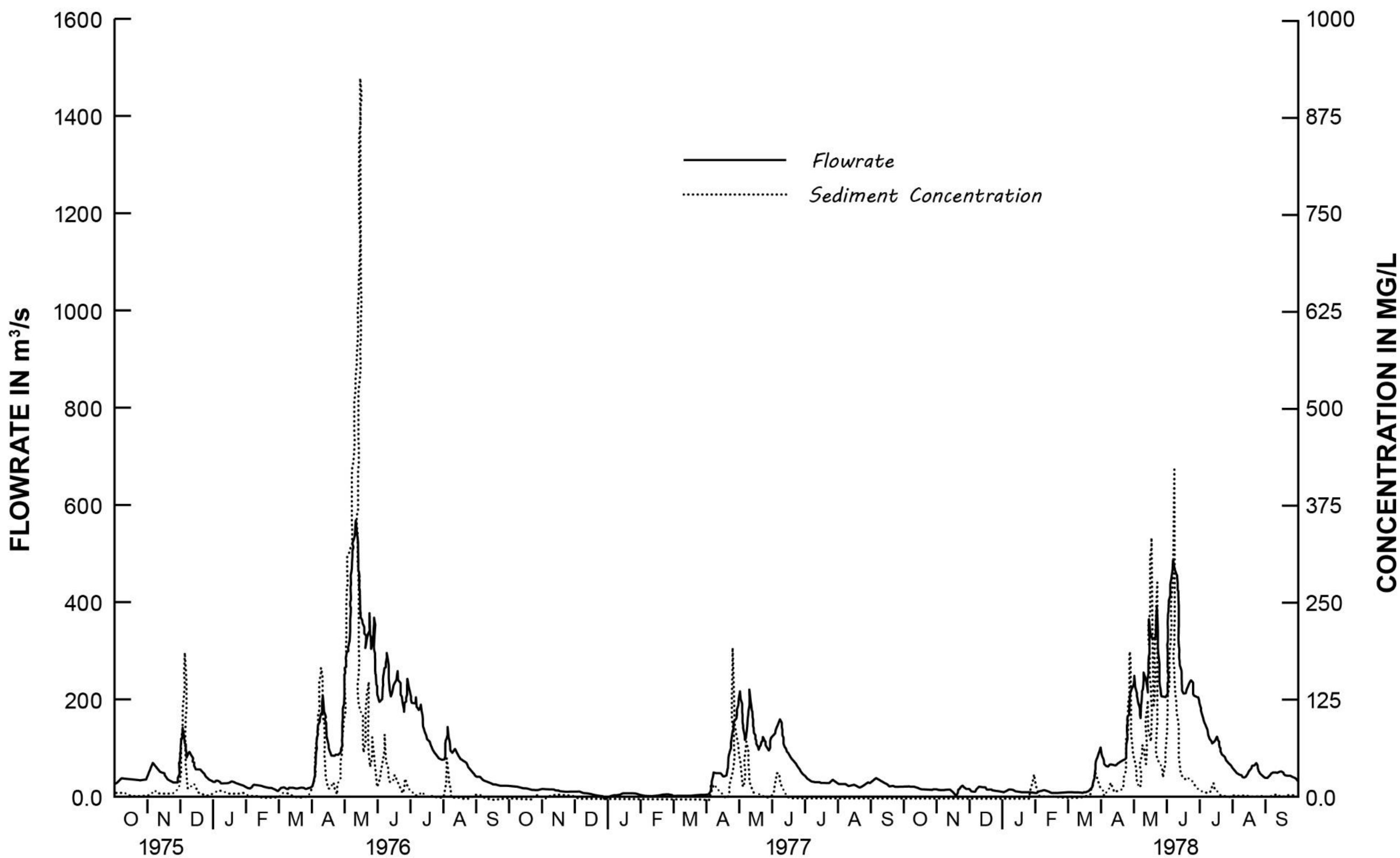

Figure 2. Flow hydrograph for the North Fork of the Flathead River.

commonly-utilized descriptive terminology (i.e., low flow and base flow, high flow or freshet flow, ascending or descending limb of the hydrograph) do not adequately identify the periods of low and elevated sediment transport in stream systems. In addition, many stream systems, such as those fed by lakes or reservoirs, run clear year-round.

Therefore, it is possible that the water quality guidelines for suspended sediments could be incorrectly applied if standard hydrological terms were utilized in this document. The clear and turbid flow periods for individual stream systems should be defined using data on the background concentrations of suspended sediment at the site-specific level.

The recommended transition value (25 mg/L) was selected by examining the hydrographs for a number of streams in British Columbia and is intended to provide an operational definition of clear flow conditions that can be applied consistently in the province.

In estuarine waters a substantial proportion of suspended sediments come from the resuspension of fine, unconsolidated sediments and detritus by wave action and currents (Appleby and Scarratt 1989). Concentrations of estuarine suspended sediments can far exceed those levels coming from freshwater sources. Apart from algal blooms, turbidity maxima in estuaries are correlated to hydrodynamic conditions and flocculation and deflocculation of river borne sediment. As sediments enter denser saline water below the saltwater/freshwater interface (halocline), there is a net upstream movement initiated by the saltwater component which resuspends particles in the upper freshwater component. This vertical mixing repeats itself and localized high concentrations (1,200 mg/L) of suspended sediment may occur (Appleby and Scarratt 1989).

2.1.2 *Sediment transport processes in B.C. river drainage basins*

In British Columbia, much of the sediments in streams and rivers originate from glacial lacustrine deposits. Some sediment is eroded from stream or riverbanks or is scoured off the bottom to be deposited further downstream or remain suspended. Suspended sediment particles that are equal to or greater than 0.5 mm (e.g., coarse sand and gravel, Table 1) will be redeposited quickly. These bedload sediments are moved by spring freshets when the hydraulic gradient is high from the pools to the riffles of streams. Generally, sediments of less than 9 mm are moved by stream currents by suspension and saltation (Culp 1996). The movement of coarse sediment particles that become entrained and deposited on downstream riffles is crucial to the construction of redds (Sidle 1988). The movement of coarse gravel (Table 1) whose grain size distribution is at the limit of bed load, occurs seldomly and only during very high stream flows (Sidle 1988). These sediment particles are protected by armoring and imbrication. In high flow situations, bottom velocities in pools approach or exceed those in riffles whose armor is transported to the next riffle (Jackson and Beschta 1997). During small storms, the velocities over the riffle far exceed those in pools thereby entraining sand-size sediment particles (Jackson and Beschta 1997) that settle in downstream pools. Silt and clay sediment particles (Table 1), perhaps the most pernicious of sediments particle types for aquatic biota, can remain in suspension for much longer periods of time as the upward component of fluid turbulence in streams is often just enough to keep these from being deposited. These suspended sediments can also be referred to as the wash load fraction of the overall suspended load. Mechanisms regulating deposition of sediment particles are gravity which controls the suspended particle settling velocity (Table 1) and entrapment of particles within the interstitial areas of stream beds (Anderson et al. 1996).

Specific sediment yield is the amount of sediment recorded per unit area of land surface. In British Columbia, glacial headwaters contribute the most to sediment specific yields which is achieved, in many rivers, in the first three months of spring freshet (Church et al. 1988). Specific sediment yield, contrary to convention, does not decline downstream in a drainage basin as the area drained increases. This is the case for basins throughout B.C. except for large river systems (e.g., Fraser). Part of the sediment that has been mobilized from the land does not go back into storage at field edges, in ditches, in stream channels, and on floodplains downstream (Church et al. 1988).

Basins that drain steep mountain valleys typically have high sediment yields. Examples of such small river systems are the John Creek and Meadow Creek, which have cut into fine-grained schists and chlorite schists together with a small amount of limestone and quartzite (Fyles 1997). Other small rivers, such as the lower Beatton and Kiskatinaw rivers, have high sediment yields due to poorly lithified mudstone, siltstones and shales in steep bluffs with many landslides (Church et al. 1988). Redfish Creek and Laird Creek in the West Kootenays have low yields because they flow through granitic bedrock and there is a relatively low level of geomorphic activity in the area (Jordan 1996). Suspended sediment yields for small non-glacierized watersheds range from 3 – 70 tonne/km^2/year and from 44 – 470 tonne/km^2/year for larger rivers (Church et al. 1988).

Sediment transport models relating sediment load and streamflows exhibit high variability especially in small streams. Empirical relationships are often the most practical sediment transport models. With the use of such models, it was found that the characteristics of suspended sediment rating curves in the Takla Lake region were influenced by flow stage and sediment production (Cheong et al. 1995). Correlations between stream discharges and sediment concentrations varied tremendously; higher correlations, however, were found at high discharges (>1.0 m^3/s, e.g., snowmelt). At low flows, spawning salmon were found to contribute to the variability of suspended sediment concentrations by winnowing fine sediment from the gravels during construction of redds (Cheong et al. 1995). Other factors affecting the variability in suspended sediment concentrations are pools, gravel bars, and debris jams acting as sediment storage sites during low flows and as supply sources during high flows. For the Takla Lake region, hysteretic effects relating the rising and falling limbs of high flow effects showed different characteristics (Cheong et al. 1995).

River-ice break-up can dramatically increase downstream water levels and velocities. Ice scour of bed and banks can significantly augment the quantities of suspended sediments that induce changes in stream morphology and fish habitat (Milburn and Prowse 1996; Newcombe and MacDonald 1991).

2.2 Anthropogenic

Anthropogenic activities such as forest harvesting, road building, construction, dredging and-gravel pit operations, can cause marked changes in the physical, chemical, and biological characteristics of the watercourses located nearby and those located downstream. The environmental impact is often attributable to the type of activity. The following discusses entrainment characteristics of suspended sediment particles by these anthropogenic activities.

2.2.1 *Logging*

Forested watersheds are known to contribute significantly to sediment export (Slaney et al. 1977a; Beshchta 1978; Webster et al. 1990; Doeg and Koehn 1990a; Martin and Hornbeck 1994). With timber harvest, the nature, extent, and severity of the environmental effects are a function of the area cut as a percent of the given watershed, the species of tree harvested, the density of the stand harvested, and the local topography of the area affected (Verry 1986). Other factors that influence the impact of forestry activities are, for example, soil type, slope (Sullivan 1985; Carson 1996), vegetation and the width of riparian buffer strips (Davies and Nelson 1993). The removal of riparian vegetation in forested areas can reverse organic matter contributions from allochthonous to autochthonous sources (Bilby and Bisson 1992; Hachmöller et al. 1991). Fish production in clear-cut sites may temporarily be increased due to such changes in the types and quantities of organic matter entering streams. This organic matter can range in size from a dissolved state to coarse particulate organic matter (>1.0 mm) (Bilby and Bisson 1992). Sediment traps in ephemeral streams of logged areas can collect two to three times more sediments ranging from less than 125 to 500 µm (Davies and Nelson 1993). Recovery of sediment profiles in streams to their background levels can take up to a five-year period (Davies and Nelson 1993).

Streams flowing through forested land are more variable in width and morphology than streams flowing through non-forested land (Bilby 1981). Organic debris dams often found in non-forested areas play a key role in modifying stream channel morphology and collecting sediments. The dams dissipate the energy of the water and reduce the capacity of streams to transport sediment. Removal of organic dams dramatically increases the transport of fine sediments and, to a lesser extent, that of coarse sediment in small stream ecosystems (Bilby 1981). Disturbance of forest floors can channelize water thereby increasing its velocity and consequently its sediment load (Martin and Hornbeck 1994). This occurred on the Carnation Creek, British Columbia as rapid flows eroded reaches without buffer zones (Culp 1996). Clear cutting in Carnation Creek caused an augmentation of the transport of <9 mm sediments to the substrate by suspension and saltation. Winter scouring was more severe in stream reaches without buffer zones (Culp 1996). Streambank erosion due to timber harvest not only increases the sediment load but reduces the diversity of substrates which contributes to fish habitat simplification (Reeves et al. 1993).

The water quality parameters that forest management activities are likely to affect are turbidity, suspended solids, color, dissolved oxygen, temperature and nitrate nitrogen (MacDonald et al. 1991; MacDonald and Smart 1993). Most of these parameters are a direct or indirect measure of sediment load. Criteria that affect stream channel morphology (e.g., bedload, particle size) are critical to designated water-uses (MacDonald and Carmichael 1996).

The extent of suspended sediment increases in streams is site-specific and highly dependent on the type of logging operation (Doeg and Koehn 1990b). Extrapolating monitoring results to other areas with different logging practices is difficult and not always recommended. First-order streams of the Pacific Northwest are subject to disruptive, periodic, natural events that may overwhelm more frequent but smaller changes due to forest management activities (MacDonald and Smart 1993). Another site-specific concern is that some activities may have relatively minor effects in one climate and adverse effects in another. Furthermore, it is important to identify sites with high erosion rather than estimating total erosion volumes for forest plots under management. For example, in California it was found that 82% of the erosion came from 24% of the plots (Peters and Litwin 1983). Probability estimates for critical rates of erosion were established in cubic yards per acre (Peters and Litwin 1983; Rice and Lewis 1991).

2.2.2 *Gravel roads*

Gravel roads in logging areas are made of compacted soil, sand, and gravel in a variety of proportions. The more an area is prone to erosion, due to climactic, hydrological, and geomorphological factors, the more sediment transport will occur from the roads. A road mainly composed of fines in a prone area will contribute excessive sediment to nearby streams. Other critical factors are road design, length and frequency of use, maintenance activities, surfacing material, and drainage characteristics (Reid and Dunne 1984; Preston 1996; Bilby et al. 1989; Anonymous 1996; Martin and Hornbeck 1994; Grayson et al. 1993). For further information on British Columbia coastal and interior watershed assessment procedures on roads and related stream sediment inputs, refer to the 1995 Ministry of Forests and Ministry of Environment Lands and Parks procedure guidebooks on these subjects (CWAP, IWAP) (Singleton pers. com. 1997. Victoria, B.C.).

When heavy load vehicles stopped using forest roads in the state of Washington, the sediment load in streams was reduced from 2,000 to 10 tonne/ha/year (Reid and Dunne 1984). Of this sediment, the majority was silt and clays (<0.004 mm); in steeper terrain, however, higher proportions of coarser particles can be found (Bilby et al. 1989). Deposition on the streambed is positively correlated with particle size (Bilby et al. 1989). Delivery of fine road sediment to larger streams often depends on its transport through smaller streams and whether mainline roads have settling ponds to reduce sediment load during instances of high suspended sediment concentration and low-to-moderate discharges (Bilby et al. 1989).

Road-related sources of sediment in the Pacific Northwest include, for example, landslides, erosion of back-cuts along hillsides, side-cast erosion, gully formation, and road-surface erosion (Anonymous 1996). Surface erosion is the most severe during the first year after road construction (Milburn and Prowse 1996). It was also found that road crossings were associated with large increases in infiltration of suspended fine sediments in adjacent riffles 30 to 50 years after construction (Davies and Nelson 1993). Older roads that used side-cast techniques and poorly protected cross-drains in steep terrain are often associated with landslides and gully erosion. Overall, in the Pacific Northwest, it was concluded that 60% of road-related sediment production was caused by landslides while 18-28% accounted for erosion from road surfaces (Anonymous 1996). Chapman Creek, British Columbia is subject to landslides (road failures) which have resulted in periodically high levels of turbidity (700 NTU) over the years, a major concern of the Regional District Water Board (Thomson 1987; Carson 1996). Because of the limited storage capacity of the reservoirs, an event could often be associated with turbid waters and dirty tap water. Carson (1996) showed in a preliminary assessment of Chapman Creek that roads and road destabilized terrain were not always the dominant cause of turbidity and that characteristics of the watershed such as ongoing sedimentation and hydrological processes had to be considered.

2.2.3 *Other sources*

Other major sources of anthropogenic sediment loading in British Columbia streams are contaminant and navigational dredging, construction (urbanization), agriculture, industrial wastewater discharge, and mining activities. Dredging, agriculture and mining activities are discussed below.

Dredging is an activity that removes bottom sediments from aquatic systems, for example, for navigational (Ankley et al. 1992) or mining purposes (Scannell 1988) or for environmental clean-up due to accumulated discharges from local industrial activities (Miles 1995). Older technologies such as the use of cranes were inappropriate as they excessively stirred bottom sediments and worsened the environmental impacts. Today's technologies use horizontal auger hydraulic suction type dredges that minimize the amount of sediment being resuspended (Miles 1995). Even with these techniques, a lotic system being dredged will maintain a level of suspended sediment that may result in the silting of gravel spawning areas of downstream reaches (Chapman 1992). More subtle effects of erosion may be regressive erosion upstream of the dredging that may prevent fish migration (Chapman 1992). In British Columbia, large-scale dredging occurs on an annual basis on the Fraser River. There are also other large capital works projects requiring dredging (e.g., Roberts Bank Coal Port project, Vancouver Harbour, the Fraser River, and various marinas throughout the province).

Agricultural drainage augments both sediment load and stream conductivity (Welch et al. 1977; Lovejoy et al. 1985). Agriculture has a moderate demand on the land-use in British Columbia (950,000 ha, 1% of the land area of the province) and can aggravate natural erosion rates. The headwaters of many agricultural river basins come from regions of high erosion with steep slopes, heavy rainfall, and erodible rock. It is difficult, however, to apportion the contribution of eroded sediment loading to a river caused by diffuse agricultural inputs. Furthermore, agricultural activities tend to occur in low-land areas of watersheds where flow rates have substantially decreased and changes in suspended sediment concentrations are not easily discernable. In addition to inorganic sediments, agriculture contributes organic particulates which affects the overall food supply (Lenat 1984). A common consequence is an increase in turbidity measures due to stimulation of plankton growth. Although soil conservation efforts are being implemented in some watersheds (e.g., Fraser River basin), complementary water quality monitoring data are not available.

Apart from the toxic inputs of mine tailings entering aquatic systems, mining activities can substantially contribute to the load in suspended fine sediments in British Columbia. Coal mining operations on the Fording River have been shown to produce relatively more fine sediment (20% more particles <2.00 mm and 40% more <6.35 mm compared to upstream sites) and to manifest differences in the geometric mean diameter and in the Fredle Index at downstream sites when compared to upstream sites (MacDonald and MacDonald 1987). Placer gold mining has several systems of mining which release sediment to water. A popular method is open cut placer mining which removes vegetation, topsoil, and gravel above placer deposits. Mean turbidity in mined areas of Alaska has been recorded at 445 NTU as compared to mean clear areas at 1.4 NTU (Scannell 1988). Increased sediment load due to mining has similar physical effects that other anthropogenic activities have; for example, mining can change channel morphology (Dames and Moore Consultants 1986), reduce water column water exchanges (Bjerklie and LaPerriere 1997), and decrease the particle size of the bottom substrate (Weber 1986).

3. DRINKING WATER

3.1 Water treatment

Total treatment of water with high turbidity involves the addition of polyelectrolytes to assist in the sedimentation of silt and other suspended matter in pre-sedimentation basins. Following this step, chlorine is added and, depending on the carbon content of the raw water, other disinfecting techniques such as ozonation can be used. Alum and activated silica are then added so that coagulation and flocculation can take place. Sedimentation follows in a sedimentation basin. A sand filter constitutes the last particulate removal step. Chlorine (also chloramines or chlorine dioxide) is then added to provide a residual in the distribution system (Huck et al. 1993). This treatment technology will produce drinking water with turbidity levels less than 1 NTU.

3.2 Effects

Suspended sediments in drinking or food processing water supplies can cause both health and aesthetic effects. Excessive suspended sediment may be visible and, therefore, aesthetically objectionable. High levels of suspended sediments can also shield pathogens from the effects of disinfection. Organic suspended matter can act as a source of nourishment thereby promoting the growth of micro-organisms in distribution systems (Singleton 1985).

Excessive suspended sediments released to raw water supplies that met drinking water criteria without treatment can require the addition of treatment facilities (Singleton 1985).

Excessive increases in concentrations of suspended sediments may subject existing treatment facilities to additional loads which they were not designed to handle. These additional loads can increase chlorine demand, increase quantities of sludge, and necessitate more frequent backwashing of filters. Also, in chlorinated waters, chlorine can react with the increased concentration of suspended sediments to cause the formation of trihalomethanes and an increase in heavy metals (Health and Welfare Canada 1980) as well as taste and odour problems. Consequently, substantial increases in treatment levels or costs for maintenance and operation of treatment facilities can be required to overcome excessive increases in suspended sediment (Singleton 1985).

3.3 Literature criteria

Criteria, objectives, and standards to protect consumers from excessive suspended matter in drinking water supplies from other jurisdictions have not changed since their last compilation by Singleton (1985). These are summarized in Tables 2 and 3. This compilation permits the comparison of approaches used by other jurisdictions for consideration in developing criteria for British Columbia waters (Singleton 1985).

The most used variable to protect drinking water supplies from excessive suspended matter is turbidity because of the aesthetic importance of clarity to drinking water. Also, for health considerations, turbidity is a reasonable indicator of the concentration of suspended matter present in the water (Singleton 1985).

Turbidity criteria have been established by other jurisdictions for raw drinking water with (Table 2) and without treatment. Raw water refers to water before it is drawn from its source for use as a drinking water supply. Raw drinking water without treatment refers to water that will not receive treatment for the removal of suspended matter before consumption. Raw drinking water with treatment refers to water that will receive some level of treatment for the removal of suspended matter before consumption (see Section 3.1 above).

3.4 Recommended Guidelines

3.4.1 *With treatment to remove particulates*

Several jurisdictions, including the provinces of Saskatchewan and Alberta and the states of Idaho, Alaska, and Washington, allow some increase (usually 5 to 25 NTU) above natural background turbidity in raw water (Table 2). The British Columbia Ministry of the Environment Lands and Parks watershed guidelines for raw water permit a turbidity increase of up to 5 NTU over environmental background. The permitted level of turbidity in raw water in Manitoba is dependent upon the level of treatment applied to comply with the acceptable limit for finished drinking water (5 NTU) established by Health and Welfare Canada (Health and Welfare Canada 1980; Singleton 1985).

Rationales for the criteria from other jurisdictions were not available, but the criteria appear to be based on the presumption that a small to moderate increase in the turbidity of raw water will not overburden treatment facilities (Singleton 1985). The rationale for the British Columbia watershed guidelines for raw water was to maintain a level which would permit the simplest form of treatment to make the water wholesome. This approach is acceptable for raw water which normally requires some level of treatment to reduce turbidity to acceptable levels (Singleton 1985).

Non-filterable residue criteria have been established by some jurisdictions (Table 4), but rationales for the choice of levels were not available. According to the "Supporting Documentation" for Canadian Drinking Water Guidelines (Health and Welfare Canada 1980) the turbidity limit was chosen primarily for aesthetic reasons but decreased drinking water clarity can also serve as a signal indicating excessive suspended matter which may create health problems (Singleton 1985).

The guideline for raw drinking water with treatment, established here, is a refinement of the British Columbia watershed guidelines for raw water, which were developed by a task force comprised of representatives from several provincial ministries in 1980.

<u>Turbidity guideline for raw drinking water with treatment to remove particulates:</u>

For raw waters that normally require some form of treatment to reduce natural turbidity to a level which complies with the standard for finished water (≤5 NTU) in British Columbia, induced turbidity should not exceed 5 NTU when background turbidity is ≤50 NTU. When background is >50 NTU, the induced turbidity should not be more than 10% of background (Singleton 1985). Induced turbidity refers to that turbidity over natural levels which is caused by anthropogenic activities.

3.4.2 *Rationale*

The guideline for turbidity in raw drinking water requiring treatment to remove particulates is a somewhat arbitrary choice to ensure reasonable water quality but also to guard against substantial increases in treatment costs. In addition, this guideline is reasonably consistent with levels established by British Columbia's neighbours and, therefore, similar protection would be afforded to transboundary waters. An absolute limit on the allowable increase in turbidity as used by some jurisdictions (e.g., 10% increase up to a maximum of 25 NTU) was not adopted because 10% is a relatively small increase that will provide the desired protection and such a limit would only apply at high background turbidity levels (e.g., >250 NTU) when there would be little practical or measurable difference between a 10% or a 25 NTU increase (Singleton 1985).

Table 2. Turbidity criteria for raw drinking water with treatment.

Criteria Statement	Criteria Values	Reference
Good source of water supply requiring usual treatment such as filtration and disinfection: 10-250 turbidity units	10-250 turbidity units (units not given)	McKee and Wolf 1963, (California)
Poor source of water supply, requiring special or auxiliary treatment and disinfection: over 250 turbidity units	>250 turbidity units (units not given)	McKee and Wolf 1963, (California)
Turbidity in water should be readily removable by coagulation, sedimentation, and filtration; it should not be present to an extent that will overload the water treatment plant facilities and it should not cause unreasonable treatment costs. In addition, turbidity should not frequently change or vary in characteristics to the extent that such changes cause upsets in water treatment plant processes		U.S. EPA 1972
Surface waters not to exceed more than 25 JTU over natural turbidity	25 JTU increase	Environment Saskatchewan 1975; Alberta Department of the Environment 1977
Shall not exceed 5 NTU above natural conditions when the natural turbidity ≤50 NTU, shall not undergo more than a 10% increase when the natural condition is >50 NTU, and shall not exceed a maximum increase of 25 NTU	5-25 NTU increase	Alaska Department of Environmental Conservation 1979; Idaho Department of Health and Welfare 1980
Criteria range from no increase above naturally occurring turbidity to 10 NTU above background, or which may be harmful or create a nuisance, depending upon the treatment level.	0-10 NTU increase	Montana Health and Environmental Sciences 1980

Criteria Statement	Criteria Values	Reference
Less than 5 JTU over environmental background	5 JTU increase	British Columbia Ministry of Environment 1980b
Turbidity shall not exceed 5 NTU over background when background turbidity is ≤50 NTU or shall not undergo more than a 10% increase when background is >50 NTU	5 NTU increase	State of Washington 1982
Turbidity shall not exceed 5 NTU over background for lakes	5 NTU increase	State of Washington 1982
Recommends the use of separate standards for raw waters that require treatment		Alaska Department of Environmental Conservation 1982
Free from materials that produce turbidity in such a degree to be objectionable or to impair any beneficial use		Manitoba Department of Environment 1983

Table 3. Non-filterable residue criteria for raw drinking water with treatment.

Criteria Statement	Criteria Values	Reference
Not to be increased by more than 10 mg/L over background values	10 mg/L increase	Environment Saskatchewan 1975
No measurable increase in concentrations of sediment above natural conditions	0 mg/L increase	Alaska Department of Environmental Conservation 1979
No man-caused suspended matter of any kind in concentrations causing nuisance or objectionable conditions or that may adversely affect designated or protected beneficial uses		Idaho Department of Health and Welfare 1980
Suggest that it is illogical to permit a turbidity increase over background without allowing for a corresponding increase in non-filterable residue		Alaska Department of Environmental Conservation 1982

4. AQUATIC LIFE (FRESHWATER, ESTUARINE AND MARINE)

4.1 Turbidity and suspended sediment effects

The transport and deposition of fine inorganic sediment in stream systems are natural processes that depend on a number of site-specific factors. Certain land use activities, such as logging and road building, tend to accelerate sediment production. In general, the deposition of fine sediment in stream ecosystems is detrimental to aquatic organisms because of reductions in streambed substrate composition, permeability, (Young et al. 1991), and stability (Cobb et al. 1996). These alterations in the physical environment can decrease egg-to-fry survival rates in fish (Valiela et al. 1987), affect stream and benthic macroinvertebrate production (Noel et al. 1986; Erman and Erman 1984; Culp 1996), and periphyton communities (Noel et al. 1986). Even greater habitat degradation can occur under reduced sediment transport regimes if flushing flows are decreased or eliminated (Burt and Mundie 1996; Nelson et al.

1996). The following sections discuss suspended sediment effects on plant, invertebrate and animal life in aquatic ecosystems.

4.1.1 *Aquatic plants*

The effects of suspended sediments on algae are associated with reduced primary productivity (Singleton 1985). Increased or excessive suspended sediments can reduce productivity by 1) inhibiting photosynthesis due to decreased light penetration; 2) physically smothering benthic communities; 3) removing periphyton by scouring; and 4) affecting community composition (Singleton 1985). It is noted, however, that temporary resuspension (e.g., dredging, logging) of sediments and nutrients in the water column can temporarily augment algal productivity (Bilby and Bisson 1992).

Natural or anthropogenic events leading to disturbances in aquatic systems and elevated suspended sediments affect whole ecosystems (Lloyd et al. 1987). Effects on algae are the first consequence of perturbation. For example, logging practices may produce shifts in the allochthonous and autochthonous energy inputs (Section 2.2). These shifts have the effect of changing the amount and quality of available food resources in streams (Culp 1996). With increased suspended sediments and nutrients and less shading of streams resulting in higher temperatures and more light available for photosynthesis, algal biomass may flourish temporarily giving rise to increased invertebrate (Behmer and Hawkins 1996) and fish abundance (Bilby and Bisson 1992). This may not be the case, however, as other factors essential to primary productivity (e.g., phosphorus) may be limiting (Shortreed and Stockner 1996). In a New England study where logging practices brought about changes in stream bottom particle size, it was found that green algae and flowering plants were more abundant in clear-cut streams than in reference streams where diatoms dominated (Noel et al. 1986). In logged areas of Vermont and New Hampshire, streams had relatively higher amounts of sand and gravel than reference streams, which had more of a pebble, cobble, and boulder bottom composition (Noel et al. 1986).

It has been shown in laboratory experiments that mineral particles (e.g., silica, kaolin, bentonite) affect many physical and biotic processes such as algal-clay flocculation and sedimentation (Threlkeld and Soballe 1988). These in turn would have an effect on trophic interactions.

In summary, plant growth is reduced by increasing levels of suspended sediments (Newcombe and Jensen 1996a); plants, however, are not the most sensitive organisms affected by excessive suspended sediments. Periphyton communities are susceptible to scouring of suspended sediments. No LC_{50} values were found in the literature (Appendix 2). Other specific severity-of-ill-effects on phytobiota are discussed in Section 4.6.

4.1.2 *Aquatic invertebrates*

Invertebrate populations are dependent on the condition and abundance of primary producers. Their numbers and composition will be affected if suspended sediment concentrations impact periphyton communities. More direct effects of suspended sediments on invertebrates include 1) physical habitat changes due to scouring of stream beds and dislodgement of invertebrates; 2) smothering of benthic communities; 3) clogging of interstices between gravel, cobbles and boulders affecting invertebrate microhabitat; and 4) abrasion of respiratory surfaces and interference of food intake for filter-feeding invertebrates (Singleton 1985).

A study in Carnation Creek, British Columbia, demonstrated the effects of streambank clear-cutting on benthic macroinvertebrate communities (Culp 1996). There were decreases in the abundance of benthos due to increased inputs of fine sediments (amounts not given) originating from clear-cut areas without adequate buffer zones around streams. It was shown that sediment saltation is a significant mechanism by which macroinvertebrates are scoured from the streambed substrate thereby reducing macroinvertebrate densities (Culp et al. 1985; Culp 1996). Another study in Padden Creek, Washington, emphasized that reduced sediment size and stream velocity are major factors influencing the macroinvertebrate community structure in streams. Upper reaches of logged streams have an abundance of shredder and predator taxa whereas downstream sites are lacking these (Hachmöller et al. 1991). Addition of fine sediments to a coastal stream from a drinking-water filtration plant had similar effects (Erman and Ligon 1988). It was also suggested that sedimentation ponds were insufficient to protect macroinvertebrate diversity in streams when impoverishment of benthic communities was due to deposition of particles on benthic habitats and particle movement at the streambed surface (Vuori and Joensuu 1996).

Embeddedness, or the degree to which the dominant particles are surrounded by fine inorganic sediments, and the presence of coarse woody debris were found to have the strongest correlations with macroinvertebrate assemblage richness and composition (Richards and Host 1994). Other subtle effects of suspended sediment (silt) deposition on streambeds include the elimination of the predation of stoneflies on benthic invertebrates, demonstrating that turbidity can override the effects of predation by predators in stream communities (Peckarsky 1985).

The macroinvertebrate community structure in streams is correlated with the average size of particles in the stream's substrate (Erman and Erman 1984). When median particle size was held constant, heterogeneity of substrate composition was not an important component structuring macroinvertebrate communities. Thus, an increase in the deposition of fines could create an imbalance of the median particle size and affect species abundance and richness.

Studies reporting the effects of suspended sediments on aquatic invertebrates are more abundant than those for aquatic plants (Appendix 2). The information gathered suggests that invertebrates are as sensitive to high levels of suspended sediments as salmonid fishes (Newcombe and MacDonald 1991). The LC_{50}'s range from 720 to 5,108 µg/L (Appendix 2). Other specific severity-of-ill-effects on invertebrates are discussed in Section 4.6.

4.1.3 *Fish*

There are a number of direct and indirect ways by which excessive suspended sediments levels in water affect fish. Effects on trophic interactions at the primary and secondary level of productivity will indirectly affect fish community structure. Direct effects include clogging and abrasion of gills, behavioural effects (e.g., movement and migration), resistance to disease, blanketing of spawning gravels and other habitat changes, the formation of physical constraints disabling proper egg and fry development, and reduced feeding (Singleton 1985).

A review of the effects of sediment release on fish and their habitats in British Columbia is provided by Anderson et al. 1996. Much of the information was taken from Newcombe (1994a). The following discussion focuses on investigations (1984-1997) that have contributed information on the effects of suspended sediment on fish.

Newcombe (1994b) described the more subtle and difficult to measure behavioural effects as easily reversible and not long-lasting. A study by Berg and Northcote (1985) demonstrated that the territorial, gill-flaring, and feeding behavior of juvenile coho salmon was disrupted by exposure to suspended sediment pulses. High turbidities (30 and 60 NTU) broke down the dominance hierarchies with territories not being defended. Social organization was re-established with the return of normal turbidities (0-20 NTU) (Berg and Northcote 1985). It was shown by an analysis of feeding and reproductive guilds that fish species with similar ecological requirements had a common response (e.g., decreased diversity) to habitat degradation by siltation (Berkman and Rabeni 1987). In another investigation, it was shown that the relationship between fine sediment and chinook salmon abundance in streams during the winter is an indication of the importance of winter habitat to their production. This suggests a cause for the fall-winter exodus in streams with high sediment loads (Hillman and Griffith 1987). The authors suggest that as the interstitial spaces between cobble are filled, juvenile fish may leave redds or take cover in less protected areas.

Growth of fish can be impaired with an excess of suspended sediment via effects through the food chain. In a laboratory experiment where turbidity was simulated with clays, kaolinite, and bentonite, feeding of 30-65 mm long steelhead (*Salmo gairdneri)* and coho salmon (*Oncorhynchus kisutch)* was affected giving rise to growth impairment and emigration of fish from experimental channels (Sigler et al. 1984). Turbidity as low as 25 NTU caused a reduction in fish growth. The quality of the light may be altered as large amounts of suspended particles intercept the wavelengths used by fish, thereby reducing their ability to see and secure food (Sigler et al. 1984). Similar effects were shown on under-yearling Arctic graylings (*Thymallus arcticus)* exposed to suspended placer mining sediments (McLeay et al. 1984). Concentrations of suspended sediments that significantly reduced fish growth ranged from 100 – 1,000 mg/L. The time required to detect and consume surface drift for native fish (previously unexposed to sediment) increases with increased suspended sediment concentrations. Other symptoms of stress were a palish colouration of fish and a decrease in tolerance to a reference toxicant as compared to controls (McLeay et al. 1984).

Physiological effects other than growth reduction include alteration in blood chemistry (Servizi and Martens 1987) and histological changes (e.g., gill damage and phagocytosis of sediment) (Newcombe and Jensen 1996b). Slightly elevated hematocrit counts (2% above controls), an indication of anoxia, were observed in sockeye salmon exposed for nine days to 1,500 mg/L of Fraser River fines (Servizi and Martens 1987). As well, plasma glucose, an indication of secondary stress, was elevated by 150% and 39% resulting from suspended concentration exposures of 1,500 and 500 mg/L. Other effects of suspended sediments (kaolin clay and volcanic ash) on blood chemistry of salmonid species may be temporarily elevated levels of plasma cortisol (max. 1,367% increase at 24 h) and reduced resistance to pathogens (Redding et al. 1985). Osmoregulatory performance of fish from fresh to salt water seems to be unaffected (Redding et al. 1985; Servizi and Martens 1987).

Histological effects of suspended sediments on gill apparatus of fish are well documented (Singleton 1985; Anderson et al. 1996). Histopathological effects of high concentrations of suspended sediments (>1,400 mg/L; <74 – 740 µm) on fish gills include gill hypertrophy, necrosis and gill lesions due to protozoan infection (Servizi and Martens 1987; Goldes et al. 1988). Based on their experimental results the authors suggest that Early Stuart adult sockeye could encounter stress-causing concentrations of suspended sediments during their spawning migration in the Fraser River. The 96 h LC50's to sockeye salmon ranged from 1,674 to 17,560 mg/L (Servizi and Martens 1987).

Generally, effects are observed when the particle size of sediments are approximately 75 µm which matches the space between gill lamellae. Not as well documented, however, are the effects of angularity and hardness of sediment particles to fish gills. Underyearling coho salmon have a reduced tolerance to an increase in angularity and particle size (Servizi and Martens 1987; Servizi and Martens 1991; Appleby and Scarratt 1989). Natural sediments, however, may be coated with organic material which would reduce the angularity of the particle (Appleby and Scarratt 1989).

Tolerance to suspended sediments may have some relationship with the temperature of the water. While low temperature favours oxygen saturation and a fish's tolerance to suspended sediments, it may also lower the capacity of fish to clear the gills of particles due to inadequate cough reflexes and ventilation rates (McLeay et al. 1987; Servizi and Martens 1991). Sediments accumulate in the buccal cavities of fish when these become too fatigued to continue clearing particles by coughing (Servizi and Gordon 1990). In their experiment the 96-h LC_{50} for sockeye and chinook salmon were 17,600 and 31,000 mg/L, respectively.

Fish eggs are very susceptible to the settling of suspended particles. Fine particles can disrupt normal gas exchanges and metabolic wastes between the egg and water with coverings as thin as a few millimeters (Anderson et al. 1996). Concentrations as low as 7 mg/L for 1,152 hours are enough to produce a mortality rate of 40% on rainbow trout (Slaney et al. 1977b). Juvenile and adult fish are more resilient to high concentrations of suspended sediment than the egg or larvae of fish (Newcombe 1994a) as these early life stages cannot use avoidance behaviour (Anderson et al. 1996). Marine fish LC_{50}'s are similar to those for fresh water fish (Cyrus and Blaber 1987). Lethal effects of turbidity on fish require concentrations far above the highest naturally occurring turbidity concentrations. Physical effects of suspended solids on marine and estuarine fish and shellfish have been summarized by Appleby and Scarratt (1989). Relevant data are included in Appendix 1 and 2.

In summary, fish (all life stages) are the most sensitive aquatic organisms to low levels of suspended sediment. The LC_{50}'s for adults and juvenile fish range from 270 to 35,000 mg/L. In estuarine waters, LC_{50}'s range from 189 to 330,000 mg/L (Appendix 1). Other specific severity-of-ill-effects on fish are discussed in Section 4.6.

4.2 Bedload sediments

The effects of bedload sediment on fish and aquatic life are poorly understood. This is primarily due to the difficulty associated with the measurement of bedload transport in stream systems. Some inferences, however, can be drawn from the information that is currently available on the effects of bedload sediments and on the nature of this sediment transport process.

4.2.1 *Aquatic plants*

No information was located on the effects of bedload sediment on the periphyton communities that occur in flowing water. It is likely, however, that increases in bedload transport rates would increase the rate of scouring of periphyton and render some of the substrate too unstable to support algal colonization. Such effects would lead to the lower levels of algal biomass and, possibly, shifts in community structure. It is likely that recolonization would occur soon after bedload transport rates and suspended sediment concentrations return to baseline levels.

4.2.2 *Aquatic invertebrates*

A review of the scientific literature failed to identify quantitative information on the effects of bedload sediment on aquatic invertebrates. Nonetheless, the results of a single study showed that increased bedload transport can have detrimental effects on the aquatic invertebrates. Specifically, increased bedload transport in stream riffles resulted in catastrophic invertebrate drift rates, substantially reduced invertebrate densities (i.e., by 50% in 24 hours), and altered community structure (Culp et al. 1985).

4.2.3 *Fish*

Little information was located on the effects of bedload sediment on freshwater fish. The results of several studies, however, indicate that increased transport of bedload sediments can adversely affect fish and fish populations. For example, populations of brown trout and rainbow trout in a Michigan stream were reduced following a 30-day exposure to 56 mg/L of bedload sediment (Alexander and Hansen 1983). Significant degradation of steelhead trout and salmon habitat was also reported in a coastal stream following a 10-day pollution episode during which bedload sediment concentrations ranged from 100 to 3,200 mg/L (Coats et al. 1985)

Although specific data were not located, those species that are closely associated with streambed substrates (e.g., bull trout) are more likely to be adversely affected by bedload sediments than would those species that are more evenly distributed throughout the water column (e.g., cutthroat trout).

Therefore, it is important to consider the structure of fish communities and habitat partitioning when evaluating the potential effects of increased bedload sediment rates.

4.3 Streambed substrate and deposited sediments

The transport and deposition of fine inorganic sediments are natural processes in stream systems. Certain land use activities, however, have the potential to accelerate sediment production and, thereby, increase the risk of problematic levels of sedimentation (Reid and Dunne 1984). To adequately assess the potential impacts of such activities on fluvial ecosystems, detailed information is required on the effects of deposited sediments on aquatic organisms. The following discussion summarizes the available data on the effects of deposited sediment on fish and aquatic life.

4.3.1 *Aquatic plants*

The deposition of fine sediment onto streambed substrates can smother periphyton and cover the stable substrates to which algae attach. Accumulations of fine sediment can also render portions of the streambed too mobile to support periphyton communities, thereby eliminating much of the primary productivity (Nuttall 1972). Primary production, however, is usually eliminated by turbidity and scouring well before this condition occurs (Langer 1980).

4.3.2 *Aquatic invertebrates*

Information from a number of studies conducted throughout North America indicates that the deposition of fine inorganic sediment in stream ecosystems can be detrimental to aquatic invertebrates. For example, Tebo (1955) reported significant reductions in the densities of macroinvertebrates in a small Appalachian stream due to smothering by sediment from logging operations. Similarly, substantial decreases in macroinvertebrate production were reported after fine sediments were deposited into a small stream from a rock quarry (Gammon 1970). Benthic macroinvertebrate biomass was also substantially reduced in stream reaches in central British Columbia which had high levels of sediment deposition, primarily as a result of logging activities. It is likely that short-term changes in population densities result from increased rates of invertebrate drift (Culp et al. 1985).

In addition to reducing densities, longer-term exposure to deposited sediments can also influence the structure of benthic macroinvertebrate communities (Williams and Mundie 1978). Importantly, many of the organisms that are favoured as food items by stream-dwelling fish species (e.g., mayflies, caddisflies, and stoneflies) prefer relatively coarse streambed substrates and are negatively impacted by intrusions of fine sediment (Everest et al. 1986). Other groups of invertebrates that are utilized less preferentially as fish-food organisms (e.g., midges) are more tolerant of fine sediment intrusions into gravel beds (Nuttall 1972). While differences in preferences for coarse and fine sediments accounts for some of the changes in community composition associated with the deposition of fine sediments, alteration of predator-prey relationships may be an important secondary factor. For example, predacious stoneflies consistently reduced both the density and colonization rates of prey species under control conditions (Peckarsky 1985). The effects of predation on stream insect prey densities and colonization rates, however, were eliminated during periods of sediment transport and silt deposition in the stream systems investigated.

4.3.3 *Fish*

The results of numerous studies demonstrate that elevated levels of fine sediment in streambed substrates have the potential to compromise the survival of salmonid eggs and alevins. The survival of salmonid eggs and alevins is dependent on the delivery of adequate amounts of oxygen and on the removal of toxic metabolic waste products. To meet these basic requirements, streambed substrates must permit the free flow of oxygenated water to incubating embryos (Vaux 1968). Deposition of fine sediment onto and into streambed substrates tends to reduce their permeabilities and, in so doing, decrease the interchange of water between the fluvial and intragravel environments (Wickett 1958; McNeil and Ahnell 1964; Phillips 1971). Low streambed permeability can result in depressed intragravel dissolved oxygen levels which, in turn, compromise the survival of incubating fish embryos (Shumway and Warren 1964; McNeil 1966). In addition, surviving sac fry tend to be smaller, weaker, and have more developmental abnormalities than alevins incubated at high levels of dissolved oxygen (Garside 1959; Silver et al. 1963). Deposited sediments can also block the emergence of fry from the gravel (Koski 1972).

While the deposition of fine sediments is generally considered to be detrimental, there are a number of mitigating factors that can reduce the severity of effects on fish. These mitigating factors can include the shape of the redds (which promotes the flow of water to the eggs), sediment removal during redd building, delivery of oxygenated water via groundwater seepage, and biological compensation for fry mortality (e.g., increase in fry-to-smolt survival due to lower fry densities; McNeil and Ahnell 1964; Stuehrenberg 1975; Klamt 1976; Scrivener and Brownlee 1982; Sowden and Power 1985; Everest et al. 1986). The following discussion provides an overview of the information that is relevant to the development of criteria for streambed substrates.

The effects of deposited sediment on the survival of eggs and alevins have been studied in a number of salmonid species that utilize freshwater habitats in British Columbia (Appendix 3 and 4). Rather than determining deposition rates of fine sediments, these studies usually rely on measures of the overall textural characteristics of streambed substrates to evaluate effects on fish. The variables that are most commonly used to assess the composition of streambed substrates include percent fines (PF), geometric mean diameter (Dg), and Fredle number (FN). Significant relationships between these variables and egg-to-fry survival rates provide a basis for identifying conditions that are hazardous to salmonid fishes. While comparisons of the sensitivities of various fish species were not located in the literature, it is assumed that salmonids represent the most sensitive species to deposited sediments in freshwater ecosystems. This high sensitivity stems from the long exposure periods associated with embryo incubation and utilization of spawning habitats within the streambed matrix.

The term percent fines is often used to describe the portion of a streambed substrate sample that is thought to be harmful to fish. A variety of particle size classes have been used to define the quantity of fine sediment in streambed substrates. From these, 1 mm, 2 mm, 3 mm, 6.35 mm, and 9.52 mm represent the upper limits of the particle size classes that have been used most commonly in studies on the effects of fine sediments on salmonid incubation success.

The results of a number of studies indicate that elevated levels of fine sediment in streambed substrates can be deleterious to fish. For example, introduction of greater than 10% fines <0.75 mm in diameter substantially reduced the survival of brown trout eggs and alevins incubated in artificial stream channels (Olsson and Persson 1986). In addition, a high proportion of the brown trout alevins emerged prematurely, underweight, and underdeveloped. Phillips et al. (1975) reported also delays in emergence timing, reductions in the size of emergent fry, and decreases in the survival of coho salmon embryos incubated in substrates containing more than 10% fines (1.0 to 3.0 mm in diameter). In chinook salmon and steelhead trout, reduced incubation success was observed when streambed substrates contained more than 12% fines <1.7 mm in diameter or 30% fines <6.35 mm in diameter (Tappel and Bjornn 1983). Similar results were obtained for coho salmon, kokanee, cutthroat trout, and rainbow trout embryos incubated under controlled laboratory conditions (Hall and Lantz. 1964; Irving and Bjornn 1984).

The results of two laboratory studies suggest that bull trout and cutthroat are more sensitive than other species to the effects of deposited sediment. In this investigation, substantially reduced egg-to-fry survival rates were observed when percent fines exceeded 4% and 10% in the <2.00 and <6.35 mm size classes, respectively (Weaver and White 1985; Weaver and Fraley 1993). Differences in the egg diameter of these species failed to account for their higher apparent sensitivity. The authors indicated that the use of eyed eggs may have increased mortality due to crushing as the eggs were buried in the gravel matrix. Limited data from a study conducted in artificial spawning channels suggests that Atlantic salmon may be somewhat less sensitive to the effects of deposited sediments than other salmonid species (Marty et al. 1986).

Numerical relationships between egg-to-fry survival rates and the prevalence of two particle size classes of fine sediment (% <2.00 mm and % <6.35 mm) have been reported in the literature. Cederholm and Salo (1979) pooled data from a number of studies and concluded that the emergence success of coho salmon, steelhead trout, cutthroat trout, and brook trout was strongly influenced by the amount of fine sediment, <2.00 mm in diameter, in the incubation medium. The relationship between percent fines (<2.00 mm) and survival to emergence was described by the following equation:

$$\text{Survival (\%)} = 104 - 2.42 \bullet (PF_{<2.00})$$

Similarly, Weaver and White (1985) reported that the survival of bull trout embryos in artificial redds was related to the percent fines less than 6.35 mm in diameter. The following equation described the relationship:

$$\text{Survival (\%)} = 225.2 - 5.13 \bullet (PF_{<6.35})$$

Both of these relationships provide a basis for predicting the survival of salmonid embryos during the incubation period. The available data relating percent fines to salmonid embryo survival are presented in Figures 3, 4, and 5.

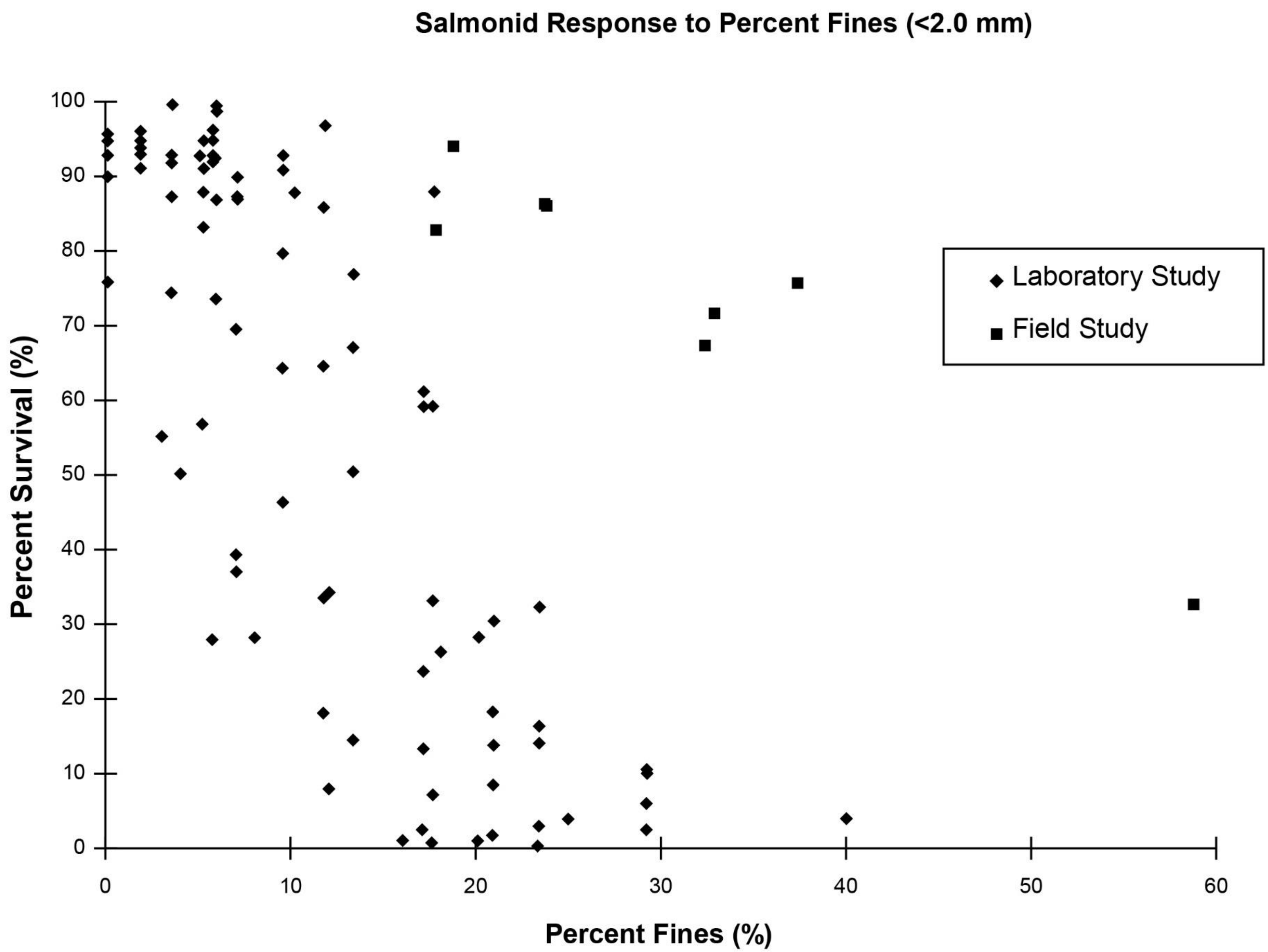

Figure 3. Relationship between embryo survival and percent fines < 2 mm diameter.

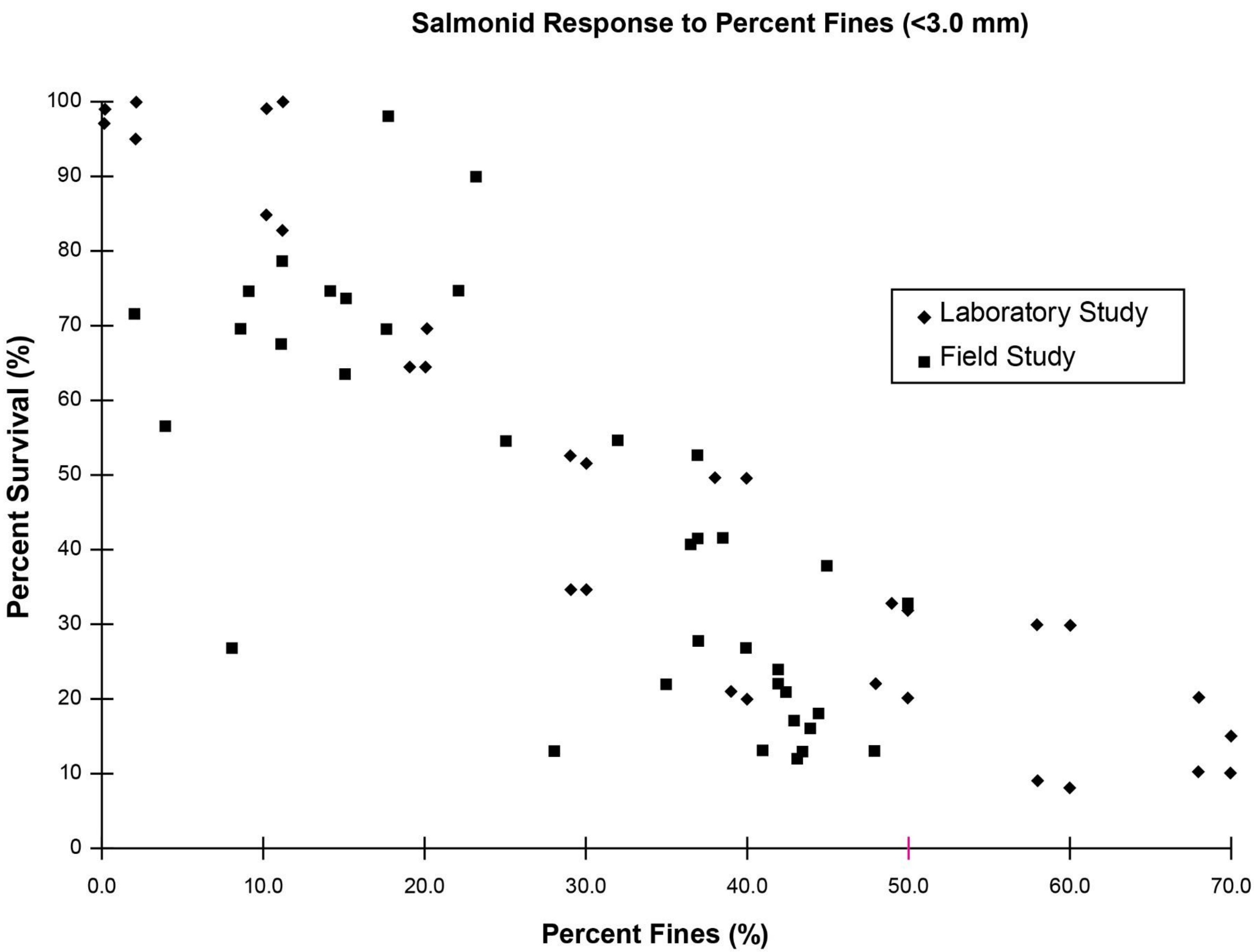

Figure 4. Relationship between embryo survival and percent fines < 3 mm diameter.

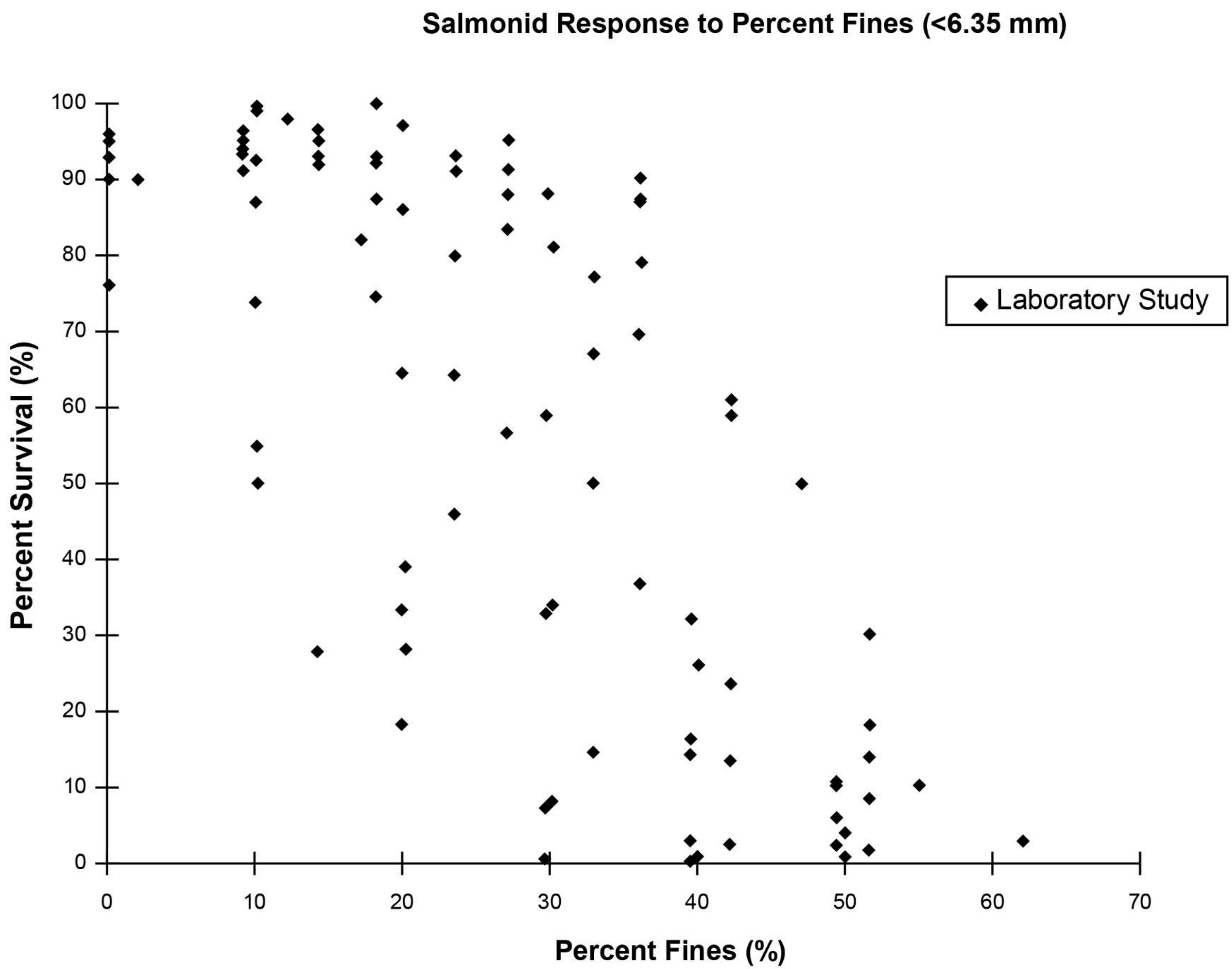

Figure 5. Relationship between embryo survival and percent fines< 6.35 mm diameter.

Natural spawning substrates contain a wide variety of particle sizes, including cobble, gravel, sand, silt, and clay. Therefore, permeability to water flow is not only dependent on the quantity of fine sediments in the substrate, but also on the presence of larger sized particles, such as gravel and cobble (Shirazi and Seim 1979). Platts et al. (1979) proposed geometric mean diameter (Dg) as an alternative measure of streambed substrate composition because it incorporates more information on the overall textural characteristics of the gravel. For any sediment sample, Dg is calculated as follows:

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

In this equation, d_{84} is the 84th percentile particle size and d_{16} is the 16th percentile particle size. Both of these parameters can be estimated from log-probability plots of the particle size distribution.

Information from various sources indicates that egg-to-fry survival rates are compromised when the overall particle size distribution of streambed substrates is reduced (Figure 6). For example, Olsson and Persson (1986) reported that the survival of brown trout embryos decreased substantially when Dg fell below 9.6 mm. Similarly, egg-to-fry survival rates for coho salmon were reduced at a Dg of roughly 15 mm or less (Koski 1966; Phillips et al. 1975; Tagart 1976; Cederholm 1997). While similar results were obtained for steelhead trout in another study (Cederholm and Lestelle 1974), Tappel and Bjorn (1983) reported that the survival of steelhead trout and chinook salmon embryos was not adversely affected until the Dg fell below 10 mm. The differences between these studies likely reflect the use of eyed eggs in the Tappel and Bjorn (1983) investigation, which reduced the period of exposure to the adverse environmental conditions. As such, the chinook salmon and steelhead trout tested in this study appeared to be less sensitive to deposited sediments. The highest levels of fine sediment tested in this study also caused premature emergence and reduced size of chinook and steelhead fry.

Significant variability is evident in the relationship between Dg and embryo survival rates. A stronger correlation between substrate composition and embryo survival was obtained when Dg was divided by the mean egg diameter (De) for the species tested (i.e., to account for differences in egg sizes between different fish species; Shirazi and Seim 1979). Among the species tested, substantially reduced embryo survival was generally observed when Dg/De fell below 3.0 mm.

The Fredle number provides a better overall indication of the composition of streambed substrates than do percent fines or geometric mean diameter because it integrates more information on the overall particle size distribution. The Fredle number, a unitless value, is calculated by dividing the geometric mean diameter of a sediment sample by the sorting coefficient [So where $So = (d_{75} \div d_{25})^{0.5}$; Lotspeich and Everest 1981] and can be used as a measure of the pore size of the streambed substrate. The relationship between Fredle number and the survival to emergence of chinook salmon and steelhead trout fry is presented in Figure 7. These data show that the survival rate of salmonid embryos during incubation drops rapidly when the Fredle number falls below five.

After emergence, deposited sediment can affect the rearing habitats utilized by juvenile salmonids by increasing the embeddedness of streambed substrates. For species that are closely associated with the streambed, sediment deposition decreases the available rearing habitat. The production of fish-food organisms can also be reduced in areas with high embeddedness. These factors can combine to reduce the carrying capacity of the stream and, thereby, lead to decreased recruitment rates when fry densities are higher than can be accommodated by the available rearing habitat (Pratt 1985).

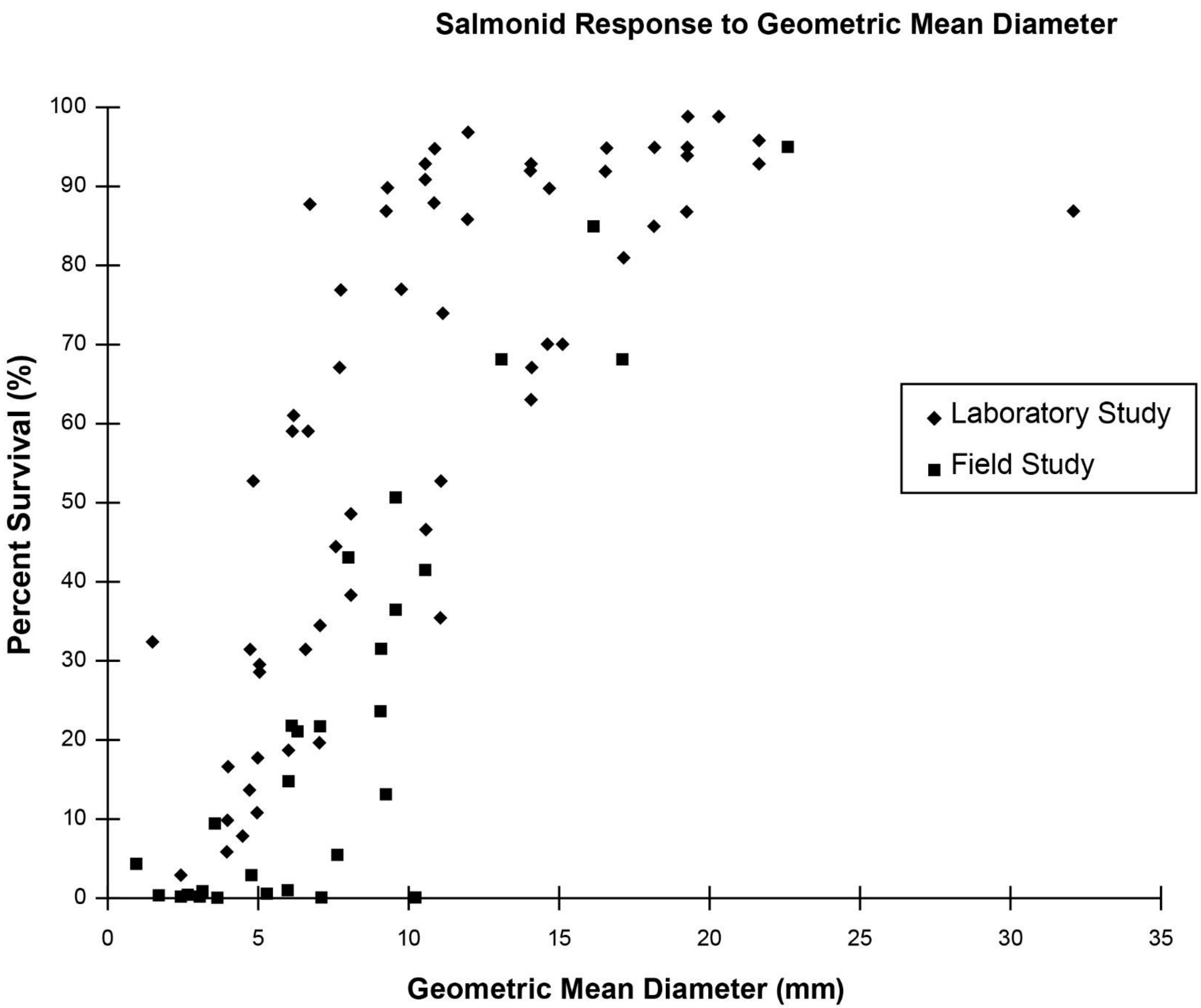

Figure 6. Relationship between embryo survival and the geometric mean diameter in streambed substrate.

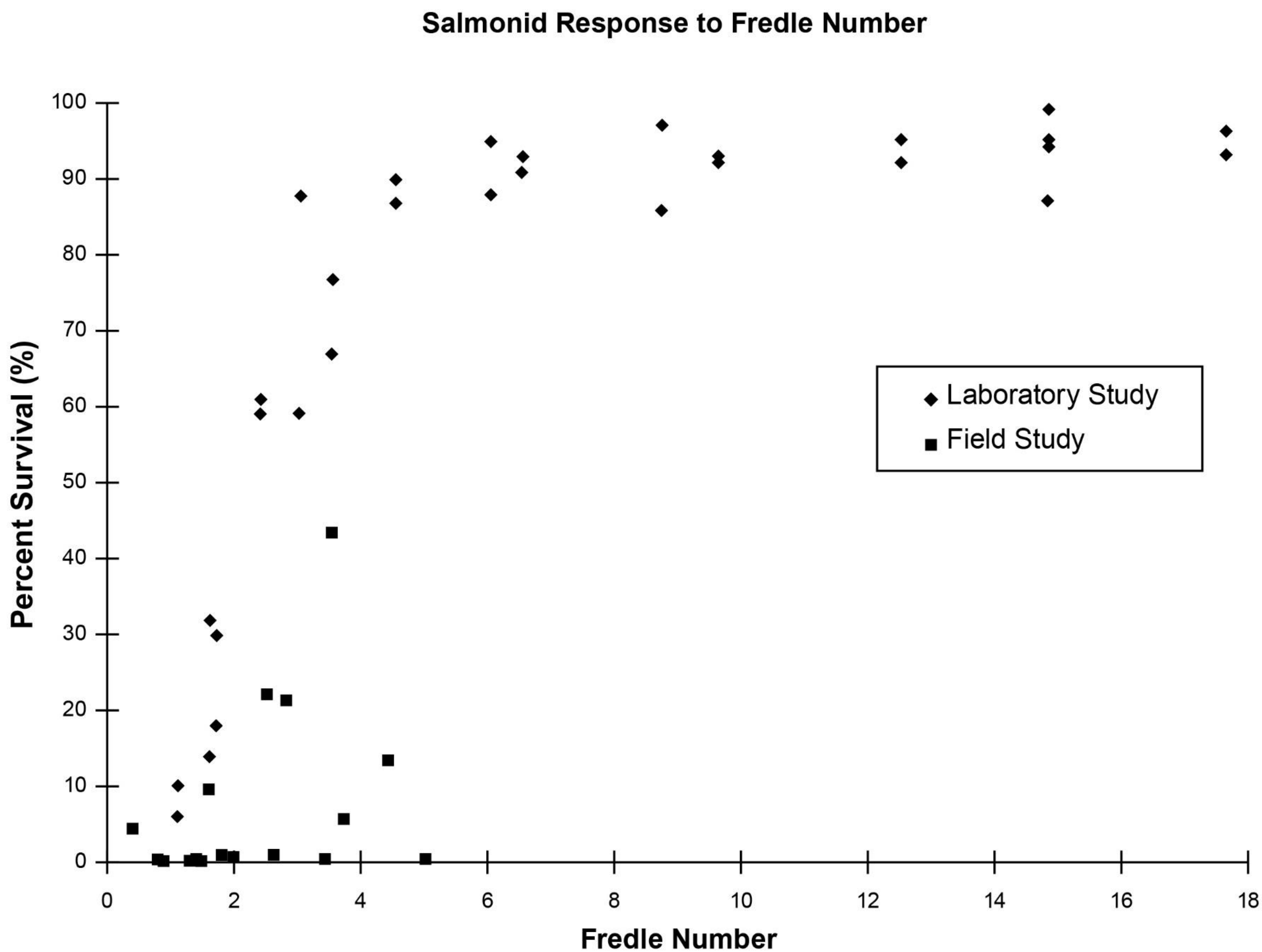

Figure 7. Relationship between embryo survival and Fredle number of streambed substrate.

4.4 Literature criteria

To our knowledge, no world jurisdictions have formulated more recent criteria using new information, approaches or methodologies than those already discussed by Singleton (1985) and Singleton et al. (1995). Instead, existing criteria levels from other jurisdictions have been adopted (e.g., CCME 1987). To assist in the application of the newly proposed guideline and for comparative purposes, the summary tables by Singleton (1985) are given below (Tables 4 to 7). For a discussion on the approaches that were used by the different jurisdictions, the reader is referred to the description given in Singleton (1985).

Table 4. Turbidity criteria for freshwater aquatic life.

Criteria Statement	Criteria Values	Reference
The combined effect of colour and turbidity should not change the compensation point more than 10% from its seasonally established norm, nor should such a change place more than 10% of the biomass of photosynthetic organisms below the compensation point		U.S. EPA 1972
Not to exceed more than 25 JTU over natural turbidity	25 JTU	Environment Saskatchewan 1975; Alberta Department of the Environment 1977
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		U.S. EPA 1976
Recommends a "two parameter'' approach using non-filterable residue (mg/L) and turbidity (NTU)		Garton et al. 1979 (no jurisdiction given)
Suspended matter should not be added to surface water in concentrations that will change the natural Secchi disk reading by more than 10%		Ontario Ministry of the Environment 1979
Shall not exceed 25 NTU above natural condition level. For all lake waters, shall not exceed 5 NTU over natural conditions	5-25 NTU increase	Alaska Department of Environmental Conservation 1979
Criteria range from no increase above naturally occurring turbidity to a maximum allowable increase of 10 NTU above background, or that which may be harmful or create a nuisance, depending on the fish species present	0-10 NTU increase	Montana Health and Environmental Sciences 1980
Wastewater from point source discharges must not increase turbidity outside the mixing zone by: -more than 5 NTU over background when background turbidity is ≤50 NTU -more than 10% over background when background turbidity is >50 NTU to a maximum increase of 25 NTU	5-25 NTU increase	Idaho Department of Health and Welfare 1980

Criteria Statement	Criteria Values	Reference
Criteria range from ≤5 NTU over background to ≤ 10 NTU over background when the background turbidity is ≤50 NTU, or not more than a 10-20% increase when background >50 NTU, depending on classification	5-10 NTU increase	State of Washington 1982
Suggests several alternatives to the 1979 Alaska Water Quality Standards		Alaska Department of Environmental Conservation 1982
Turbidity not to exceed 5 JTU over background	5 JTU increase	B.C. Pollution Control Board 1974-1980
Free from materials that produce turbidity in such a degree as to be objectionable or to impair any beneficial use		Manitoba Department of Environment 1983

Table 5. Non-filterable residue criteria for freshwater aquatic life.

Criteria Statement	Criteria Values	Reference
"No evidence of harmful effects on fisheries at less than 25 mg/L of suspended solids..."	25 mg/L	EIFAC 1965 Alabaster and Lloyd 1982
Good or moderate fisheries may exist in waters normally containing 25 to 80 mg/L	25-80 mg/L	EIFAC 1965 Alabaster and Lloyd 1982
Waters with 80 to 400 mg/L suspended solids sometimes support fisheries, but they are unlikely to be good, even in the lower part of that range	80-400 mg/L	EIFAC 1965 Alabaster and Lloyd 1982
Only poor fisheries are likely to be found in waters which normally contain > 400 mg/L suspended solids	>400 mg/L	EIFAC 1965 Alabaster and Lloyd 1982
The spawning grounds of salmonids should be kept as free as possible from finely divided solids		EIFAC 1965 Alabaster and Lloyd 1982
Guideline levels of protection to aquatic communities as follows: high 25 mg/L moderate 80 mg/L low 400 mg/L very low >400 mg/L		U.S. EPA 1972
Formula based on the natural seasonal maximum for effects of finely divided solids on aquatic organisms		Environment Canada 1972
Not to be increased by more than 10 mg/L over background value	10 mg/L increase	Environment Saskatchewan 1975; Alberta Department of the Environment 1977
Settleable and suspended solids should not reduce the depth of the compensation point for		U.S. EPA 1976

Criteria Statement	Criteria Values	Reference
photosynthesis activity by more than 10% from the seasonally established norm for aquatic life		
Recommend a “two parameter” approach using non-filterable residue (mg/L) and turbidity (NTU)		Garton et al. 1979 (no jurisdiction given)
No sediment loads (suspended or deposited) that can cause adverse effects on aquatic animal or plant life, their reproduction or habitat		Alaska Department of Environmental Conservation 1982
No anthropogenic-caused suspended matter of any kind in concentrations causing nuisance or objectionable conditions or that may adversely affect designated or protected beneficial uses		Idaho Department of Health and Welfare 1980
For protection of aquatic life, waters should be free from substances attributable to municipal, industrial, or other discharges resulting from human activity that will settle to form putrescent or otherwise objectionable sludge deposits or that will alter the value of Secchi disk depth by more than 10%		Walker 1980 IJC
For salmonid dominated waters, 95^{th} and 99^{th} percentiles for suspended solids are 80 and 150 mg/L, respectively	80 mg/L 150 mg/L	Anglian Water Authority 1982
For cyprinid dominated waters, 95th and 99th percentiles for suspended solids are 100 and 200 mg/L, respectively	100 mg/L 200 mg/L	Anglian Water Authority 1982
Suggests several alternatives to the 1979 Alaska Water Quality Standards		Alaska Department of Environmental Conservation 1982
Maximum acceptable concentration for protection of aquatic life = 25 mg/L	25 mg/L	Manitoba Department of Environment 1983
Suggested criteria for salmonid hatcheries are 3 mg/L for incubation and 25 mg/L for rearing and holding	3-25 mg/L	D.F.O. 1983

Table 6. Turbidity criteria for marine aquatic life.

Criteria Statement	Criteria Values	Reference
Shellfish growth and propagation, none of mineral origin in excess of 25 units	25 units	B.C. Health 1969
Where natural turbidity is between 0-50 JTU any increase shall not exceed 20%. Where natural turbidity is between 50-100 JTU any increase shall not exceed 10 JTU. Where natural turbidity is >100 JTU any increase shall not exceed 10%	20% or 10 JTU increase	State of California Marine Water Quality Standards, Central Coastal Basin 1978

Criteria Statement	Criteria Values	Reference
Shall not reduce the depth of the compensation point for photosynthetic activity by more than 10%. In addition, shall not reduce the maximum Secchi disc depth by more than 10%		Alaska Department of Environmental Conservation 1979
Suggests several alternatives to the 1979 Alaska Water Quality Standards		Alaska Department of Environmental Conservation 1982
Criteria for the protection of aquatic life ranges from 5 NTU over background to ≤10 NTU over background when the background turbidity is ≤50 NTU, or not more than a 10-20% increase when background >50 NTU, depending upon classification	5-10 NTU increase	State of Washington 1982

Table 7. Non-filterable residue criteria for marine aquatic life.

Criteria Statement	Criteria Values	Reference
No measurable increase in concentrations above natural conditions	0 mg/L	Alaska Department of Environmental Conservation 1979
Suggests several alternatives to 1979 Alaska Water Quality Standards		Alaska Department of Environmental Conservation 1982

4.5 Recommended guidelines

The British Columbia guideline to protect aquatic life in fresh, estuarine, and coastal marine waters from excessive suspended sediments originating from anthropogenic sources are established according to the amount of non-filterable residue and the turbidity of the aquatic system. Guidelines for substrate composition and for bedload transport have also been developed, which are specific to salmonid spawning and mariculture areas. As the biotic, physical, and chemical conditions describing aquatic ecosystems are diverse, the recommended guidelines will need to be compared to natural background levels.

4.5.1 *Suspended sediments*

Distinct water quality guidelines for suspended sediments are recommended for the protection of aquatic life during clear flow and turbid flow periods. During clear flow periods (see definition Section 2.1.1), anthropogenic activities should not increase suspended sediment concentrations (or non-filterable residue levels) by more than 25 mg/L over background levels during any 24 hour period. For sediment inputs that last between 24 hours and 30 days, suspended sediment concentrations should not be increased by more than 5 mg/L over background levels. During the course of sampling (n = 24, 24 h every hour: n = 30, 30 d every day), if any sample is above the criteria, it is considered an exceedance. To statistically test whether exceedances have occurred, comparisons between control (e.g., upstream) and impacted (e.g., downstream) sites are required (see Technical Appendix Addendum). This can only be done if all samples (i.e., 24 samples in 24 hours and 30 samples in 30 days) are considered in the analysis.

During high flow periods, anthropogenic activities should not increase suspended sediment concentrations by more than 10 mg/L at any time when background levels are between 25 and 100 mg/L.

When background levels exceed 100 mg/L, suspended sediment concentrations should not be increased by more than 10% of the measured background level at any one time.

4.5.2 *Rationale*

This two-pronged approach to guideline setting for suspended sediments recognizes that exposure duration plays a key role in the toxicity response. The 25 mg/L criterion is based on the severity-of-ill-effects (SEV) concentration-duration response curve approach (described below, Section 4.6). The criterion is based on the change in suspended sediment concentration causing an increase of one in a severity of-ill-effects score for the most sensitive taxonomic group of aquatic organisms. For B.C. waters, adult salmonids are the most sensitive, i.e., the concentration-response SEV slope at a given time is the steepest (24-48 h; slope 2.08, Table 11). The guideline level is slightly above the former B.C. guideline level and those of other jurisdictions such as Saskatchewan, Alberta and the CCME which have all been set arbitrarily at 10 mg/L. It is consistent with levels from Alaska, Manitoba, D.F.O and the European Inland Advisory Commission (see Table 6).

In clear stream systems, small, induced exceedances in suspended sediment concentration above a 25 mg/L (for 24 h) change from background levels are likely to cause behavioural and low SEV sublethal effects on fish, all of which are reversible. Conditions in these systems should be rectified to prevent possible further damage of the designated water use. Very low suspended sediments levels are known to cause egg mortality (40%) to rainbow trout at long durations (7 mg/L at 48 d; 0.5-75 µm). Based on extrapolation from the analysis described below (Section 4.6), a long-term guideline has been set at a non-filterable residue change in 5 mg/L over a 30-day period. According to the SEV scale this concentration-duration exposure translates to a SEV score of five or minor physiological stress, and increased rates of coughing and respiration (Table 9).

The guideline has been based on a large database (Newcombe 1994a; Newcombe and Jensen 1996a, b) that reports effects to biota on a province-wide scale. Site-specific objectives may need to be developed to account for the particular set of biotic and abiotic conditions of a region. Site-specific objectives consider scientific, economical and social values that may, for example, lower the guideline in very sensitive habitats with high resource use or, raise the guideline for the setting of industrial abatement targets.

4.5.3 *Turbidity*

Induced turbidity should not exceed a change of 8 NTU for a duration of 24 h above the background concentration in all waters during clear flows (see definition Section 2.1.1). As well, a long-term guideline has been set stating that turbidity should not exceed a change of 2 NTU for a 30-d period during clear flows. <u>During the course of sampling (n = 24, 24 h every hour; n = 30. 30 d every day), if any sample is above the guideline, it is considered an exceedance. To statistically test whether exceedances have occurred, comparisons between control (e.g., upstream) and impacted (e.g., downstream) sites are required (see Technical Appendix Addendum).</u> This can only be done if all samples (i.e., 24 samples in 24 hours and 30 samples in 30 days) are considered in the analysis.

During high flows and in turbid waters, the former guideline is adopted i.e., turbidity should not exceed 5 NTU at any time when background turbidity is between 8 and 50 NTU, nor should there be an increase of more than 10% of background when background is >50 NTU at any time.

4.5.4 *Rationale*

These guidelines are based on the effects-based non-filterable residue criteria of a 25 mg/L and 5 mg/L change from background for a duration of 24 h and 30 d, respectively, according to the non-filterable residue and the general turbidity correlation of 3 to 1 described in Section 1.2. A turbidity of 8.33 NTU has been rounded off to 8 NTU and 1.67 to 2 NTU for practical reasons. This is further supported in Figure 1 by extrapolating from a log suspended sediment concentration.

The turbidity guideline of a 8 NTU change from background turbidity for a duration of 24 h is consistent with criteria from other jurisdictions (Washington, Montana), however, it specifies a duration for the event and is effects-based. This guideline is a recommended check in every routine field sampling program as it can be taken with accuracy and rapidity with field nephelometers. If problem areas are found, joint turbidity and filterable residue measurements are recommended (see complement Sampling Protocol). The long-term turbidity guideline of 2 NTU, as for the long-term guideline for non-filterable residues, will protect against low anthropogenic suspended sediment inputs that persist over the long term.

The guideline will protect against harm to all aquatic life in freshwaters, marine and estuarine waters. There may be a need to develop site-specific objectives to account for the particular set of biotic and abiotic conditions of a region. Site-specific objectives consider scientific, economical and social values that may, for example, lower the guideline in very sensitive habitats with high resource use or, raise the guideline for the setting of industrial abatement targets.

4.5.5 *Bedload sediments*

Insufficient information is currently available to develop numerical water quality guidelines for bedload sediments.

4.5.6 *Rationale*

Bedload transport of fine sediment in stream systems plays an important role in maintaining the habitats utilized by anadromous and resident fish species. During low and moderate flow periods, fine sediment can accumulate in streambed substrates through the process of sedimentation. Such accumulations of fine sediment can render the streambed substrate unsuitable for salmonid incubation, alter the abundance and species composition of benthic macroinvertebrate communities, and influence primary productivity (Tappel 1981; Culp et al. 1985). The hydraulic energy of stream systems, however, increases during storm events which facilitates transport of the fine sediment that is stored in the streambed (Diplas 1987). This cyclical deposition and resuspension of fine sediment is a natural process that maintains the stream channel morphology (i.e., pool-riffle sequences) that is essential for many coldwater fish species (Sidle 1988).

Insufficient data were located on the biological effects of bedload sediment in fresh water ecosystems to support the derivation of numerical water quality guideline. Nonetheless, it is apparent that increased bedload transport of inorganic sediments could adversely affect streambed and stream channel morphology, with associated effects on habitat structure and availability (Parker and Andrews 1985). Reduced bedload transport could result in accumulations of fine sediment in streambed substrates and, potentially, degradation of spawning and rearing habitats (Sidle 1988).

4.5.7 *Streambed substrate and deposited sediments*

The quantity of fine sediment in streambed substrates (i.e., percent fines) should not exceed 10% <2.00 mm, 19% <3.00 mm, and 25% <6.35 mm at potential salmonid spawning sites. The geometric mean diameter and Fredle number of streambed substrates should not be less than 12.0 mm and 5.0, respectively. The minimum and 30-day average criteria for intragravel dissolved oxygen levels are 6.0 and 8.0 mg/L, respectively. These guidelines apply to actual and potential spawning sites in streams throughout the province.

4.5.8 *Rationale*

Two distinct approaches were employed to establish water quality guidelines for deposited sediments. These guidelines considered both streambed substrate composition and intragravel water quality conditions. The criteria for both groups of variables were selected to minimize the potential for adverse effects on salmonid egg-to-fry survival rates associated with fine sediment deposition in streambed substrates. This approach was considered to be appropriate because the available data suggest that salmonid embryos are likely the most sensitive life stage in freshwater ecosystems.

In stream systems, incubating fish embryos are subjected to a number of hazards that can influence their survival. Losses of incubating eggs and alevins can occur due to redd superimposition, redd dewatering, extreme streamflows, extreme temperatures, disease organisms, and a host of other factors. To minimize the potential for adversely affecting salmonid populations, the criteria for deposited sediments should be established at levels that will support high egg-to-fry survival rates (i.e., ≥80%).

The available data linking streambed substrate composition to survival of salmonid embryos provide a scientific basis for establishing numerical guidelines for the protection of aquatic life. For example, information from several studies indicates that egg-to-fry survival rates can be reduced by 40% or more when significant quantities of fine sediment (i.e., <2.00 mm and 6.35 mm in diameter) are incorporated into streambed substrates. Similarly, strong linkages between egg-to-fry survival rates and both geometric mean diameter and Fredle number are evident in the literature. For this reason, it was considered appropriate to develop guidelines for several size classes of fine sediment and for two additional variables that more completely describe the textural characteristics of streambed substrates (i.e., Dg and Fredle number).

Both laboratory and field-collected data were compiled from the scientific literature. The results of field studies, however, are more difficult to interpret because the survival of salmonid embryos in natural systems can be influenced by numerous factors other than streambed substrate composition (e.g., redd dewatering, extreme streamflows, etc.). For this reason, the results of the laboratory studies were considered to provide the most direct basis for identifying the levels of fine sediments that would adversely affect embryo survival and for deriving water quality criteria for deposited sediments.

In each of the laboratory studies, the egg-to-fry survival of salmonid embryos was determined for a range of streambed substrate compositions. In some cases, negative control tests were also performed to verify the health and vitality of the egg stock that was used in the test (i.e., by incubating the eggs using Heath trays or comparable fish culture methods). While protocols have been established for conducting toxicity tests using the early life stages of salmon fish (Environment Canada 1992), standard procedures have not been developed for bioassays that are conducted in gravel media. As a result, differences in test procedures and test durations could be substantial, potentially accounting for much of the variability that is evident in the toxicological database. In an effort to increase the comparability of the toxicological information, data sets were included in the data analyses only if the conditions in the lowest treatment group were indicative of conditions in unimpacted watersheds (MacDonald and MacDonald 1987) and egg-to-fry survival in that treatment group was ≥80%.

Following data screening, the data were normalized to account for differences in control survival rates between studies (i.e., by dividing the survival in the treatment group by the survival in the control group and multiplying by 100). Subsequently, these data were transformed [i.e., using a logistic transformation of the response data and logarithmic (ln) transformation of the concentration data] to linearize the relationship between substrate statistics and embryo survival. This analytical approach was utilized because the untransformed data appeared to resemble typical dose-response curves and this type of data transformation would facilitate analysis by linear regression.

Evaluation of the toxicological data set using these procedures yielded the following equations, which describe the relationships between embryo survival (S) and three key streambed substrate statistics (% <2.00 mm, % <3.00 mm, % <6.35 mm):

$$\ln (S/1-S) = -2.66 \ln (PF_{2.00\ mm}) + 7.43\ (r^2 = 0.57)$$

$$\ln (S/1-S) = -2.35 \ln (PF_{3.00\ mm}) + 8.43\ (r^2 = 0.77)$$

$$\ln (S/1-S) = -2.30 \ln (PF_{6.35\ mm}) + 8.85\ (r^2 = 0.47)$$

The results of these analyses indicated that egg-to-fry survival rates of 80% would generally be achieved (relative to control survival) if percent fines <2.00 mm, <3.00 mm, and 6.35 mm in diameter remained below 10%, 19%, and 25%, respectively. These values were adopted directly as the water quality criteria for percent fines in streambed substrates.

For geometric mean diameter (Dg) and Fredle number (FN), the raw data were better correlated to embryo survival than the normalized data. For this reason, the raw data were transformed and utilized to develop the following linear equations, which relate streambed substrate characteristics to egg-to-fry survival rates (S):

$$\ln (S/1-S) = 2.48 \ln (Dg) - 4.60\ (r^2 = 0.66)$$

$$\ln (S/1-S) = 1.87 \ln (FN) - 1.66\ (r^2 = 0.81)$$

Using these equations, it is possible to determine that adequate egg-to-fry survival rates (i.e., ≥80%) will likely be maintained When Dg and FN remain above roughly 12 mm and 5.0, respectively. These values were adopted directly as the water quality guidelines for these streambed substrate statistics.

While the field-collected data were not used to derive the water quality guidelines, they do provide useful information for evaluating the predictability of these guidelines. In this assessment, streambed substrate samples were considered to be toxic if egg-to-fry survival rates were lower than the mean control survival that was observed in the acceptable laboratory studies by ≥20% (i.e., if survival in the control group was 93%, than threshold toxic responses were identified when survival in the treatment group was ≤93%), as follows:

Statistic	Percent of Samples Toxic When Criterion is Exceeded
$PF_{2.00\ mm}$	18 of 23 (78%)
$PF_{3.00\ mm}$	23 of 26 (88%)
$PF_{6.35\ mm}$	No data
Dg	27 of 27 (100%)
FN	19 of 19 (100%)

Therefore, the data from field studies show that reduced survival of salmonid embryos can be expected when the characteristics of streambed substrates lie outside the ranges specified by the water quality guidelines.

Information on streambed substrate composition provides important indicators of the physical conditions that salmonid embryos may be exposed to during incubation. The egg-to-fry survival rates also depend, however, on the maintenance of adequate chemical conditions throughout the incubation period. Data from a number of studies show that low streambed permeability can result in depressed intragravel dissolved oxygen levels and, hence, reduced egg-to-fry survival rates during incubation (Wickett 1958; Shumway and Warren 1964; McNeil 1966; Phillips 1971). Davis and Lathrop-Davis (1986) reviewed the literature on the sensitivity, responses, response thresholds, and minimum oxygen requirements of freshwater fish species. The results of this study indicate that embryos incubated at low oxygen tensions had lower growth rates, retarded development, higher incidence of deformities, and reduced survival rates than those incubated at higher dissolved oxygen concentrations. The results of a more recent study showed that critical dissolved oxygen levels for rainbow trout eggs and alevins varied with water temperature, stage of development, and egg density (Rombough 1986). Under a range of experimental conditions, adverse effects on survival and development were generally avoided if dissolved oxygen levels averaged above 8.0 mg/L and did not fall below 6.0 mg/L, particularly during hatching. These critical values are the recommended B.C. water quality guidelines for intragravel dissolved oxygen concentrations (Ministry of the Environment Lands and Parks 1997).

4.6 Severity-of-Ill-Effects Approach

The rationale behind using the severity-of-ill-effects approach (SEV) for guideline development is that it is the first reported representation of effects of suspended sediments on biota. It has incorporated the duration of the exposure in the description of the response variable which is essential in understanding toxicity responses. The approach originally developed by Newcombe (1994b) and Newcombe and Jensen (1996b), is further described below. For the purpose of guideline development several new statistical procedures have been conducted and are also outlined below.

The severity-of-ill-effects approach (SEV) is based on the concept that effects of suspended sediment pollution on fish increases as a function of sediment concentration and duration of exposure (Newcombe and MacDonald 1991). The approach was originally based on data triplets consisting of suspended sediment concentration, duration of exposure, and severity-of-ill effect embracing a wide taxonomic range from phytoplankton to fish. The models were refined for particular fish taxa, natural history, life history stage, and predominant sizes of the sediment particles responsible for ill effects (Newcombe 1994b). These have later been expanded to further include invertebrates and flora (Newcombe 1996).

The SEV concentration-duration response curve relationship is described as a three-dimensional (3-D) toxicity surface. Estimation of toxicity by fitting concentration time response surfaces is an old concept that has re-emerged only recently with the advent of powerful computers able to manipulate large databases (Sun et al. 1995).

Using regression analysis one is able to determine the effect of an increase in concentration in suspended sediments for a known duration on a particular taxonomic group of organisms. The effect is based on severity-of-ill-effects scores that encompass no effect, behavioral, sublethal, lethal and paralethal effects. The SEV scale is a 15-point scale (0-14). The scales for fish, aquatic invertebrates and flora are shown in Table 8.

Table 8. Severity-of-ill-effects (SEV) score description of effects (adapted from Newcombe and Jensen 1996b).

SEV	Fish	Invertebrates	Flora
0	No behavioral effects	No harmful effects	No harmful effects
1	Alarm reaction	No data. Intermediate value	No data. Intermediate value
2	Abandonment of cover	No data. Intermediate value	No data. Intermediate value
3	Avoidance response	Increased drift	No data. Intermediate value
4	Short-term reduction in feeding rates or feeding success	Short-term (<1 h) reduction in feeding rates, (~10%) including ingestion and incorporation	No data. Intermediate value
5	Minor physiological stress; increase in rate of coughing: increased respiration rate	Short-term (<1 h) reduction in feeding. rates, (~90%) including ingestion and incorporation	No data. Intermediate value
6	Moderate physiological stress	No data. Intermediate value	No data. Intermediate value
7	Moderate habitat degradation; impaired homing	No data. Intermediate value	No data. Intermediate value
8	Indications of major physiological stress; long-term reduction in feeding rate or success; poor condition	Silt intolerant species less abundant; long-term (>24 h) reduction in feeding rate or success; temporary changes in community structure; potential starvation of invertebrates	No data. Intermediate value
9	Reduced growth rate; delayed hatching; reduced fish density	No data. Intermediate value	No data. Intermediate value
10	0-20% mortality; increased predation; moderate to severe habitat degradation	Number of taxa reduced; standing crop reduced by 0-20%; survival and fecundity reduced; species diversity reduced; taxonomic diversity reduced; gills or gut, or both, clogged with particles; 0-20% mortality; increased predation	Number of taxa reduced; standing crop reduced by 0-20%; survival or fecundity reduced; species diversity reduced; taxonomic diversity reduced; 0-20% mortality;

11	>20-40% mortality	>20-40% mortality; abundance of invertebrates reduced by similar percentage (>20-40%)	>20-40% mortality; abundance or distribution reduced by similar percentage (>20-40%)
12	>40-60% mortality	>40-60% mortality, or reduced abundance	>40-60% mortality, or reduced abundance or reduced distribution (>40-60%)
13	>60-80% mortality	>60-80% mortality, or reduced abundance	>60-80% mortality, or reduced abundance or reduced distribution (>60-80%) leaves severely damaged by abrasion
14	>80-100% mortality	>80-100% mortality, or reduced abundance	>80-100% mortality, or reduced abundance or reduced distribution (>80-100%)

0 = nil effect
1-3 = behavioural effects
4-8 = sublethal effects
9-14 = lethal and paralethal effects

A comprehensive account of the toxicological interpretation of the SEV scale is given in Newcombe and Jensen (1996b). As well as direct effects on fish, the scale incorporates habitat damages such as reduced porosity of spawning gravel and impaired homing. Scientific expertise and judgement have gone into delineating between the severity of effect (e.g., from moderate to severe), as the science cannot be precise. For example, a pollution event resulting in suspended sediment being deposited in or on spawning habitat during egg incubation could be characterized as "moderately severe" if the area is a small portion of the total area available for fish spawning whereas "severe" habitat changes may be characterized substantial reduction in fish populations. Habitat damage is interpreted as an accumulative measure of ill effects at various life stages of fish that can only be studied in the field (Newcombe and Jensen 1996b).

The database (Appendix 1 and 2) has been divided into eight data groups. Six of these are fish groups, one is an invertebrate group and the other an invertebrate and flora group (fauna). The fish groups have been chosen according to four distinctive selection criteria.

Box 1: Selection criteria for six fish data groups (modified from Newcombe and Jensen 1996a, b)

Taxonomy: Salmonids were distinguished from non-salmonids, although some groupings were not exclusively one or the other.

Life stage: Life stages were allocated among four categories including: 1) eggs; 2) larvae (recently hatched fish including yolk-sac fry, that had not passed through final metamorphosis); 3) juveniles (including fry, parr, and smelts, that had passed through larval metamorphosis but were sexually immature); and 4) adults.

Life history: Estuarine species were categorized separately from anadromous and freshwater species, although these two groups were combined for early life stages.

Sediment particle size: Two categories of sizes were taken from the literature where particle size ranged up to 250 µm. Fine particles <75 µm are small enough to pass through gill membranes into the interlamellar spaces of gill tissue and include clay, silt, and very fine sand particles. Coarse particles >75 µm are large enough to cause mechanical abrasion of gills and include very fine to fine sand particles.

These are: 1) taxonomy; 2) life stage; 3) life history; and 4) sediment particle size (Box 1). The eight data groups for which concentration-duration response models were fitted are given below with their respective sample size and particle size measure. Appendices 1 and 2 list all the individual referenced studies in each group, reporting species, life stage, exposure concentration and duration and a description of the severity-of-ill-effect.

Group 1: Juvenile and adult salmonids, 0.5-250 µm, N = 171

Group 2: Adult salmonids, 0.5-250 µm, N = 63

Group 3: Juvenile salmonids, 0.5-75 µm, N = 108

Group 4: Eggs and larvae of salmonids and non-salmonids, 0.5-75 µm, N = 43

Group 5: Adult estuarine non-salmonids, 0.5-75 µm, N = 28

Group 6: Adult freshwater non-salmonids, 0.5-75 µm, N = 22

Group 7: Aquatic invertebrates, 0.5-250 µm, N = 69

Group 8: Aquatic invertebrates and aquatic flora, 0.5-250 µm, N = 61

These different data groups are essential to the development of site-specific objectives for B.C. waters. For the purpose of deriving the aquatic life guidelines for the province, Group 2 was found to be the most sensitive to changes in suspended sediment levels over a period of time. Sensitivity to increased concentrations and durations of suspended sediments was based on the steepness of the slope from the concentration-duration response curves. The steeper the slope, the smaller the change in suspended sediment concentration invoking an effect. Apparent effects are defined as an increase in the SEV score of one unit in the 15-point scale.

The first step in the analysis is to obtain the best-fit model to the data in each group. The analyses were made in 3-D using log concentration, log duration and response as input variables. The regression analyses showed that a sigmoid shaped model (Weibull) could be fitted to all data sets with good fits according to a *G* goodness-of-fit test. The findings are in concordance with the models fitted by Newcombe and Jensen (1996b), but with an improvement in fit. The statistics used to judge curve fit and to calculate prediction confidence intervals are also different, but better suited for criteria development.

A summary table of the analyses (Table 9) indicates that two models (i.e., log-linear; log-Weibull model) provided adequate goodness-of-fit for all eight data sets ($p > 0.05$, i.e., the model predictions are not significantly different from the observed responses). The Weibull model, however, produced a better fit as measured by the *G*-test in each case. This is to be expected as the Weibull function can adapt to toxicity related data which are typically sigmoid shaped dose-response curves. The following paragraphs describe the analyses in more detail.

The two parameter Weibull model equation is as follows:

$$Y = (1 - e^{-\mathrm{K}x^{\tau}})\,100$$

where x is the exposure concentration, and K and τ are the fitted parameters. For each data set analyzed, a *G*-test was conducted to determine goodness of fit for each of the eight data sets for the log-Weibull and log-linear models. These statistics were used to test the null hypothesis that the fit of the model was adequate (i.e., if $p > 0.05$, the model estimates were not significantly different from the observed responses). The G-test is recommended over the more traditional chi-square test for goodness of fit because the two give very similar results, but the *G*-test is computationally simpler, especially with more complicated designs (Sokal and Rohlf 1981). To compute G, we used the equation:

$$G = 2 \sum_{i=1}^{a} \mathrm{f}_i \ln\left(\frac{f_i}{\hat{f}_i}\right)$$

where a is the number of samples, f_i is the observed response for treatment *i*, and $\hat{f}_i$ is the corresponding model estimate. The value of *G* is then compared with the critical value of x^2 for $a - p - 1$ degrees of freedom at $\propto = 0.05$. *p* is the number of parameters in the model equation.

Table 9. Summary statistics for model curve fitting.

LOG-linear Model								
Groups	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Gstat	89	30	53	15	8.4	3.9	16	14
Probability%	100	99.9	100	100	100	100	100	100
R-squared	60	64	61	58	66	74	82	84
Intercept a	1.055	1.68	0.707	3.745	3.497	4.081	4.63.	4.378
Parameter b (conc.)	0.739	0.757	0.715	0.312	0.267	0.283	0.422	0.458
Parameter c (dur.)	0.608	0.477	0.706	1.095	1.965	0.713	0.738	0.763
f-value	130	520	811	280	241	270	1480	1510
Probability	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001	0.0001
Mean Squares	555	205	360	87	64	52	330	321
Log-WEIBULL Model								
Groups	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Gstat	19	6	11.7	2.8	4.2	0.79	3.5	3
Probability%	100	100	100	100	100	100	100	100
R-squared	58	60	59	58	41	69	76	79
Intercept a	−2.025	−1.802	−2.168	−1.355	−1.299	−1.184	−1.081	−1.162
Parameter b (conc.)	0.2	0.205	0.195	0.074	0.071	0.073	0.123	0.136
Parameter c (dur.)	0.154	0.113	0.184	0.294	0.505	0.173	0.173	0.182
f-value	115	440	760	271	81	210	1051	1090
Probability	0.0001	0.0001	0.0001	0.0001	0.0015	0.0001	0.0001	0.0001
Mean Squares	38	14	25	6	4	3	19	19

Group 1: Juvenile and adult salmonids
Group 2: Adult salmonids
Group 3: Juvenile salmonids
Group 4: Eggs and larvae of salmonids and non-salmonids
Group 5: Adult estuarine non-salmonids
Group 6: Adult freshwater non-salmonids
Group 7: Aquatic invertebrates
Group 8: Aquatic invertebrates and aquatic flora

All statistical analyses were performed using the software package SAS (SAS Inst. Inc., Toronto, Ontario, Canada). Prediction confidence intervals were also obtained on the model estimates (Appendices 5 to 13). Because predictions are made from the model curves, prediction confidence intervals, which are slightly wider than regular ones, were used for criteria development.

Representation of the data are 3-D plots of the Weibull model response surface for each data group (Figures 8 to 15). The data group plots demonstrate that there are differences in the concentration-duration toxicity response in each case. For example, Groups 3, 7 and 8 are influenced by suspended sediment concentration and duration equally. Groups 1 and 2 are influenced by concentration more than by duration. Groups 4-6 are influenced by duration more than by concentration. Differences between adult (Group 2) and juvenile (Group 3) salmonid curves can be explained by the fact that: 1) SEV scores have been chosen to reflect suspended particle size in the 0.5 - 75 µm range for Group 3 and 0.5 - 250 µm for Group 2; and 2) adult salmonids are more sensitive to shorter duration, higher concentrations of suspended sediments than juvenile salmonids. Groups 4-6, which include eggs and larvae of salmonids and non-salmonids, are clearly relatively more sensitive to the duration of the suspended sediment event than to the concentration. This is consistent with the knowledge that low concentrations of suspended sediments for long durations have an impact on the early life stages of fish (Newcombe and MacDonald 1991).

Along with 3-D plots of the data, SEV score matrices are given for the eight Weibull model generated response surfaces (Figures 16 to 23). The matrices permit estimation of the minimum concentrations and durations that trigger sublethal and lethal effects (Newcombe and Jensen 1996b). For visual effectiveness, four 2-D plots of the eight curves at different times (i.e., 1, 24, 336 and 7,392 h) showing prediction confidence intervals are illustrated (Appendices 6 to 13). The prediction confidence intervals are wide which is expected from this large data set made up of entries from multiple field and laboratory investigations with varying experimental conditions and species.

The next step was to determine which of the curves had the steepest slope at any given concentration and duration. To this end, linear regression on the suspended sediment concentrations were performed at 11 different event durations (i.e., 1, 3, 7, 24, 48, 144, 336, 1,176, 2,688, 7,392 and 20,160 hours) for each of the eight data groups. The steepest slope (2.08; 24 or 48 h) was observed in Group 2 (adult salmonids) (Table 10). By a simple extrapolation from Group 2 concentration response data at 24 h, an SEV score increase of one represents a 25 mg·L^{-1} increase in suspended sediments.

Table 10. Weibull model curve slopes for eight data groups.

Groups / Duration	1	2	3	4	5	6	7	8
1 h	1.783	1.966	1.592	0.64	0.63	0.69	1.46	1.33
3h	1.9	2.02	1.74	0.75	0.79	0.75	1.5	1.37
7h	1.97	2.06	1.85	0.82	0.82	0.8	1.5	1.37
24 h	2.03	2.08	1.96	0.86	0.58	0.84	1.44	1.33
48 h	2.05	2.08	1.99	0.83	0.36	0.85	1.37	1.28
144 h	2.05	2.07	2.01	0.71	0.09	0.84	1.23	1.16
336h	2.02	2.04	1.97	0.56	0.01	0.81	1.09	1.04
1176 h	1.92	1.97	1.86	0.32	1.06	0.74	0.86	0.84
2688 h	1.83	1.91	1.74	0.18	7E−07	0.67	0.7	0.7
7392 h	1.69	1.82	1.56	0.07	1E−18	0.56	0.52	0.53
20160 h	1.53	1.72	1.36	0.02	1E−18	0.44	0.36	0.38

Steepest slope = 2.08, Group 2, at 24 and 48 h
Group 1: Juvenile and adult salmonids
Group 2: Adult salmonids
Group 3: Juvenile salmonids
Group 4: Eggs and larvae of salmonids and non-salmonids
Group 5: Adult estuarine non-salmonids
Group 6: Adult freshwater non-salmonids
Group 7: Aquatic invertebrates
Group 8: Aquatic invertebrates and aquatic flora

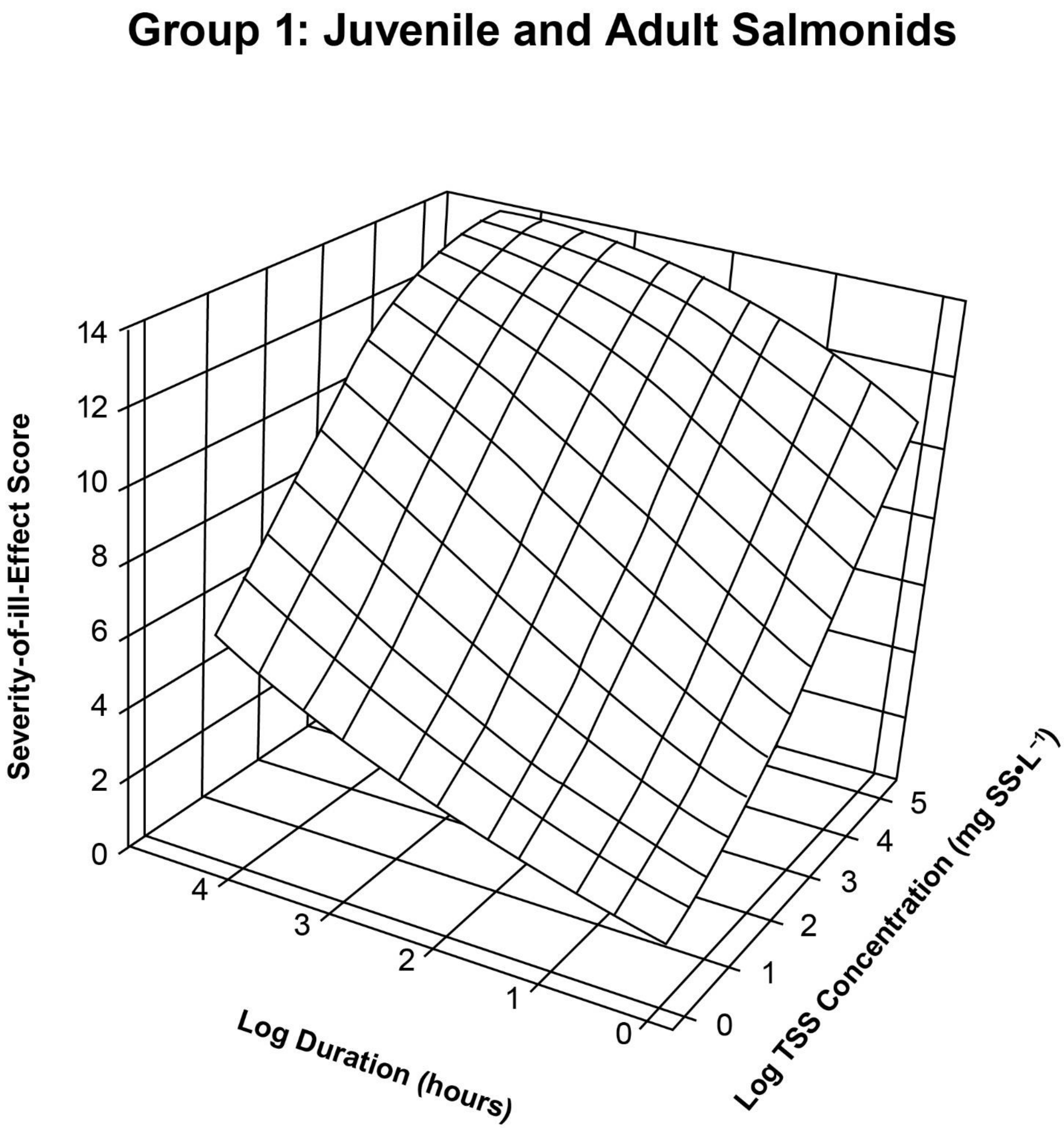

Figure 8. XYZ Cartesian plot for Group 1, juvenile and adult salmonids. Severity-of-ill-effects score vs. concentration and duration.

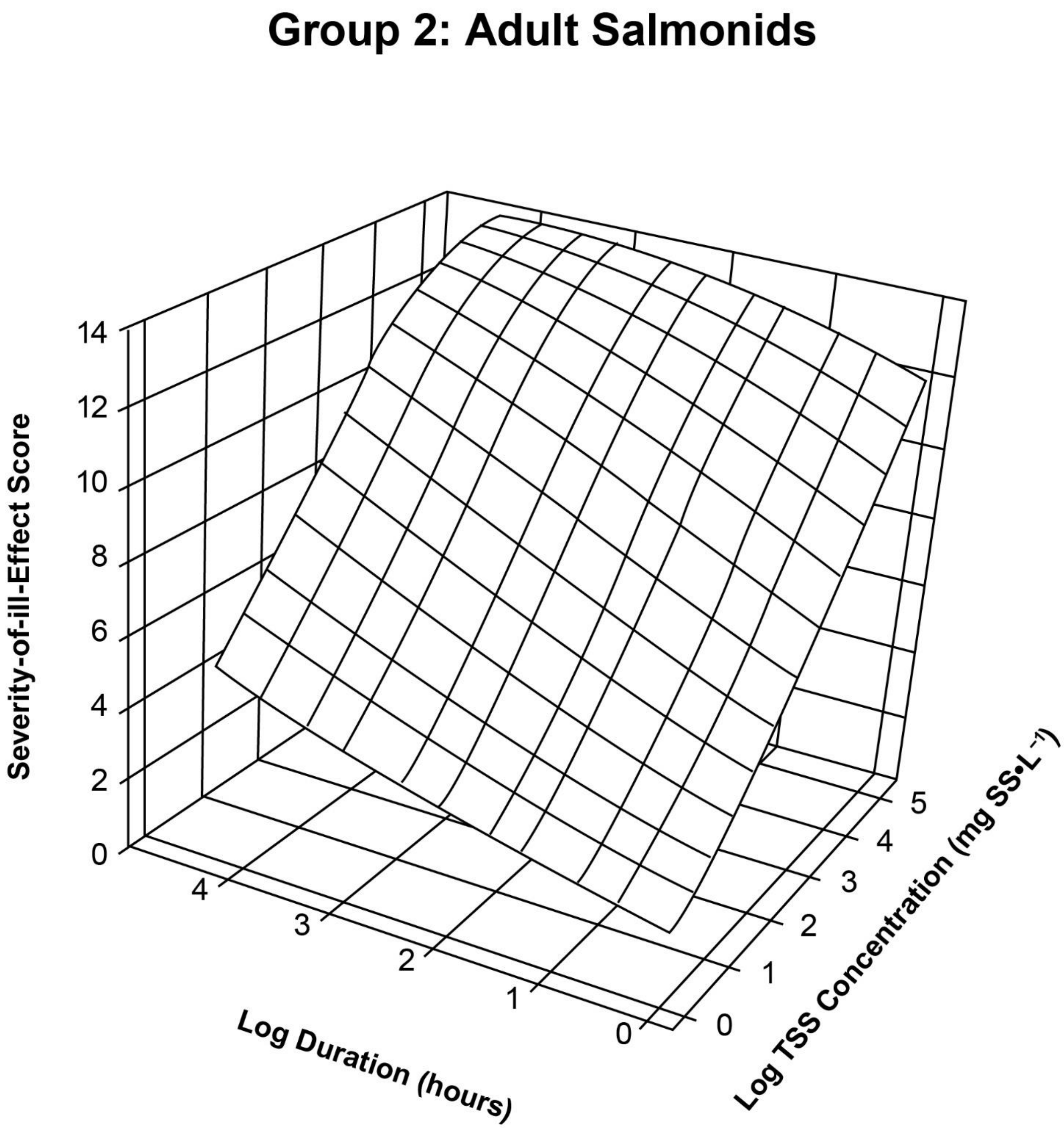

Figure 9. XYZ Cartesian plot for Group 2, adult salmonids. Severity-of-ill-effects score vs. concentration and duration.

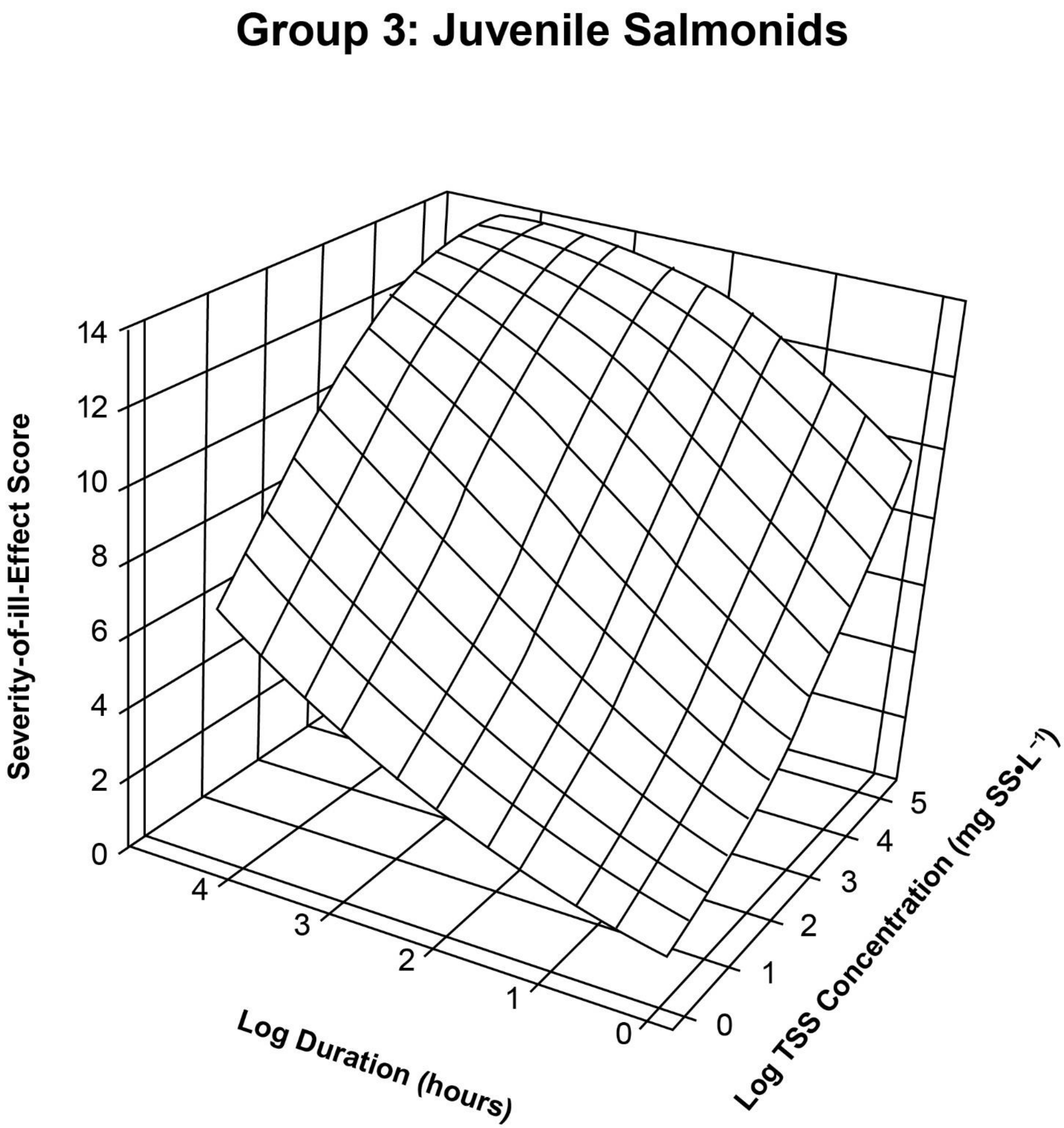

Figure 10. XYZ Cartesian plot for Group 3, juvenile salmonids. Severity-of-ill-effects score vs. concentration and duration.

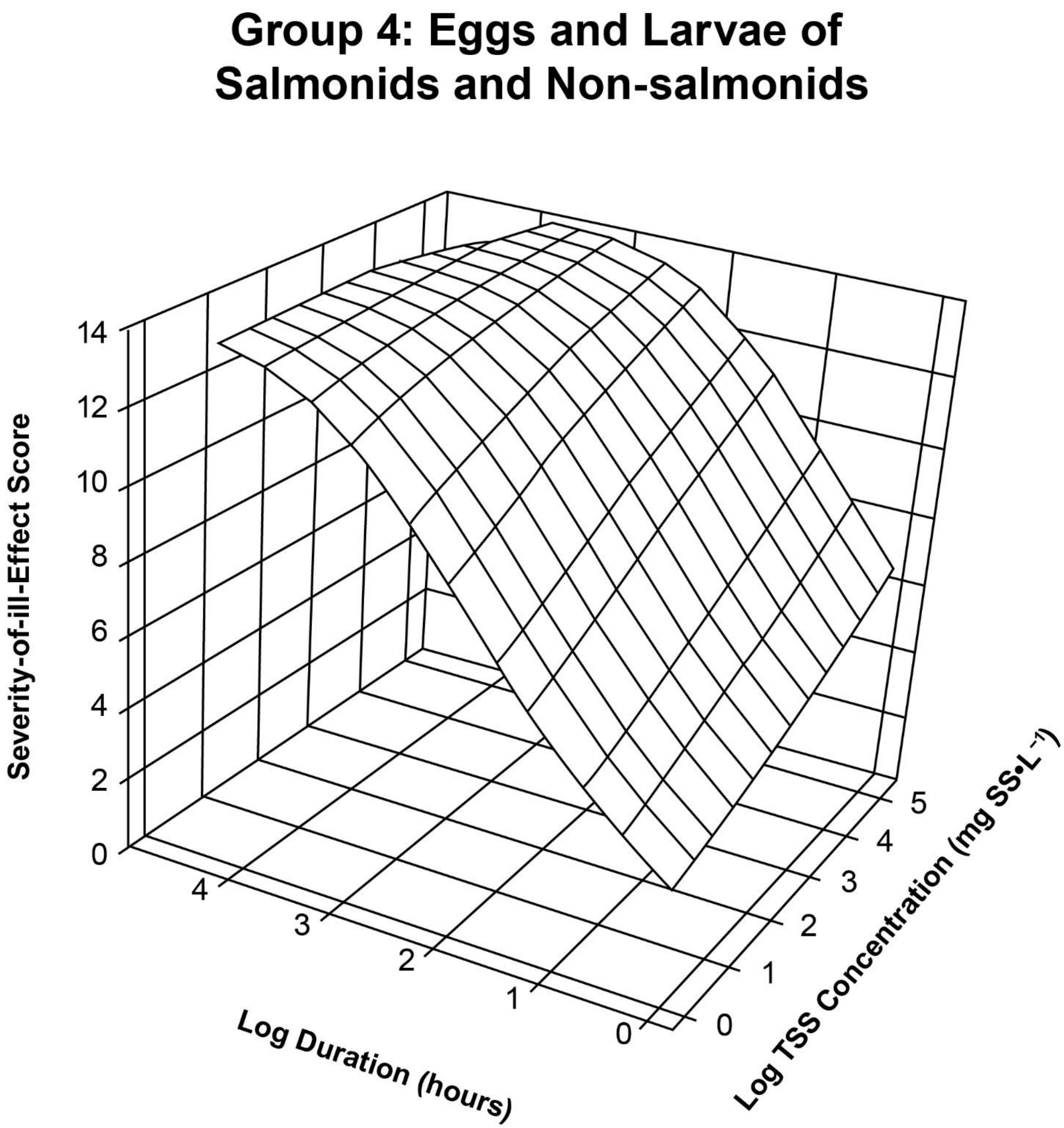

Figure 11. XYZ Cartesian plot for Group 4, eggs and larvae of salmonids and non-salmonids. Severity-of-ill-effects score vs. concentration and duration.

Group 5: Adult Estuarine Non-salmonids

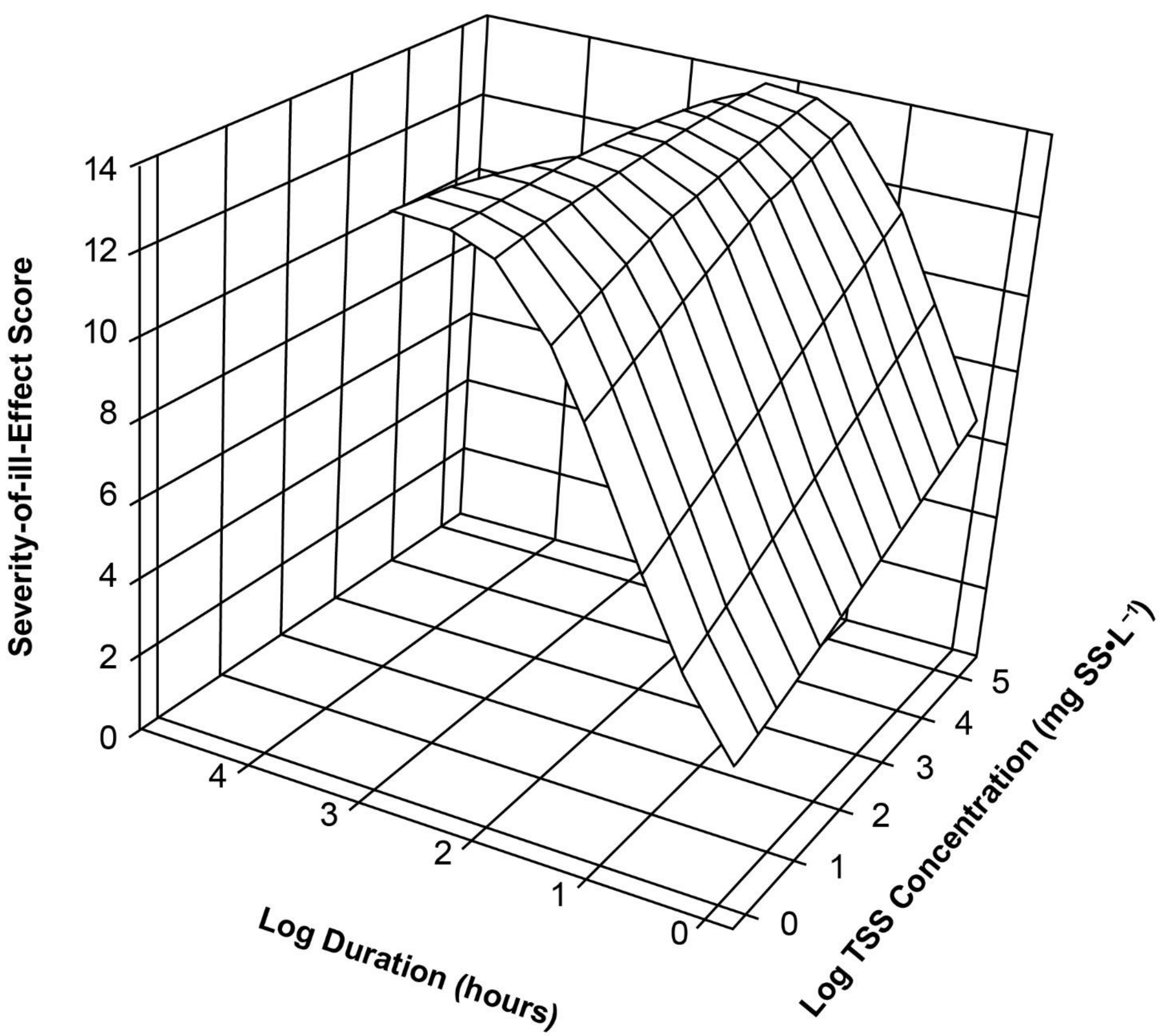

Figure 12. XYZ Cartesian plot for Group 5, adult estuarine non-salmonids. Severity-of-ill-effects score vs. concentration and duration.

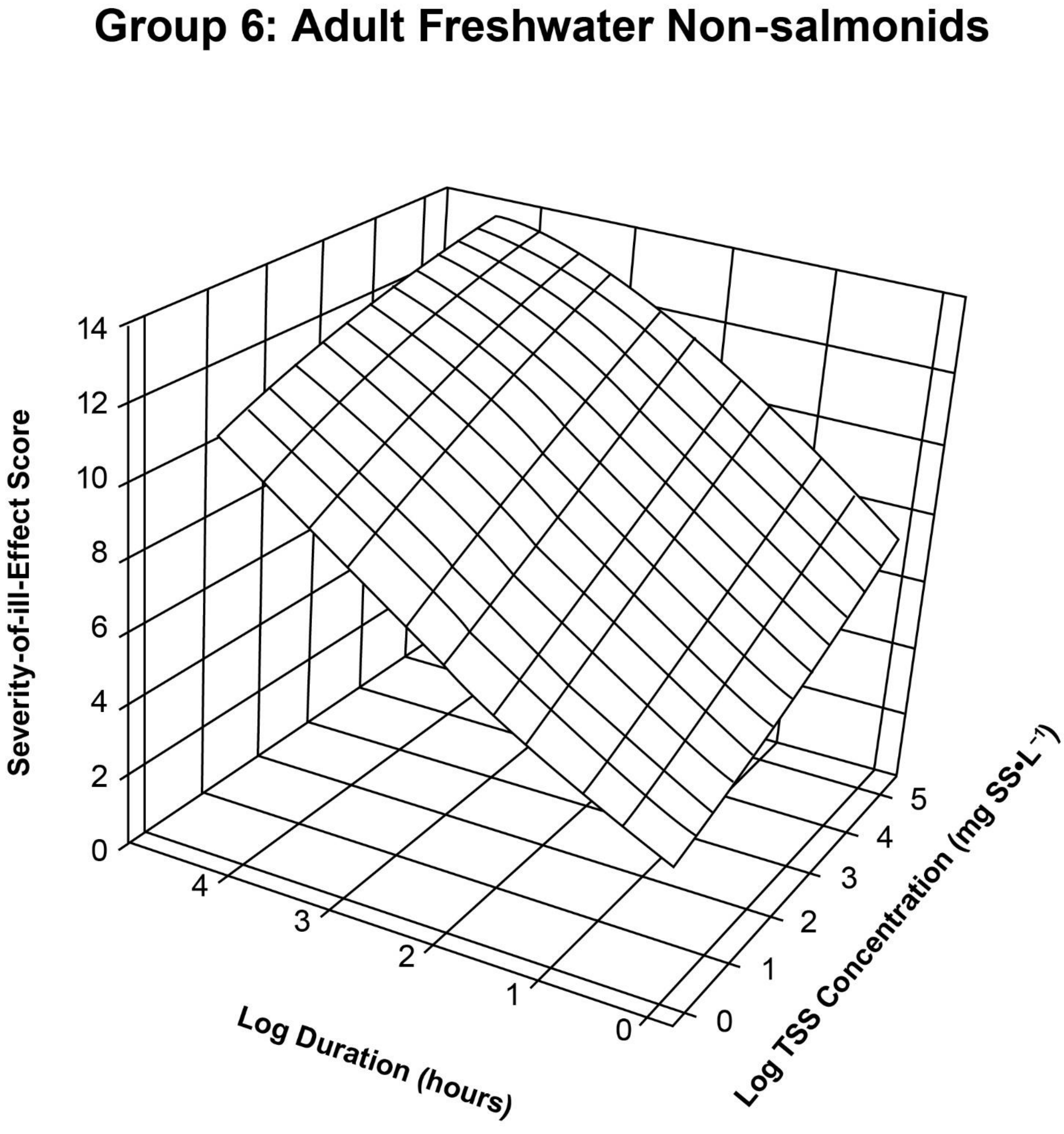

Figure 13. XYZ Cartesian plot for Group 6, adult freshwater non-salmonids. Severity of-ill-effects score vs. concentration and duration.

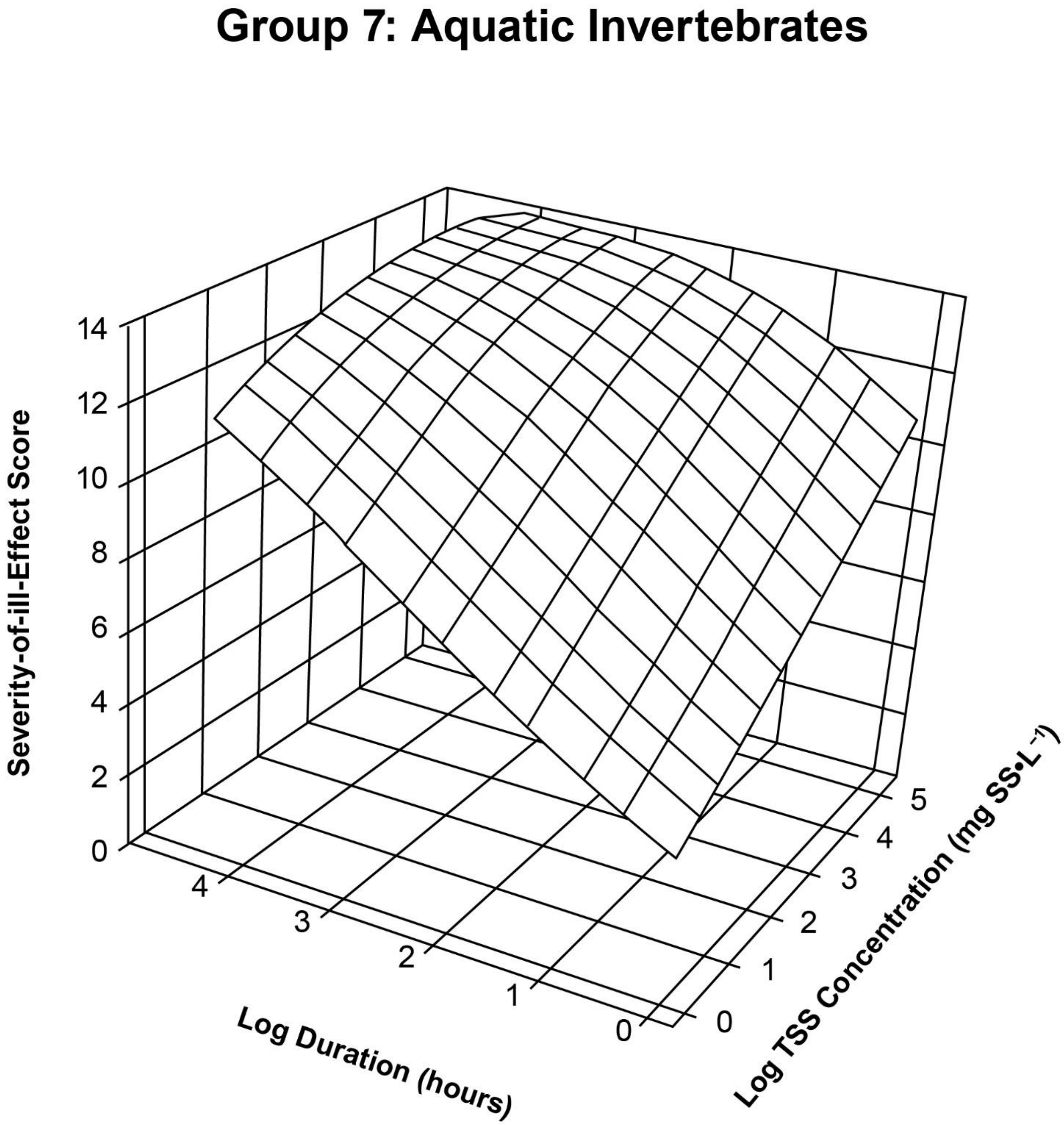

Figure 14. XYZ Cartesian plot for Group 7, aquatic invertebrates. Severity-of-ill-effects score vs. concentration and duration.

Group 8: Aquatic Invertebrates and Aquatic Flora

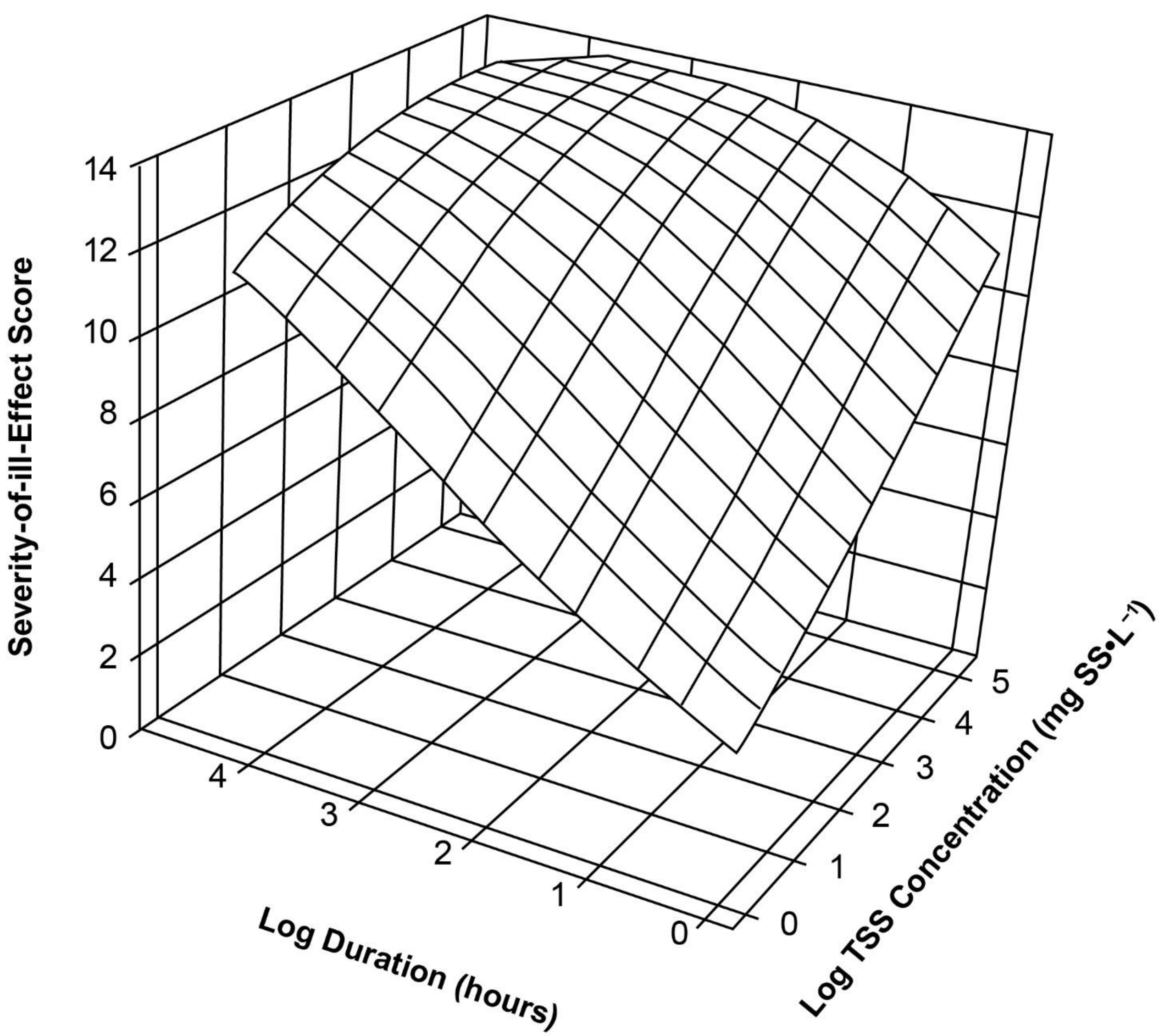

Figure 15. XYZ Cartesian plot for Group 8, aquatic invertebrates and flora. Severity of-ill-effects score vs. concentration and duration.

Duration of exposure to SS (loge hours)

Average severity-of-ill-effect scores (calculated)

Concentration (mg SS/L)	0	1	2	3	4	5	6	7	8	9	10	LOGe (mg SS/L)
162,755	11	12	12	13	13	13	14	14	14	14	14	12
59,874	10	11	11	12	12	13	13	14	14	14	14	11
22,026	9	10	10	11	12	12	13	13	14	14	14	10
8,103	8	9	9	10	11	12	12	13	13	13	14	9
2,981	7	8	8	9	10	11	11	12	13	13	13	8
1,097	6	7	7	8	9	10	10	11	12	12	13	7
403	5	6	6	7	8	9	9	10	11	12	12	6
148	4	5	5	6	7	8	8	9	10	11	11	5
55	4	4	5	5	6	7	7	8	g	10	10	4
20	3	3	4	5	5	6	6	7	8	9	9	3
7	2	3	3	4	4	5	5	6	7	8	8	2
3	2	2	3	3	4	4	5	5	6	7	7	1
1	2	2	2	3	3	3	4	5	5	6	6	0
	1	3	7	1	2	6	2	7	4	11	30	
	Hours			Days			Weeks		Months			

Figure 16. Average severity-of-ill-effects (SEV) scores matrix. Group 1, Juvenile and adult salmonids. N = 171.

Duration of exposure to SS (loge hours)

Average severity-of-ill-effect scores (calculated)

Concentration (mg SS/L)	0	1	2	3	4	5	6	7	8	9	10	LOGe (mg SS/L)
162,755	12	12	13	13	13	14	14	14	14	14	14	12
59,874	11	12	12	13	13	13	13	14	14	14	14	11
22,026	10	11	11	12	12	13	13	13	13	14	14	10
8,103	9	10	10	11	11	12	12	13	13	13	13	9
2,981	8	9	9	10	10	11	11	12	12	13	13	8
1,097	7	8	8	9	9	10	10	11	11	12	12	7
403	6	7	7	8	8	9	9	10	10	11	12	6
148	5	6	6	7	7	8	8	9	9	10	11	5
55	4	5	5	6	6	7	7	8	8	9	10	4
20	4	4	4	5	5	6	6	7	7	8	9	3
7	3	3	4	4	4	5	5	6	6	7	7	2
3	3	3	3	4	4	4	5	5	6	6	7	1
1	2	2	3	3	3	4	4	4	5	5	6	0
	1	3	7	1	2	6	2	7	4	11	30	
	Hours			Days			Weeks		Months			

Figure 17. Average severity-of-ill-effects (SEV) scores matrix. Group 2, adult salmonids. N = 63.

Duration of exposure to SS (loge hours)

Average severity-of-ill-effect scores (calculated)

Concentration (mg SS/L)	0	1	2	3	4	5	6	7	8	9	10	LOGe (mg SS/L)
162,755	10	11	11	12	13	13	14	14	14	14	14	12
59,874	9	10	11	12	12	13	13	14	14	14	14	11
22,026	8	9	10	11	11	12	13	13	14	14	14	10
8,103	7	8	9	10	10	11	12	13	13	14	14	9
2,981	6	7	8	9	9	10	11	12	13	13	14	8
1,097	5	6	7	8	8	9	10	11	12	13	13	7
403	4	5	6	7	7	8	9	10	11	12	13	6
148	4	4	5	6	6	7	8	9	10	11	12	5
55	3	4	4	5	6	7	7	8	9	10	11	4
20	3	3	4	4	5	6	6	7	8	9	10	3
7	2	3	3	4	4	5	5	6	7	8	9	2
3	2	2	3	3	4	4	5	6	6	7	8	1
1	2	2	2	3	3	3	4	5	5	6	7	0
	1	3	7	1	2	6	2	7	4	11	30	
	Hours			Days			Weeks		Months			

Figure 18. Average severity-of-ill-effects (SEV) scores matrix. Group 3, Juvenile salmonids. N = 108.

Duration of exposure to SS (loge hours)

Average severity-of-ill-effect scores (calculated)

Concentration (mg SS/L)	0	1	2	3	4	5	6	7	8	9	10	LOGe (mg SS/L)
162,755	7	8	9	11	12	13	14	14	14	14	14	12
59,874	6	8	9	11	12	13	13	14	14	14	14	i1
22,026	6	7	9	10	11	13	13	14	14	14	14	10
8,103	6	7	8	10	11	12	13	14	14	14	14	9
2,981	5	7	8	10	11	12	13	14	14	14	14	8
1,097	5	6	8	9	10	12	13	14	14	14	14	7
403	5	6	7	9	10	12	12	13	14	14	14	6
148	4	6	7	9	10	11	12	13	14	14	14	5
55	4	5	6	8	9	11	12	13	14	14	14	4
20	4	5	6	8	9	11	12	13	13	14	14	3
7	4	5	6	7	8	10	11	13	13	14	14	2
3	3	4	5	7	8	10	11	13	13	14	14	1
1	3	4	5	7	8	9	11	12	13	14	14	0
	1	3	7	1	2	6	2	7	4	11	30	
	Hours			Days			Weeks		Months			

Figure 19. Average severity-of-ill-effects (SEV) scores matrix. Group 4, eggs and larvae of salmonids and non-salmonids. N = 43.

Duration of exposure to SS (loge hours)

Concentration (mg SS/L)	0	1	2	3	4	5	6	7	8	9	10	LOGe (mg SS/L)
	Average severity-of-ill-effect scores (calculated)											
162,755	7	9	11	13	14	14	14	14	14	14	14	12
59,874	6	9	11	13	14	14	14	14	14	14	14	11
22,026	6	9	11	13	14	14	14	14	14	14	14	10
8,103	6	8	10	13	14	14	14	14	14	14	14	9
2,981	5	8	10	13	14	14	14	14	14	14	14	8
1,097	5	8	10	13	13	14	14	14	14	14	14	7
403	5	7	9	12	13	14	14	14	14	14	14	6
148	5	7	9	12	13	14	14	14	14	14	14	5
55	4	7	9	12	13	14	14	14	14	14	14	4
20	4	6	8	11	13	14	14	14	14	14	14	3
7	4	6	8	11	12	14	14	14	14	14	14	2
3	4	6	8	11	12	14	14	14	14	14'	14	1
1	3	5	7	10	12	14	14	14	14	14	14	0
	1	3	7	1	2	6	2	7	4	11	30	
	Hours			Days			Weeks		Months			

Figure 20. Average severity-of-ill-effects (SEV) scores matrix. Group 5, adult estuarine non-salmonids. N = 28.

Duration of exposure to SS (loge hours)

Concentration (mg SS/L)	0	1	2	3	4	5	6	7	8	9	10	LOGe (mg SS/L)
	Average severity-of-ill-effect scores (calculated)											
162,755	7	8	9	10	11	12	12	13	13	14	14	12
59,874	7	8	9	10	10	11	12	13	13	13	14	11
22,026	7	8	8	9	10	11	12	12	13	13	14	10
8,103	6	7	8	9	10	11	11	12	13	13	13	9
2,981	6	7	8	9	9	10	11	12	12	13	13	8
1,097	6	6	7	8	9	10	11	12	12	13	13	7
403	5	6	7	8	8	9	10	11	12	12	13	6
148	5	6	6	7	8	9	10	11	12	12	13	5
55	5	5	6	7	8	9	9	11	11	12	13	4
20	4	5	6	7	7	8	9	10	11	12	12	3
7	4	5	5	6	7	8	9	10	10	11	12	2
3	4	5	5	6	7	8	8	9	10	11	12	1
1	4	4	5	6	6	7	8	9	10	11	11	0
	1	3	7	1	2	6	2	7	4	11	30	
	Hours			Days			Weeks		Months			

Figure 21. Average severity-of-ill-effects (SEV) scores matrix. Group 6, adult freshwater non-salmonids. N = 22.

Duration of exposure to SS (loge hours)

Average severity-of-ill-effect scores (calculated)

Concentration (mg SS/L)	0	1	2	3	4	5	6	7	8	9	10	LOGe (mg SS/L)
162,755	11	12	12	13	13	14	14	14	14	14	14	12
59,874	10	11	12	13	13	13	14	14	14	14	14	11
22,026	10	11	11	12	13	13	13	14	14	14	14	10
8,103	9	10	11	12	12	13	13	14	14	14	14	9
2,981	8	9	10	11	12	12	13	13	14	14	14	8
1,097	8	9	9	11	11	12	12	13	13	14	14	7
403	7	8	9	10	11	11	12	13	13	14	14	6
148	7	7	8	9	10	11	11	12	13	13	14	5
55	6	7	8	9	9	10	11	12	12	13	13	4
20	5	6	7	8	9	10	10	11	12	13	13	3
7	5	6	6	7	8	9	10	11	11	12	13	2
3	5	5·	6	7	7	8	9	10	11	12	12	1
1	4	5	5	6	7	8	8	10	10	11	12	0
	1	3	7	1	2	6	2	7	4	11	30	
	Hours			Days			Weeks		Months			

Figure 22. Average severity-of-ill-effects (SEV) scores matrix. Group 7, aquatic invertebrates. N = 69.

Duration of exposure to SS (loge hours)

Average severity-of-ill-effect scores (calculated)

Concentration (mg SS/L)	0	1	2	3	4	5	6	7	8	9	10	LOGe (mg SS/L)
162,755	11	12	13	13	13	14	14	14	14	14	14	12
59,874	11	11	12	13	13	14	14	14	14	14	14	11
22,026	10	11	12	12	13	13	14	14	14	14	14	10
8,103	9	10	11	12	12	13	13	14	14	14	14	9
2,981	8	10	10	11	12	13	13	14	14	14	14	8
1,097	8	9	10	11	11	12	13	13	14	14	14	7
403	7	8	9	10	11	12	12	13	13	14	14	6
148	6	7	8	9	10	11	12	13	13	13	14	5
55	6	7	8	9	9	10	11	12	13	13	13	4
20	5	6	7	8	9	10	10	11	12	13	13	3
7	5	5	6	7	8	9	10	11	11	12	13	2
3	4	5	6	7	7	8	9	10	11	12	12	1
1	4	4	5	6	7	8	8	9	10	11	12	0
	1	3	7	1	2	6	2	7	4	11	30	
	Hours			Days			Weeks		Months			

Figure 23. Average severity-of-ill-effects (SEV) scores matrix. Group 8, aquatic invertebrates and flora. N = 61.

5. WILDLIFE

5.1 Effects

Suspended sediment effects on wildlife are characterized by effects through the food chain and on behavioural responses. For example, effects on primary productivity can have repercussions all the way through to top predators (Singleton 1985). Primary productivity may be reduced due to reduced transmittance of light in turbid waters; these systems characterized by increased inputs of organic material, however, can have thriving secondary productivities (Power et al. 1994). Recent investigations reporting the effects of habitat disturbances such as increased turbidity, demonstrate that loons and grebes have behavioral adaptation specific to aquatic habitats (Alvo et al. 1988). It was shown that the fledging success of loons on oligothrophic lakes in Saskatchewan and Ontario was twice that of eutrophic lakes in Alberta and Minnesota (Fox et al. 1980). The turbidity of the water affects habitat selection and foraging efficiency for Pacific loons, as these are sight feeders (Heglund et al. 1994). In Quebec it was demonstrated that loons preferred clear lakes (DesGranges 1989). Some turtles (Blanding turtle), however, were shown to concentrate in the turbid waters of lake inflows, while other turtles (Snapping turtle) reached their highest densities on clear water lakes (Power et al. 1994).

5.2 Literature criteria

Wildlife criteria from other jurisdictions have been summarized in Tables 11 and 12. The criteria have not changed since 1985 (Singleton 1985) and are reiterated in this document for comparative purposes to other criteria and for practical reasons. The reader is referred to Singleton (1985) for a discussion on the interpretation of these criteria.

Table 11. Turbidity criteria for wildlife.

Criteria Statement	Criteria Values	Reference
Surface waters not to exceed more than 25 JTU over natural turbidity	25 JTU increase	Environment Saskatchewan 1975; Alberta Department of the Environment 1977
Shall not exceed 25 NTU above natural condition level. For all lake waters, shall not exceed 5 NTU over natural conditions	5-25 NTU increase	Alaska Department of Environmental Conservation 1979
Criteria range from no increase above naturally occurring turbidity to a maximum allowable increase of 10 NTU above background, or that which may be harmful or create a nuisance, depending upon the classification	0-10 NTU increase	Montana Health and Environmental Sciences 1980
Wastewater from point source discharges must not increase turbidity outside the mixing zone by: -more than 5 NTU over background when background turbidity is ≤50 NTU -more than 10% over background when background turbidity is >50 NTU to a maximum increase of 25 NTU	5-25 NTU increase	Idaho Department of Health and Welfare 1980
Criteria range from ≤5 NTU over background to ≤10 NTU over background when the background turbidity	0-10 NTU increase	State of Washington 1982

Criteria Statement	Criteria Values	Reference
is ≤50 NTU, or not more than a 10-20% increase when background >50 NTU, depending on classification		
Suggests several alternatives to the 1979 Alaska Water Quality Standards, but category also includes aquatic life		Alaska Department of Environmental Conservation 1982

Table 12. Non-filterable residue criteria for wildlife.

Criteria Statement	Criteria Values	Reference
Not to be increase by more than 10 mg/L over background value	10 mg/L increase	Environment Saskatchewan 1975; Alberta Department of the Environment 1977
No sediment loads (suspended or deposited) which can cause adverse effects.		Alaska Department of Environmental Conservation 1979
No man-caused suspended matter of any kind in concentrations causing nuisance or objectionable conditions or that may adversely affect designated or protected beneficial uses		Idaho Department of Health and Welfare 1980
Suggests several alternatives to the 1979 Alaska Water Quality Standards, but category also includes aquatic life	25 mg/L	Alaska Department of Environmental Conservation 1982

5.3 Recommended guidelines

Guidelines to protect wildlife from excessive suspended sediment in B.C. waters are presented in terms of turbidity and non-filterable residue. The guidelines are based on natural background levels as follows:

5.3.1 *Turbidity*

Induced turbidity should not exceed 10 NTU when background turbidity is ≤50 NTU, nor should induced turbidity be more than 20% of background when background is >50 NTU.

5.3.2 *Suspended sediments*

Induced non-filterable residue should not exceed 20 mg/L when background suspended sediment (measured as non-filterable residue) is ≤100 mg/L, nor should induced suspended sediment be more than 20% of background when background is >100 mg/L.

5.4 Rationale

The guidelines for the protection of wildlife are those that were established formerly by Singleton (1985). No new evidence was found that could be used to update the criteria. The wildlife guidelines were based on the former aquatic life guidelines. These previous guidelines accounted for the fact that because early life stages of wildlife did not dwell in aquatic systems, these were not as affected as aquatic biota. The guideline values are somewhat arbitrary and are subject to change with site-specific conditions.

6. LIVESTOCK

Livestock are defined as all animal species kept for economic profit (Caux et al. 1994). Thus, apart from traditional livestock such as cattle, sheep, swine, goats, horses and poultry, other species such as rabbit, fox, mink, elk, and buffalo, which are often more sensitive, are to be considered. The adsorptive properties of suspended particles can lead to a concentration of heavy metal ions and biocides in turbid waters (Health Canada 1995). Coliform bacteria have been detected in waters with turbidity higher than 2 NTU even when there is a free chlorine residual (Health Canada 1995). As livestock water originates from surface and groundwater that may or may not be treated, the physical, chemical, and biological composition of the water will substantially vary from region to region.

6.1 Effects

Of concern to livestock producers are the possible pathogenic microorganisms that can be associated with suspended sediment in livestock drinking water. Livestock should not be watered from lakes where blue-green algae have rendered these waters turbid (CCME 1987).

6.2 Literature criteria

Livestock watering criteria from other jurisdictions have been summarized in Tables 13 and 14. The criteria have not changed since 1985 (Singleton 1985) and are reiterated in this document for comparative purposes to other criteria and for practical reasons. The reader is referred to Singleton (1985) for a discussion on the interpretation of these criteria.

Table 13. Turbidity criteria for livestock watering.

Criteria Statement	Criteria Values	Reference
Surface waters not to exceed more than 25 JTU over natural turbidity	25 JTU increase	Environment Saskatchewan 1975; Alberta Department of the Environment 1977
Shall not cause detrimental effects on indicated use		Alaska Department of Environmental Conservation 1979
Criteria range from no increase above naturally occurring turbidity to a maximum allowable increase of 10 NTU above background, or that which may be harmful or create a nuisance, depending upon the classification	0-10 NTU increase	Montana Health and Environmental Sciences 1980
Wastewater from point source discharges must not increase turbidity outside the mixing zone by: -more than 5 NTU over background when background turbidity is ≤50 NTU -more than 10% over background when background turbidity is >50 NTU to a maximum increase of 25 NTU	5-25 NTU increase	Idaho Department of Health and Welfare 1980

Criteria Statement	Criteria Values	Reference
Criteria range from ≤5 NTU over background to ≤10 NTU over background when the background turbidity is s 50 NTU, or not more than a 10-20% increase when background >50 NTU, depending on classification	5-10 NTU increase	State of Washington 1982

Table 14. Non-filterable residue criteria for livestock watering.

Criteria Statement	Criteria Values	Reference
Not to be increase by more than 10 mg/L over background value	10 mg/L increase	Environment Saskatchewan 1975; Alberta Department of the Environment 1977
Shall be free of particles of 0.074 mm or coarser. Shall not exceed 200 mg/L for an extended period of time	200 mg/L	Alaska Department of Environmental Conservation 1979
No man-caused suspended matter of any kind in concentrations causing nuisance or objectionable conditions or that may adversely affect designated or protected beneficial uses		Idaho Department of Health and Welfare 1980

6.3 Recommended guidelines

Guidelines to protect livestock from excessive suspended sediment in livestock waters are presented in terms of turbidity and non-filterable residue. The guidelines are based on natural background levels.

6.3.1 *Turbidity*

Induced turbidity should not exceed 5 NTU when background turbidity is ≤50 NTU, nor should induced turbidity be more than 10% of background when background is >50 NTU.

6.3.2 *Suspended sediments*

Induced non-filterable residue should not exceed 10 mg/L when background suspended sediments (measured as non-filterable residue) is ≤100 mg/L, nor should induced suspended sediment be more than 10% of background when background is >100 mg/L.

6.4 Rationale

Because the occurrence and persistence of micro-organisms that can have detrimental effects on livestock have been correlated with turbidity, the livestock watering guidelines have been derived to account for the effects of these harmful pathogens. In this regard, young dairy cattle can contract diarrhea by drinking waters that contain 1 in 100 ml of total coliform bacteria (Ministère de l'Environnement du Québec 1990). Reduced turbidity by itself, however, cannot be assumed to prevent waterborne diseases (American Water Works Association 1981). Furthermore, because disinfection efficiency is reduced substantially with an increase in turbidity (Health Canada 1995), it may become overtly costly to livestock producers who require clean water. Thus, a turbidity guideline of 5 NTU and suspended sediment guideline of 10 mg/L should be protective of livestock in British Columbia. Site-specific adjustments may be needed to either protect very sensitive livestock (e.g., poultry) by deriving a more restrictive objective or, guidelines may be raised to permit higher objectives. The latter may be encountered in cases where livestock are tolerant of suspended sediment concentrations (e.g., free-ranging cattle in pastures) and the water source (e.g., groundwater) has had historical records of being bacteria free.

7. IRRIGATION

7.1 Effects

The effects of suspended sediments in irrigation waters were summarized by Singleton (1985) as being an impedance of seed emergence and photosynthetic activity and growth, and a reduction in desirability for food consumption (e.g., dirty lettuce). The effects, however, are not confined to biological effects as excessive suspended sediment can also clog the mechanical components of irrigation systems (Singleton 1985).

7.2 Literature criteria

Irrigation criteria from other jurisdictions have been summarized in Tables 15 and 16. The criteria have not changed since 1985 (Singleton 1985) and are reiterated in this document for comparative purposes to other criteria and for practical reasons. The reader is referred to Singleton (1985) for a discussion on the interpretation of these criteria.

Table 15. Turbidity criteria for irrigation.

Criteria Statement	Criteria Values	Reference
Surface waters not to exceed more than 25 JTU over natural turbidity	25 JTU increase	Environment Saskatchewan 1975; Alberta Department of the Environment 1977
Shall not cause detrimental effects on indicated use		Alaska Department of Environmental Conservation 1979
Criteria range from no increase above naturally occurring turbidity to a maximum allowable increase of 10 NTU above background, or that which may be harmful or create a nuisance, depending upon the classification	0-10 NTU increase	Montana Health and Environmental Sciences 1980
Wastewater from point source discharges must not increase turbidity outside the mixing zone by: -more than 5 NTU over background when background turbidity is ≤50 NTU -more than 10% over background when background turbidity is >50 NTU to a maximum increase of 25 NTU	5-25 NTU increase	Idaho Department of Health and Welfare 1980
Criteria range from ≤5 NTU over background to ≤10 NTU over background when the background turbidity is ≤50 NTU, or not more than a 10-20% increase when background >50 NTU, depending on	5-10 NTU increase	State of Washington 1982

Criteria Statement	Criteria Values	Reference
classification		

Table 16. Non-filterable residue criteria for irrigation.

Criteria Statement	Criteria Values	Reference
Not to be increase by more than 10 mg/L over background value	10 mg/L increase	Environment Saskatchewan 1975; Alberta Department of the Environment 1977
Shall be free of particles of 0.074 mm or coarser. Shall not exceed 200 mg/L for an extended period of time	200 mg/L	Alaska Department of Environmental Conservation 1979
No man-caused suspended matter of any kind in concentrations causing nuisance or objectionable conditions or that may adversely affect designated or protected beneficial uses		Idaho Department of Health and Welfare 1980

7.3 Recommended guidelines

Guidelines to protect crops and irrigation equipment from excessive suspended sediment in B.C. waters are presented in terms of turbidity and non-filterable residue. The guidelines are based on natural background levels.

7.3.1 *Turbidity*

Induced turbidity should not exceed 10 NTU when background turbidity is ≤50 NTU, nor should induced turbidity be more than 20% of background when background is >50 NTU.

7.3.2 *Suspended sediments*

Induced non-filterable residue should not exceed 20 mg/L when background suspended sediments (measured as non-filterable residue) is ≤100 mg/L, nor should induced suspended sediment be more than 20% of background when background is >100 mg/L.

7.4 Rationale

The guidelines for the protection of crops and irrigation equipment are those established formerly by Singleton (1985). No new evidence was found that could be used to update the guidelines. The reader is referred to Singleton (1985) for a discussion on the rationale for these guidelines.

8. RESEARCH AND DEVELOPMENT NEEDS

The guidelines developed to protect aquatic life from excessive suspended sediment loads are the first effects-based criteria to be recommended for this parameter.

Research is needed to obtain effects data for many concentration-duration exposures within groups of organisms (e.g., invertebrates and flora) in order to better characterize response curves and to increase the reliability of the severity-of-ill-effects model approach (SEV).

The SEV approach will be used concurrently with a sampling protocol to determine whether guideline exceedances are occurring, an approach that relies on the characterization of British Columbia streams and rivers. A province-wide data base describing these water bodies in the long-term would facilitate the procurement of base or background level data essential for scientific evaluation and management decisions. This type of continuous monitoring would, for example, identify clear periods in the stream under study and establish turbidity - suspended sediment relationships required to set site-specific objectives.

Information is generally lacking on the effects of bedload sediments on aquatic organisms. Future research should focus on evaluating the short- and long-term effects of bedload sediments on periphyton, benthic macroinvertebrates, and fish species that associate strongly with the streambed substrate. Such research will need to be supported by standard methods for measuring bedload transport in streams.

While a substantial quantity of data was located on the effects of deposited sediment on stream-dwelling fish species, little information was found on the other components of freshwater ecosystems. Future research should focus on evaluating the effects of deposited sediment on periphyton and benthic macroinvertebrates. More information is also needed on the effects of intrusions of fine sediment into streambed substrates on benthic-associated fish species (i.e., through the loss or degradation of rearing habitats).

In the 1985 guidelines document and in this current document, information on livestock and wildlife were scarce or not available. Guideline levels or narratives would benefit from basic toxicological data on suspended sediment effects on livestock and wildlife.

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10. APPENDICES

Appendix 1. Dose-response database for fish exposed to suspended sediment (adapted from Newcombe, 1994; Newcombe and Jensen 1996)

		Sediment dose		Fish response		
Species	**Life stage†**	**Exposure concentration (mg/L)**	**Exposure duration (hours)**	**SEV††**	**Description**	**References**
Adult salmonids and rainbow smelt (freshwater, groups 1 and 2)						
Grayling (Arctic)	A	100	0.10	3	Fish avoided turbid water.	Suchanek et al. (1984a. 1984b)
Grayling (Arctic)	A	100	1,008	8	Fish had decreased resistance to environmental stresses.	McLeay et al. 1984
Grayling (Arctic)	A	100	1,008	9	Impaired feeding.	McLeay et al. 1984
Grayling (Arctic)	A	100	1,008	9	Reduced growth.	McLeay et al. 1984
Salmon	A	25	4	4	Feeding activity reduced.	Phillips 1970
Salmon	A	17	24	4	Feeding behavior apparently reduced.	Townsend (1983); Ott (1984).
Salmon	A	1,650	240	7	Loss of habitat caused by excessive sediment transport.	Coats et al. (1985)
Salmon	A	75	168	7	Reduced quality of rearing habitat.	Slaney et al. (1977b)
Salmon	A	210	24	10	Fish abandoned their traditional spawning habitat.	Hamilton (1961)
Salmon (Atlantic)	A	2,500	24	10	Increased risk of predation.	Gibson (1933)
Salmon (chinook)	A	650	168	5	No histological signs of damage to olfactory epithelium.	Brannon et al. (1981)
Salmon (chinook)	A	350	0.2	7	Home water preference disrupted.	Whitman and others (1982)
Salmon (chinook)	A	650	168	7	Homing behavior normal, but fewer test fish returned.	Whitman and others (1982)
Salmon (chinook)	A	39,300	24	10	No mortality (VA, <5 - 100 µm; median, <15 µm).	Newcomb and Flagg (1983)
Salmon (chinook)	A	82,400	6	12	Mortality rate 60% (VA, <5 - 100 µm).	Newcomb and Flagg (1983)
Salmon (chinook)	A	207,000	1	14	Mortality rate 100% (VA, <5 - 100 µm).	Newcomb and Flagg (1983)
Salmon (Pacific)	A	525	588	10	No mortality (other end points not investigated).	Griffin (1938)
Salmon (sockeye)	A	500	96	8	Plasma glucose levels increased 39%.	Servizi and Martens (1987)
Salmon (sockeye)	A	1,500	96	8	Plasma glucose levels increased 150%.	Servizi and Martens (1987)
Salmon (sockeye)	A	39,300	24	10	No mortality (VA, <5 - 100 µm; median <15 µm).	Newcomb and Flagg (1983)
Salmon (sockeye)	A	82,400	6	12	Mortality rate 60% (VA, <5 - 100 µm; median <15 µm).	Newcomb and Flagg (1983)
Salmon (sockeye)	A	207,000	1	14	Mortality rate 100% (VA).	Newcomb and Flagg (1983)
Smelt (rainbow)	A	4	168	7	Increased vulnerability to predation.	Swenson (1978)
Steelhead	A	500	3	5	Signs of sublethal stress (VA).	Redding and Schreck (1980)
Steelhead	A	1,650	240	7	Loss of habitat caused by excessive sediment transport.	Coats et al. (1985)
Steelhead	A	500	9	8	Blood cell count and blood chemistry change.	Redding and Schreck (1980)
Trout	A	17	24	4	Feeding behavior apparently reduced.	Townsend (1983); Ott (1984)

		Sediment dose		Fish response		
Species	Life stage[†]	Exposure concentration (mg/L)	Exposure duration (hours)	SEV[††]	Description	References
Trout	A	75	168	7	Reduced quality of rearing habitat.	Slaney et al. (1977b)
Trout	A	270	312	8	Gill tissue damaged.	Herbert and Merkens (1961)
Trout	A	525	588	10	No mortality (other end points not investigated).	Griffin (1938)
Trout	A	300	720	12	Decrease in population size.	Peters (1967)
Trout (brook)	A	5	168	3	Fish more active and less dependent on cover.	Gradall and Swenson (1982)
Trout (brown)	A	1,040	17,520	8	Gill lamellae thickened (VFSS).	Herbert et al. (1961)
Trout (brown)	A	1,210	17,520	8	Some gill lamellae become fused (VFSS).	Herbert et al. (1961)
Trout (brown)	A	18	720	10	Abundance reduced.	Peters (1967)
Trout (brown)	A	100	720	11	Population reduced.	Scullion and Edwards (1980)
Trout (brown)	A	1,040	8,760	14	Population one-seventh of expected size (River Fal).	Herbert et al. (1961)
Trout (brown)	A	5,838	8,760	14	Fish numbers one-seventh of expected (River Par).	Herbert et al. (1961)
Trout (cutthroat)	A	35	2	4	Feeding ceased: fish sought cover.	Cordone and Kelly (1961)
Trout (lake)	A	4	168	3	Fish avoided turbid areas.	Swenson (1978)
Trout (rainbow)	A	66	1	3	Avoidance behavior manifested part of the time.	Lawrence and Scherer (1974)
Trout (rainbow)	A	665	1	3	Fish attracted to turbidity.	Lawrence and Scherer (1974)
Trout (rainbow)	A	100	0.10	3	Fish avoid turbid water avoidance behaviour).	Suchanek et al. (1984a; 1984b)
Trout (rainbow)	A	100	0.25	5	Rate of coughing increased (FSS).	Hughes (1975)
Trout (rainbow)	A	250	0.25	5	Rate of coughing increased (FSS).	Hughes (1975)
Trout (rainbow)	A	810	504	8	Gills of fish that survived had thickened epithelium.	Herbert and Merkens (1961)
Trout (rainbow)	A	17,500	168	8	Fish survived; gill epithelium proliferated and thickened.	Slanina (1962)
Trout (rainbow)	A	50	960	9	Rate of weight gain reduced (CWS).	Herbert and Richards (1963)
Trout (rainbow)	A	50	960	9	Rate of weight gain reduced.	Herbert and Richards (1963)
Trout (rainbow)	A	810	504	10	Some fish died.	Herbert and Merkens (1961)
Trout (rainbow)	A	270	3,240	10	Survival rate reduced.	Herbert and Merkens (1961)
Trout (rainbow)	A	200	24	10	Test fish began to die on the first day (WF).	Herbert and Richards (1963)
Trout (rainbow)	A	80,000	24	10	No mortality.	D.W.M. Herbert \e/
Trout (rainbow)	A	18	720	10	Abundance reduced.	Peters (1967)
Trout (rainbow)	A	59	2,232	10	Habitat damage; reduced porosity of gravel.	Slaney et al. (1977b)
Trout (rainbow)	A	4,250	588	12	Mortality rate 50% (CS).	Herbert and Wakeford (1962)
Trout (rainbow)	A	49,838	96	12	Mortality rate 50%.	Lawrence and Scherer (1974)
Trout (rainbow)	A	3,500	1,488	13	Catastrophic reduction in population size.	Herbert and Merkens (1961)
Trout (rainbow)	A	160,000	24	14	Mortality rate 100%.	D.W.M Herbert \e/

		Sediment dose		Fish response		
Species	**Life stage**[†]	**Exposure concentration (mg/L)**	**Exposure duration (hours)**	**SEV**[††]	**Description**	**References**
Trout (sea)	A	210	24	10	Fish abandoned traditional spawning habitat.	Hamilton (1961J
Whitefish (lake)	A	1	1	3	Swimming behavior changed.	Lawrence and Scherer (1974)
Whitefish (lake)	A	16,613	96	12	Mortality rate 50% (DM).	Lawrence and Scherer (1974)
Whitefish (mountain)	A	10,000	24	10	Fish died; silt-clogged gills.	Langer (1980)
Juvenile salmonids (freshwater, groups 1 and 3)						
Grayling (Arctic)	U	20	24	3	Fish avoided parts of the stream.	Birtwell et al. (1984)
Grayling (Arctic)	U	10,000	96	3	Fish swam near the surface.	McLeay et al. (1987)
Grayling (Arctic)	J	86	0.42	3	78% of fish avoided turbid water (NTU >20).	Scannell (1989)
Grayling (Arctic)	U	100	1	4	Catch rate reduced (unfamiliar prey: Drosophila).	McLeay et al (1987)
Grayling (Arctic)	U	100	1	4	Catch rate reduced (unfamiliar prey species: Tubificids).	McLeay et al (1987)
Grayling (Arctic)	U	300	1	4	Catch rate reduced (unfamiliar prey species: Drosophila).	McLeay et al. (1987)
Grayling (Arctic)	U	1,000	1	4	Feeding rate reduced (unfamiliar prey: Tubificids).	McLeay et al. (1987)
Grayling (Arctic)	U	1,000	1	4	Feeding rate reduced (unfamiliar prey: Drosophila).	McLeay et al. (1987)
Grayling (Arctic)	YY	3,810	144	4	Food intake severely limited.	Simmons (1982)
Grayling (Arctic)	U	100	12	6	Reduced ability to tolerate high temperatures.	McLeay et al. (1987)
Grayling (Arctic)	U	100	756	7	Fish moved out of the test channel.	McLeay et al. (1987)
Grayling (Arctic)	U	1,000	1,008	8	Fish had frequent mis-strikes while feeding.	McLeay et al. (1987)
Grayling (Arctic)	U	1,000	1,008	8	Fish responded very slowly to prey.	McLeay et al. (1987)
Grayling (Arctic)	U	300	1,008	8	Rate of feeding reduced.	McLeay et al. (1987)
Grayling (Arctic)	U	1,000	840	8	Rate of feeding reduced.	McLeay et al. (1987)
Grayling (Arctic)	U	1,000	1,008	8	Fish failed to consume all prey.	McLeay et al. (1987)
Grayling (Arctic)	U	300	840	8	Serious impairment of feeding behavior.	McLeay et al. (1987)
Grayling (Arctic)	U	300	1,008	8	Respiration rate increased (FSS).	McLeay et al. (1987)
Grayling (Arctic)	U	300	1,008	8	Fish less tolerant of pentachlorophenol.	McLeay et al. (1987)
Grayling (Arctic)	YY	3,810	144	8	Mucus and sediment accumulated in the gill lamellae.	Simmons (1982)
Grayling (Arctic)	YY	3,810	144	8	Fish display. many signs of poor condition.	Simmons (1982)
Grayling (Arctic)	YY	1,250	48	8	Moderate damage to gill tissue.	Simmons (1982)
Grayling (Arctic)	YY	1,388	96	8	Hyperplasia and hypertrophy of gill tissue.	Simmons (1982)
Grayling (Arctic)	U	100	1,008	9	Growth rate reduced.	McLeay et al. (1984)
Grayling (Arctic)	U	100	840	9	Fish responded less rapidly to drifting food.	McLeay et al. (1987)
Grayling (Arctic)	U	300	1,008	9	Weight gain reduced.	McLeay et al. (1987)
Grayling (Arctic)	U	1,000	1,008	9	Weight gained reduced by 33%.	McLeay et al. (1987)

		Sediment dose		Fish response		
Species	Life stage[†]	Exposure concentration (mg/L)	Exposure duration (hours)	SEV[††]	Description	References
Grayling (Arctic)	U	300	756	10	Fish displaced from their habitat.	McLeay et al. (1987)
Grayling (Arctic)	U	100,000	168	5	No changes in gill histology (not an end point).	McLeay et al. (1983)
Salmon (chinook)	S	943	72	8	Tolerance to stress reduced (VA).	Stober et al. (1981)
Salmon (chinook)	J	6	1,440	9	Growth rate reduced (LNFH).	MacKinlay et al. (1987)
Salmon (chinook)	J	1,400	36	12	Mortality rate 50%.	Newcomb and Flagg (1983)
Salmon (chinook)	J	9,400	36	12	Mortality rate 50%.	Newcomb and Flagg (1983)
Salmon (chinook)	S	488	96	12	Mortality rate 50%.	Stober et al. (1981
Salmon (chinook)	S	11,000	96	12	Mortality rate 50%.	Stober et al. (1981)
Salmon (chinook)	S	19,364	96	12	Mortality rate 50%.	Stober et al. (1981)
Salmon (chinook)	J	39,400	36	14	Mortality rate 90% (VA).	Newcomb and Flagg (1983)
Salmon (chum)	J	28,000	96	12	Mortality rate 50%.	Smith (1940)
Salmon (chum)	J	55,000	96	12	Mortality rate 50% (winter).	Smith (1940)
Salmon (coho)	J	54	0.02	1	Alarm reaction.	Berg (1983)
Salmon (coho)	J	88	0.02	1	Alarm reaction.	Bisson and Bilby (1982)
Salmon (coho)	U	20	0.05	1	Cough frequency not increased.	Servizi and Martens (1992)
Salmon (coho)	J	54	12	3	Changes in territorial behavior.	Berg and Northcote (1985)
Salmon (coho)	J	88	0	3	Avoidance behavior.	Bisson and Bilby (1982)
Salmon (coho)	J	6,000	1	3	Avoidance behavior.	Noggle (1978)
Salmon (coho)	U	300	0.17	3	Avoidance behavior within minutes.	Servizi and Martens (1992)
Salmon (coho)	J	25	1	4	Feeding rate decreased.	Noggle (1978)
Salmon (coho)	J	100	1	4	Feeding rate decreased to 55% of maximum.	Noggle (1978)
Salmon (coho)	J	250	1	4	Feeding rate decreased to 10% of maximum.	Noggle (1978)
Salmon (coho)	J	300	1	4	Feeding ceased.	Noggle (1978)
Salmon (coho)	U	2,460	0.05	5	Coughing behavior manifest within minutes.	Servizi and Martens (1992)
Salmon (coho)	J	54	12	6	Increased physiological stress.	Berg and Northcote (1985)
Salmon (coho)	U	2,460	1	6	Cough frequency greatly increased.	Servizi and Martens (1992)
Salmon (coho)	U	240	24	6	Cough frequency increased more than 5-fold.	Servizi and Martens (1992)
Salmon (coho)	U	530	96	6	Blood glucose levels increased.	Servizi and Martens (1992)
Salmon (coho)	J	1,547	96	8	Gill damage.	Noggle (1978)
Salmon (coho)	U	2,460	24	8	Fatigue of the cough reflex.	Servizi and Martens (1992)
Salmon (coho)	U	3,000	48	8	High level sublethal stress: avoidance.	Servizi and Martens (1992)
Salmon (coho)	J	102	336	9	Growth rate reduced. (FC, BC).	Sigler and others (1984)

		Sediment dose		Fish response		
Species	Life stage†	Exposure concentration (mg/L)	Exposure duration (hours)	SEV††	Description	References
Salmon (coho)	U	8,000	96	10	Mortality rate 1%.	Servizi and Martens (1991)
Salmon (coho)	J	1,200	96	12	Mortality rate 50%.	Noggle (1978)
Salmon (coho)	J	35,000	96	12	Mortality rate 50%.	Noggle (1978)
Salmon (coho)	U	22,700	96	12	Mortality rate 50%.	Servizi and Martens (1991)
Salmon (coho)	F*	8,100	96	12	Mortality rate 50%.	Servizi and Martens (1991)
Salmon (coho)	PS	18,672	96	12	Mortality rate 50%.	Stober et al. (1981)
Salmon (coho)	S	509	96	12	Mortality rate 50%.	Stober et al. 1981)
Salmon (coho)	S	1,217	96	12	Mortality rate 50% (VA).	Stober et al. (1981)
Salmon (coho)	S	28,184	96	12	Mortality rate 50% (VA).	Stober et al. (1981)
Salmon (coho)	S	29,580	96	12	Mortality rate 50%.	Stober et al. (1981)
Salmon (sockeye)	S	1,261	96	8	Body moisture content reduced.	Servizi and Martens (1987)
Salmon (sockeye)	S	7,447	96	8	Plasma chloride levels increased slightly.	Servizi and Martens (1987)
Salmon (sockeye)	U	1,465	96	8	Hypertrophy and necrosis of gill tissue (CSS).	Servizi and Martens (1987)
Salmon (sockeye)	U	3,143	96	8	Hypertrophy and necrosis of gill tissue (FSS).	Servizi and Martens (1987)
Salmon (sockeye)	U	9,851	96	8	Hypertrophy and necrosis of tissue (MCSS).	Servizi and Martens (1987)
Salmon (sockeye)	U	17,560	96	8	Hypertrophy of gill tissue (FSS).	Servizi and Martens (1987)
Salmon (sockeye)	U	23,790	96	8	Hypertrophy and necrosis of gill tissue (FSS).	Servizi and Martens (1987)
Salmon (sockeye)	U	2,688	96	8	Hypertrophy and necrosis of gill tissue (MCSS).	Servizi and Martens (1987)
Salmon (sockeye)	U	2,100	96	10	No fish died (MFSS).	Servizi and Martens (1987)
Salmon (sockeye)	U	9,000	96	10	No mortality.	Servizi and Martens (1987)
Salmon (sockeye)	U	13,900	96	10	Mortality rate 10% (FSS).	Servizi and Martens (1987)
Salmon sockeye	U	9,850	96	10	Gill hyperplasia. hypertrophy, separation, necrosis (MFSS)	Servizi and Martens (1987)
Salmon sockeye)	J	1,400	36	12	Mortality rate 50%.	Newcomb and Flagg (1983)
Salmon (sockeye)	J	9,400	36	12	Mortality rate 50%.	Newcomb and Flagg (1983)
Salmon (sockeye)	U	1,700	96	12	Mortality rate 50% (CSS).	Servizi and Martens (1987)
Salmon (sockeye)	U	4,850	96	12	Mortality rate 50% (MCSS).	Servizi and Martens (1987)
Salmon (sockeye)	U	8,200	96	12	Mortality rate 50% (MFSS).	Servizi and Martens (1987)
Salmon (sockeye)	U	17,560	96	12	Mortality rate was 50% (FSS).	Servizi and Martens (1987)
Salmon (sockeye)	J	39,400	36	14	Mortality rate 90% (VA).	Newcomb and Flagg (1983)
Salmon (sockeye)	U	13,000	96	14	Mortality rate 90% (MFSS).	Servizi and Martens (1987)
Salmon (sockeye)	U	23,900	96	14	Mortality rate 90% (FSS).	Servizi and Martens (1987)
Steelhead	J	102	336	9	Growth rate reduced (FC, BC).	Sigler et al. (1984)

		Sediment dose		Fish response		
Species	Life stage†	Exposure concentration (mg/L)	Exposure duration (hours)	SEV††	Description	References
Trout (brook)	FF	12	5,880	9	Growth rates decline.	Sykora et al. (1972)
Trout (brook)	FF	24	5,208	9	Growth rate reduced (LNFH).	Sykora et al. (1972)
Trout (brook)	FF*	100	1,176	9	Test fish weighed 16% of controls (LNFH).	Sykora et al. (1972
Trout (brook)	FF	50	1,848	9	Growth rates declined (LNFH).	Sykora et al. (1972)
Trout (rainbow)	FF	1,750	480	12	Mortality rate 57% (controls 5%).	Campbell (1954)
Trout (rainbow)	J	4,887	384	8	Hyperplasia of gill tissue.	Goldes (1983)
Trout (rainbow)	J	4,887	384	8	Parasitic infection of gill tissue.	Goldes (1983)
Trout (rainbow)	J	171	96	8	Particles penetrated cells of branchial epithelium.	Goldes (1983)
Trout (rainbow)	Y	90	456	10	Mortality rate 0 to 20% (DE).	Herbert and Merkens (1961)
Trout (rainbow)	Y	90	456	10	Mortality rate 0 to 15% (KC).	Herbert and Merkens (1961)
Trout (rainbow)	Y	270	456	11	Mortality rate 10% to 35% (KC).	Herbert and Merkens (1961)
Trout (rainbow)	Y	810	456	12	Mortality rate 35% to 85% (DE).	Herbert and Merkens (1961)
Trout (rainbow)	Y	810	456	12	Mortality ranged from 5% to 80% (KC).	Herbert and Merkens (1961)
Trout (rainbow)	Y	270	456	12	Mortality rate 25% to 80% (DE).	Herbert and Merkens (1961)
Trout (rainbow)	Y	7,433	672	11	Mortality rate was 40% (CS).	Herbert and Wakeford (1962)
Trout (rainbow)	Y	4,250	672	12	Mortality rate 50%.	Herbert and Wakeford (1962)
Trout (rainbow)	Y	2,120	672	14	Mortality rate 100%.	Herbert and Wakeford (1962)
Trout (rainbow)	J	4,315	57	14	Mortality rate ~100% (CSS).	Newcombe et al. (1995)
Salmonid eggs and larvae (freshwater, group 4)						
Grayling (Arctic)	SF	25	24	10	Mortality rate 5.7%.	J. LaPerriere (pers. comm.)
Grayling (Arctic)	SF	23	48	10	Mortality rate 14.0%.	J. LaPerriere (pers. comm.)
Grayling (Arctic)	SF	65	24	10	Mortality rate 15.0%.	J. LaPerriere (pers. comm.)
Grayling (Arctic)	SF	22	72	10	Mortality rate 14.7%.	J. LaPerriere (pers. comm.)
Grayling (Arctic)	SF	20	96	10	Mortality rate 13.4%.	J. LaPerriere (pers. comm.)
Grayling (Arctic)	SF	143	48	11	Mortality rate 26%.	J. LaPerriere (pers. comm.)
Grayling (Arctic)	SF	185	72	12	Mortality rate 41.3%.	J. LaPerriere (pers. comm.)
Grayling (Arctic)	SF	230	96	12	Mortality rate of 47%.	J. LaPerriere (pers. comm.)
Salmon	E	117	960	10	Mortality; deterioration of spawning gravel.	Cederholm et al. (1981)
Salmon (chum)	E	97	2,808	13	Mortality rate 77% (controls, 6%).	Langer (1980)
Salmon (coho)	E	157	1,728	14	Mortality rate 100% (controls, 16.2%).	Shaw and Maga 1943
Steelhead	E	37	1,488	12	Hatching success 42% (controls, 63%).	Slaney et al. (1977b)
Trout	E	117	960	10	Mortality; deterioration of spawning gravel.	Cederholm el al. (1981)

		Sediment dose		Fish response		
Species	Life stage†	Exposure concentration (mg/L)	Exposure duration (hours)	SEV††	Description	References
Trout (rainbow)	EE	1,750	144	10	Mortality rate greater than controls (controls, 6%).	Campbell (1954)
Trout (rainbow)	E	7	1,152	11	Mortality rate 40%.	Slaney et al. (1977b)
Trout (rainbow)	E	57	1,488	12	Mortality rate 47% (controls, 32%).	Slaney et al. (1977b)
Trout (rainbow)	E	120	384	13	Mortality rate ~60% to 70% (controls, 38.6%).	Erman and Lignon 1988
Trout rainbow)	E	21	1,152	13	Mortality rate 72%.	Slaney et al. (1977a)
Trout rainbow)	E	47	1,152	14	Mortality rate 100%.	Slaney et al. (1977b)
Trout (rainbow)	E	101	1,440	14	Mortality rate 98% (controls, 14.6%).	Turnpenny and Williams (1980)
Non-salmonid eggs and larvae (estuarine, group 4)						
Bass (striped)	L	200	0.42	4	Feeding rate reduced 40%.	Breitburg (1988)
Bass (striped)	E	800	24	9	Development rate slowed significantly.	Morgan et al. (1983)
Bass (striped)	E	100	24	9	Hatching delayed.	Schubel and Wang (1973)
Bass (striped)	E	1,000	168	10	Reduced hatching success.	Auld and Schubel (1978)
Bass (striped)	L	1,000	68	11	Mortality rate 35% (controls, 16%).	Auld and Schubel (1978)
Bass (striped)	L	500	72	12	Mortality rate 42% (controls, 17%).	Auld and Schubel &1978)
Bass (striped)	L	485	24	12	Mortality rate 50%.	Morgan et al. (1973)
Herring	L	10	3	3	Depth preference changed.	Johnson and Wildish (1982)
Herring (lake)	L	16	24	3	Depth preference changed.	Swenson and Matson (1976)
Herring (Pacific)	L	2,000	2	4	Feeding rate reduced.	Boehlert and Morgan (1985)
Herring (Pacific)	L	1,000	24	8	Mechanical damage to epidermis.	Boehlert (1984)
Herring (Pacific)	L	4,000	24	8	Epidermis punctured; micro ridges less distinct.	Boehlert 1984)
Perch (while)	E	800	24	9	Egg development slowed significantly.	Morgan et al. (1983)
Perch (white)	E	100	24	9	Hatching delayed.	Schubel and Wang (1973)
Perch (white)	E	1,000	168	10	Reduced hatching success.	Auld and Schubel (1978)
Perch (white)	L	155	48	12	Mortality rate 50%.	Morgan et al. (1973)
Perch (white)	L	373	24	12	Mortality rate 50%.	Morgan et al. (1973)
Perch (white)	L	280	48	12	Mortality rate 50%.	Morgan el al. (1973)
Perch (yellow)	L	500	96	11	Mortality rate 37% (controls, 7%).	Aul and Schubel (1978)
Perch (yellow)	L	1,000	96	11	Mortality rate 38% (controls, 7%).	Auld and Schubel (1978)
Shad (American)	L	100	96	10	Mortality rate 18% (controls, 5%).	Auld and Schubel (1978)
Shad (American)	L	500	96	11	Mortality rate was 36% (controls, 4%).	Auld and Schubel (1978)
Shad (American)	L	1,000	96	11	Mortality rate was 34% (controls, 5%).	Auld and Schubel (1978)
Adult non-salmonids (estuarine, or riverine-estuarine, group 5)						

		Sediment dose		Fish response		
Species	Life stage†	Exposure concentration (mg/L)	Exposure duration (hours)	SEV††	Description	References
Anchovy (bay)	A	231	24	10	Mortality rate 10% (FE).	Sherk et al. (1975)
Anchovy (bay)	A	471	24	12	Mortality rate 50% (FE).	Sherk et al. (1975)
Anchovy (bay)	A	960	24	14	Mortality rate 90%.	Sherk et al. (1975)
Bass (striped)	A	1,500	336	8	Haematocrit increased (FE).	Sherk et al. (1975)
Bass (striped)	A	1,500	336	8	Plasma osmolality increased (FE).	Sherk et al. (1975)
Cunner	A	28,000	24	12	Mortality rate 50% (20.0-25.0°C).	Rogers (1969)
Cunner	A	133,000	12	12	Mortality rate was 50% (15°C).	Rogers (1969)
Cunner	A	100,000	24	12	Mortality rate 50% (15°C).	Rogers (1969)
Cunner	A	72,000	48	12	Mortality rate 50% (15°C).	Rogers (1969)
Fish	A	3,000	240	10	Fish died.	Kemp (1949)
Herring (Atlantic)	A	20	3	4	Reduced feeding rate.	Johnson and Wildish (1982)
Hogchoker	A	1,240	24	8	Energy utilization increased.	Sherk el al. p975)
Hogchoker	A	1,240	120	8	Erythrocyte count increased.	Sherk et al. 1975)
Hogchoker	A	1,240	120	8	Haematocrit increased.	Sherk et al. (1975)
Killifish (striped)	A	960	120	8	Haematocrit increased.	Sherk et al. (1975)
Killifish (striped)	A	3,277	24	10	Mortality rate 10% (FE).	Sherk et al. (1975)
Killifish (striped)	A	9,720	24	10	Mortality rate 10%.	Sherk et al. (1975)
Killifish (striped)	A	3,819	24	12	Mortality rate 50%.	Sherk et al. (1975)
Killifish (striped)	A	12,820	24	12	Mortality rate 50%.	Sherk et al. (1975)
Killifish (striped)	A	16,930	24	13	Mortality rate 90%.	Sherk et al. (1975)
Killifish (striped)	A	6,136	24	14	Mortality rate 90%.	Sherk et al. (1975)
Menhaden (Atlantic)	A	154	24	10	Mortality rate 10% (FE).	Sherk et al. (1975)
Menhaden (Atlantic)	A	247	24	12	Mortality rate 50% (FE).	Sherk et al. 1975)
Menhaden (Atlantic)	A	396	24	14	Mortality rate 90% (FE).	Sherk et al. 1975)
Minnow (sheepshead)	A	200,000	24	10	Mortality rate 10% (15°C).	Rogers (1969)
Minnow (sheepshead)	A	300,000	24	11	Mortality rate 30% (10°C).	Rogers (1969)
Minnow (sheepshead)	A	100,000	24	14	Mortality rate 90% (19°C).	Rogers (1969)
Mummichog	A	300,000	24	10	No mortality (15°C).	Rogers (1969)
Mummichog	A	2,447	24	10	Mortality rate 10% (FE).	Sherk et al. (1975)
Mummichog	A	3,900	24	12	Mortality rate 50% (FE).	Sherk et al. (1975)
Mummichog	A	6,217	24	14	Mortality rate 90%.	Sherk et al. (1975)
Perch (white)	A	650	120	6	Haematocrit increased.	Sherk et al. (1975)

		Sediment dose		Fish response		
Species	Life stage†	Exposure concentration (mg/L)	Exposure duration (hours)	SEV††	Description	References
Perch (white)	A	650	120	6	Erythrocyte count increased.	Sherk et al. (1975)
Perch (white)	A	650	120	6	Haemoglobin concentration increased.	Sherk et al. (1975)
Perch (white)	A	305	120	8	Gill tissue may have been damaged.	Sherk et al. (1975)
Perch (white)	A	650	120	8	Histological damage to gill tissue.	Sherk et al. (1975)
Perch (white)	A	305	24	10	Mortality rate 10% (FE).	Sherk et al. (1975)
Perch (white)	A	985	24	12	Mortality rate 50%.	Sherk et al. (1975)
Perch (white)	A	3,181	24	14	Mortality rate 90% (FE).	Sherk et al. (1975)
Rasbora	A	40,000	24	10	Fish died (BC).	Alabaster and Lloyd (1980)
Rasbora	A	6,000	168	10	No mortality.	Alabaster and Lloyd (1980)
Shad (American)	A	150	0.25	3	Change in preferred swimming depth.	Dadswell et al. (1 83)
Silverside (Atlantic)	A	58	24	10	Mortality rate 10% (FE).	Sherk et al. (1975)
Silverside (Atlantic)	A	250	24	12	Mortality rate 50% (FE).	Sherk et al. (1975)
Silverside (Atlantic)	A	1,000	24	14	Mortality rate 90% (FE).	Sherk et al. (1975)
Spot	A	114	48	10	Mortality rate 10% (FE).	Sherk et al. (1975)
Spot	A	1,309	24	10	Mortality rate 10% (FE).	Sherk et al. (1975)
Spot	A	6,875	24	10	Mortality rate 10%.	Sherk et al. (1975)
Spot	A	189	48	12	Mortality rate 50% (FE).	Sherk et al. (1975
Spot	A	2,034	24	12	Mortality rate 50%.	Sherk et al. (1975
Spot	A	8,800	24	12	Mortality rate 50%.	Sherk el al. (1975
Spot	A	317	48	14	Mortality rate 90% (FE).	Sherk et al. (1975)
Spot	A	11,263	24	14	Mortality rate 90%.	Sherk et al. (1975)
Stickleback (threespine)	A	28,000	96	10	No morality in test designed to identify lethal threshold.	LeGore and DesVoigne (1973)
Stickleback (fourspine)	A	100	24	10	Mortality rate <1% (IA).	Rogers (1969)
Stickleback (fourspine)	A	10,000	24	10	No mortality (KS; 10-12°C).	Rogers (1969)
Stickleback (fourspine)	A	300	24	12	Mortality rate ~50% (IA).	Rogers (1969)
Stickleback (fourspine)	A	18,000	24	12	Mortality rate 50% (15.0 -16.0°C).	Rogers (1969)
Stickleback (fourspine)	A	50,000	24	12	Mortality rate 50% (KS).	Rogers (1969)
Stickleback (fourspine)	A	53,000	24	12	Mortality rate 50% (10-12°C).	Rogers (1969)
Stickleback (fourspine)	A	330,000	24	12	Mortality rate 50% (9.0-9.5°C).	Rogers (1969)
Stickleback (fourspine)	A	500	24	14	Mortality rate 100%.	Rogers (1969)
Stickleback (fourspine)	A	200,000	24	14	Mortality rate 95% (KS).	Rogers (1969)
Toadfish (oyster)	A	3,360	1	6	Oxygen consumption more variable in prestressed fish.	Neumann et al. (1975)

		Sediment dose		Fish response		
Species	Life stage†	Exposure concentration (mg/L)	Exposure duration (hours)	SEV††	Description	References
Toadfish (oyster)	A	14,600	72	8	Fish largely unaffected but developed latent ill effects.	Neumann et al. (1975)
Toadfish (oyster)	A	11,090	72	9	Latent ill effects manifested in subsequent test at low SS.	Neumann et al. (1975)
Adult non-salmonids (freshwater, group 6)						
Bass (largemouth)	A	63	720	9	Weight gain reduced ~50%.	Buck (1956)
Bass (largemouth)	A	145	720	9	Growth retarded.	Buck (1956)
Bass (largemouth)	A	145	720	12	Fish unable to reproduce.	Buck (1956)
Bluegill	A	423	0.05	4	Rate of feeding reduced.	Gardner (1981)
Bluegill	A	15	1	4	Reduced capacity to locate prey.	Vinyard and O'Brien (1976)
Bluegill	A	145	720	9	Growth retarded.	Buck (1956)
Bluegill	A	63	720	9	Weight gain reduced ~50%.	Buck (1956)
Bluegill	A	145	720	12	Fish unable to reproduce.	Buck (1956)
Carp (common)	A	25,000	336	10	Some mortality.	Wallen (1951)
Darters	A	2,045	8,760	14	Darters absent.	Vaughan (1979) \f/
Fish	A	120	384	10	Density of fish reduced.	Erman and Lignon (1988)
Fish	A	620	48	10	Fish kills downstream from sediment source.	Hesse and Newcomb (1982)
Fish	A	900	720	12	Fish absent or marked! reduced in abundance.	Herbert and Richards (1963)
Fish	A	2,045	8,760	12	Habitat destruction: fish populations smaller than expected	Vaughan (1979) \f/
Fish (warmwater)	A	100,000	252	10	Some fish died; most survived.	Wallen (1951)
Fish (warmwater)	A	200,000	1	10	Fish died: opercular cavities and gill filaments clogged.	Wallen (1951)
Fish (warmwater)	A	22	8,760	12	Fish populations destroyed.	Menzel et al. (1984)
Goldfish	A	25,000	336	10	Some mortality (MC).	Wallen (1951)
Sunfish (green)	A	9,600	1	5	Rate of ventilation increased.	Horkel and Pearson (1976)
Sunfish (redear)	A	63	720	9	Weight gain reduced ~50% compared to controls.	Buck (1956)
Sunfish (redear)	A	145	720	9	Growth retarded.	Buck (1956)
Sunfish (redear)	A	145	720	12	Fish unable to reproduce.	Buck (1956)

†: A = adult; E = egg; EE = eyed egg; F = fry; F* = swim-up fry; FF = young fry (<3 weeks old); FF* = older fry (>30 weeks old); J = juvenile; L = larva; PS = pre-smolt; SF = sac fry; U = underyearling; Y = approximate yearling; YY = young of the year.

††: Severity-of-ill-effect (SEV) score ranging from 0 (no detectable effect) to 14 (maximum effect).

Other abbreviations: BC = bentonite clay; CS = calcium sulphate; CSS = course SS; CWS = coal washery solids; DE = diatomaceous earth; DM = drilling mud; FC = fire clay; FE = fuller's earth; FSS = fine SS; IA = incinerator ash; KC = kaolin clay; KS = Kingston silt; LNFH = lime-neutralized ferric hydroxide; MC = montmorillonite clay; MCSS = medium to course SS; MFSS = medium to fine SS; NTU = nephelometric turbidity unit; SS = suspended sediment; VA = volcanic ash; VFSS = very fine SS.

Appendix 2. Severity of ill effects of suspended sediment on invertebrates and plants of lakes, and streams, including data from some relatively sensitive marine organisms.

Organism		mg SS/L	hr	LN (mg.SS/L*h)	SEV	Description	Reference
Zooplankton	fauna		0.0833	-2.48490665	4	Feeding rate (ingestion + incorporation) slightly reduced.	Arruda and others 1983
Zooplankton	fauna	10	0.0833	-0.182321557	4	Efficiency of incorporation of food reduced.	Arruda and others 1983
Oysters (marine)	fauna	100	0.0167	0.510825624	4	Change in filtering rate.	Loosanoff and Tommers 1948
Oysters (marine)	fauna	100	0.0167	0.510825624	3	Change in pattern of shell movements.	Loosanoff and Tommers 1948
Zooplankton	fauna	24.5	0.0833	0.713766468	4	Feeding rate (ingestion + incorporation) reduced by 10%.	Arruda and others 1983
Zoobenthos	fauna	10	0.25	0.916290732	3	Increased invertebrate drift.	Rosenberg and Snow 1980
Zooplankton	fauna	24	0.1667	1.386294361	4	Reduced efficiency of food assimilation.	McCabe and O'Brien 1983
Zooplankton (Copepoda)	fauna	50	0.0833	1.427116356	5	Ingestion of food drastically reduced.	Sherk and others 1975
Macrobenthos	fauna	23	0.25	1.749199855	3	Increased drift of macrobenthos.	Rosenberg and Snow 1975
Zooplankton	fauna	100	0.0833	2.120263536	5	Efficiency of incorporation of food greatly reduced.	Arruda and others 1983
Zooplankton	fauna	245	0.0833	3.016351561	5	Feeding rate (ingestion + incorporation) reduced by ~90%.	Arruda and others 1983
Invertebrates	fauna	7.6	5	3.63758616	3	Increased drift of macrobenthic invertebrates.	Rosenberg and Wiens 1978
Zooplankton (Copepoda)	fauna	250	0.1667	3.729701449	4	Maximum rate of food ingestion reduced.	Sherk and others 1975
Zooplankton	fauna	2451	0.0833	5.319344734	5	Feeding rate (ingestion + incorporation) reduced by 99.2%.	Arruda and others 1983
Invertebrates (macro)	fauna	72.5	24	7.461640392	8	Silt-intolerant species less abundant; increased drift.	Gammon 1970
Invertebrates (benthic)	fauna	1700	2	8.131530711	8	Temporary changes to community structure.	Fairchild et al 1987
Invertebrates (benthic)	fauna	1700	2	8.131530711	8	Temporary changes in community structure.	Fairchild et al. 1987
Zoobenthos	fauna	5	720	8.188689124	10	Fewer taxa in lakes flooded with turbid water.	Rosenberg and Snow 1977
Zoobenthos	fauna	5	720	8.188689124	10	Reduced standing crop in lakes flooded with turbid water.	Rosenberg and Snow 1977
Algae (benthic)	flora	90	48	8.371010681	10	Mean productivity reduced by approximately half.	Van Nieuwenhuyse 1983
Zoobenthos	fauna	12.5	720	9.104979856	11	Decrease in size of zoobenthic population.	Rosenberg and Snow 1977
Zooplankton	fauna	50	192	9.169518377	8	Zooplankton eat fewer algae and could starve.	Arruda 1983
Insects (benthic)	fauna	75	168	9.441452093	11	Abundance of benthic insects markedly reduced.	Slaney et al. 1977b
Zooplankton (Cladocera)	fauna	273	72	9.886137914	10	Survival reduced.	Robertson 1957 [**]
Zooplankton (Cladocera)	fauna	273	72	9.886137914	10	Reproduction reduced.	Robertson 1957 [**]
Invertebrate	fauna	29	720	9.946547042	12	Habitat conditions unsuitable for Ephemeroptera.	M.P. Vivier, pers comm. [**]
Invertebrate	fauna	29	720	9.946547042	12	Habitat conditions unsuitable for Trichoptera.	M.P. Vivier, pers. comm. [**]
Invertebrate	fauna	29	720	9.946547042	12	Habitat conditions unsuitable for Crustacea.	M.P. Vivier, pers. comm. [**]
Invertebrate	fauna	29	720	9.946547042	12	Habitat conditions unsuitable for Mollusca.	M.P. Vivier, pers. comm. [**]
Zooplankton	fauna	104	220	10.03801845	13	Reproduction severely curtailed.	Edmunsdon and Koenings 1985
Zooplankton	fauna	104	220	10.03801845	13	Mortality rate of zooplankters was 85% (controls	Edmunsdon and Koenings 1985

Organism		mg SS/L	hr	LN (mg.SS/L*h)	SEV	Description	Reference
						55%).	
Invertebrates	fauna	16	1488	10.07777694	10	Standing crop of benthic invertebrates reduced.	Slaney et al. 1977a
Zoobenthos	fauna	17.6	1488	10.17308712	10	Taxonomic diversity in zoobenthic communities reduced.	McCart 1979
Zoobenthos	fauna	17.6	1488	10.17308712	10	Species diversity of zoobenthic communities reduced.	McCart 1979
Zoobenthos	fauna	17.6	1488	10.17308712	10	Species equitability in zoobenthic communities reduced.	McCart 1979
Zooplankton	fauna	150	192	10.26813067	8	Zooplankton eat fewer algae and may starve.	Arruda 1983
Zooplankton (Cladocera)	fauna	400	72	10.26813067	10	Gills and gut clogged with particles of sediment.	Stephan 1953 [**]
Zooplankton (Copepoda)	fauna	400	72	10.26813067	10	Gills and gut clogged with particles of sediment.	Stephan 1953 [**]
Invertebrates	fauna	620	48	10.30092049	14	Catastrophic loss of many invertebrate species.	Hesse and Newcomb 1982
Invertebrates (macro)	fauna	4610	8	10.51542468	12	19.4% reduction in number of taxa	Doeg and Koehn 1994 [***]
Invertebrates (macro)	fauna	4610	8	10.51542468	13	63.9% reduction in abundance of organisms	Doeg and Koehn 1994 [***]
Invertebrates	fauna	120	384	10.7381343	13	Reductions in numbers and taxa of invertebrates.	Erman and Lignon 1988
Invertebrates (benthic)	fauna	32	1488	10.77092412	13	Standing crop reduced.	Slaney et al. 1977a
Aquatic Moss	flora	100	504	10.82774645	13	Leaves of aquatic plants severely abraded.	Lewis 1973
Zoobenthos	fauna	100	672	11.11542853	13	Standing crop reduced.	Rosenberg and Snow 1977
Benthos	fauna	1461	48	11.15807742	10	Alteration of habitat for benthic organisms.	Schubert Vinikour Gartman 1985
Aquatic Moss	flora	500	168	11.33857208	13	Leaves severely damaged by abrasion of coal dust.	Lewis 1973
Invertebrates	fauna	62	2400	11.9103584	13	Density of invertebrate populations reduced 77%.	Wagener and LaPerriere 1985
Invertebrates	fauna	62	2400	11.9103584	13	Invertebrate biomass reduced.	Wagener and LaPerriere 1985
Invertebrates (macro)	fauna	17	8760	11.91116453	14	Many taxa eliminated. (Coal Particles).	Learner and others 1971
Invertebrates	fauna	77	2400	12.12702944	12	Invertebrate populations reduced 50%.	Wagener and LaPerriere 1985
Invertebrates	fauna	77	2400	12.12702944	12	Invertebrate biomass reduced: 1.9 mg per 0.1 m^2 (controls 3.4).	Wagener and LaPerriere 1985
Algae	flora	570	336	12.16274752	12	Algal growth decreased as a function of increasing turbidity.	Wang 1974
Vascular Plants	flora	570	336	12.16274752	12	Plant growth decreased as a function of increasing turbidity.	Wang 1974
Benthic Fauna	fauna	325.5	720	12.36461367	13	Numbers of organisms per unit area reduced 75%.	Tebo 1955
Invertebrates (benthic)	fauna	390	720	12.54539795	12	The density of benthic invertebrates decreased.	Tebo 1955
Invertebrates (macro)	fauna	46.8	8760	12.92383439	13	Species diversity reduced; several species absent. (Sand).	Nuttall 1972
Invertebrates	fauna	278	2400	13.41084513	13	Invertebrate populations reduced by 80%.	Wagener and LaPerriere 1985
Invertebrates	fauna	278	2400	13.41084513	13	Invertebrate biomass reduced.	Wagener and LaPerriere 1985
Oysters (marine)	fauna	3000	240	13.48700649	13	Severe damage lo oyster beds.	Kemp 1949 [****]

Organism		mg SS/L	hr	LN (mg.SS/L*h)	SEV	Description	Reference
Invertebrates (stream)	fauna	130	8760	13.94548563	13	Species diversity reduced.	Nuttall and Bielby 1973
Invertebrates	fauna	743	2400	14.39392006	13	Invertebrate populations reduced 85%.	Wagener and LaPerriere 1985
Invertebrates	fauna	743	2400	14.39392006	10	Invertebrate biomass reduced.	Wagener and LaPerriere 1985
Algae	flora	570	8760	15.42358754	12	Reduced distribution.	Wang 1974
Vascular Plants	flora	570	8760	15.42358754	12	Reduced distribution.	Wang 1974
Invertebrates	fauna	5108	2400	16.32178723	14	Invertebrate populations reduced: 94% of expected.	Wagener and LaPerriere 1985
Invertebrates	fauna	5108	2400	16.32178723	12	Invertebrate biomass reduced: 56% of control site.	Wagener and LaPerriere 1985
Diatoms	flora	2045	8760	16.70110425	12	Size of diatom populations reduced.	Vaughan 1979; Vaughan et al. 1982
Invertebrates	fauna	25000	8760	19.20458229	14	Macrofauna absent.	Nuttall and Bielby 1973

Appendix 3. Summary of the available information on biotic responses to streambed substrate composition under laboratory conditions.

Species	Study Type	PF (<1.00 mm)	PF (<2.00 mm)	PF (<3.00 mm)	PF (<6.35 mm)	Dg (mm)	Dg/De	Fredle Number	Percent Survival to Emergence	Reference
Brown trout	L					1.5	0.33		33	Olsson & Persson 1986
Brown trout	L					4.8	1.07		53	Olsson & Persson 1986
Brown trout	L					9.6	2.13		77	Olsson & Persson 1986
Brown trout	L					18	4		95	Olsson & Persson 1986
Brown trout	L					32	7.11		87	Olsson & Persson 1986
Brown trout	L	0							90	Olsson & Persson 1988
Brown trout	L	5							93	Olsson & Persson 1988
Brown trout	L	10							88	Olsson & Persson 1988
Brown trout	L	20							28	Olsson & Persson 1988
Brown trout	L	40							4	Olsson & Persson 1988
Bull trout	L	0[5]			0				38	Weaver & White 1985
Bull trout	L	5.9[5]			9.9				48	Weaver & White 1985
Bull trout	L	11.7[5]			19.8				38	Weaver & White 1985
Bull trout	L	17.6[5]			29.6				21	Weaver & White 1985
Bull trout	L	23.4[5]			39.5				1	Weaver & White 1985
Bull trout	L	29.3[5]			49.4				0	Weaver & White 1985
Bull trout	L	0			0				90	Weaver & White 1985
Bull trout	L	4			10				50	Weaver & White 1985
Bull trout	L	8			20				28	Weaver & White 1985
Bull trout	L	12			30				8	Weaver & White 1985
Bull trout	L	16			40				1	Weaver & White 1985
Bull trout	L	20			50				1	Weaver & White 1985
Chinook salmon	L				2				58	Bjornn 1968
Chinook salmon	L				10				65	Bjornn 1968
Chinook salmon	L				20				64	Bjornn 1968
Chinook salmon	L		30						61	Bjornn 1968
Chinook salmon	L		40						2	Bjornn 1968
Chinook salmon	L		50						0	Bjornn 1968

Species	Study Type	PF (<1.00 mm)	PF (<2.00 mm)	PF (<3.00 mm)	PF (<6.35 mm)	Dg (mm)	Dg/De	Fredle Number	Percent Survival to Emergence	Reference
Chinook salmon	L		60						0	Bjornn 1968
Chinook salmon	L		70						0	Bjornn 1968
Chinook salmon	L		0[5]		0	21.5	3.31	17.6	96	Tappel 1981
Chinook salmon	L		1.8[5]		9	19.1	2.94	14.8	95	Tappel 1981
Chinook salmon	L		3.5[5]		18	13.9	2.14	9.6	92	Tappel 1981
Chinook salmon	L		5.2[5]		26.9	10.7	1.65	6	88	Tappel 1981
Chinook salmon	L		5.7[5]		14.1	16.4	2.52	12.5	95	Tappel 1981
Chinook salmon	L		5.9[5]		9.9	19.1	2.94	14.8	99	Tappel 1981
Chinook salmon	L		7[5]		35.9	9.1	1.4	4.5	87	Tappel 1981
Chinook salmon	L		9.5[5]		23.4	10.4	1.60	6.5	93	Tappel 1981
Chinook salmon	L		11.7[5]		19.8	11.8	1.82	8.7	97	Tappel 1981
Chinook salmon	L		13.3[5]		32.8	7.6	1.17	3.5	77	Tappel 1981
Chinook salmon	L		11.1[5]		42.2	6.1	0.94	2.4	61	Tappel 1981
Chinook salmon	L		17.6[5]		29.6	6.6	1.02	3	88	Tappel 1981
Chinook salmon	L		20.9[5]		51.6	5	0.77	1.7	18	Tappel 1981
Chinook salmon	L		23.4[5]		39.5	4.7	0.72	1.6	32	Tappel 1981
Chinook salmon	L		29.3[5]		49.4	4	0.62	1.1	6	Tappel 1981
Coho salmon	L					5	0.95		11	Cederholm Unpublished
Coho salmon	L					6	1.14		19	Cederholm Unpublished
Coho salmon	L					7	1.33		35	Cederholm Unpublished
Coho salmon	L					7.5	1.43		45	Cederholm Unpublished
Coho salmon	L					8	1.52		39	Cederholm Unpublished
Coho salmon	L					10.5	2		47	Cederholm Unpublished
Coho salmon	L					14	2.67		67	Cederholm Unpublished
Coho salmon	L					14.5	2.76		70	Cederholm Unpublished
Coho salmon	L					17	3.24		81	Cederholm Unpublished
Coho salmon	L		2[3]						95	Hall and Lantz 1969
Coho salmon	L		11[3]						83	Hall and Lantz 1969
Coho salmon	L		19[3]						65	Hall and Lantz 1969
Coho salmon	L		29[3]						35	Hall and Lantz 1969

Species	Study Type	PF (<1.00 mm)	PF (<2.00 mm)	PF (<3.00 mm)	PF (<6.35 mm)	Dg (mm)	Dg/De	Fredle Number	Percent Survival to Emergence	Reference
Coho salmon	L		39[3]						21	Hall and Lantz 1969
Coho salmon	L		43[3]						22	Hall and Lantz 1969
Coho salmon	L		58[3]						9	Hall and Lantz 1969
Coho salmon	L		68[3]						10	Hall and Lantz 1969
Coho salmon	L					4.5	0.86		8	Phillips et al.1975*
Coho salmon	L					7	1.33		20	Phillips et al. 1975*
Coho salmon	L					11	2.10		36	Phillips et al. 1975*
Coho salmon	L					14	2.67		63	Phillips et al. 1975*
Coho salmon	L					18	3.43		85	Phillips et al. 1975*
Coho salmon	L					22.5	4.29		95	Phillips et al. 1975*
Coho salmon	L		0						97	Phillips et al. 1975
Coho salmon	L		10						85	Phillips et al. 1975
Coho salmon	L		20						65	Phillips et al. 1975
Coho salmon	L		30						35	Phillips et al. 1975
Coho salmon	L		40						20	Phillips et al. 1975
Coho salmon	L		50						20	Phillips et al. 1975
Coho salmon	L		60						8	Phillips et al. 1975
Coho salmon	L		70						10	Phillips et al. 1975
Cutthroat trout	L	0[4]	0[5]		0				95	Irving and Bjornn 1984
Cutthroat trout	L	0.7[4]	1.8[5]		9				91.2	Irving and Bjornn 1984
Cutthroat trout	L	1.4[4]	3.5[5]		18				74.5	Irving and Bjornn 1984
Cutthroat trout	L	2[4]	5.2[5]		26.9				56.7	Irving and Bjornn 1984
Cutthroat trout	L	2.7[4]	7[5]		35.9				36.7	Irving and Bjornn 1984
Cutthroat trout	L	3.5[4]	5.7[5]		14.1				27.7	Irving and Bjornn 1984
Cutthroat trout	L	3.9[4]	5.9[5]		9.9				73.7	Irving and Bjornn 1984
Cutthroat trout	L	5.8[4]	9.5[5]		23.4				46.1	Irving and Bjornn 1984
Cutthroat trout	L	7.8[4]	11.7[5]		19.8				18	Irving and Bjornn 1984
Cutthroat trout	L	8.2[4]	13.3[5]		32.8				14.4	Irving and Bjornn 1984
Cutthroat trout	L	1.05[4]	17.1[5]		42.2				2.5	Irving and Bjornn 1984
Cutthroat trout	L	11.7'4)	17.6[5]		29.6				0.7	Irving and Bjornn 1984

Species	Study Type	PF (<1.00 mm)	PF (<2.00 mm)	PF (<3.00 mm)	PF (<6.35 mm)	Dg (mm)	Dg/De	Fredle Number	Percent Survival to Emergence	Reference
Cutthroat trout	L	12.9[4]	20.9[5]		51.6				1.8	Irving and Bjornn 1984
Cutthroat trout	L	15.6[4]	23.4[5]		39.5				0.4	Irving and Bjornn 1984
Cutthroat trout	L	19.5[4]	29.3[5]		49.4				2.5	Irving and Bjornn 1984
Cutthroat trout	L		0		0				76	Weaver & Fraley 1993
Cutthroat trout	L		3		10				55	Weaver & Fraley 1993
Cutthroat trout	L		7		20				39	Weaver & Fraley 1993
Cutthroat trout	L		12		30				34	Weaver & Fraley 1993
Cutthroat trout	L		18		40				26	Weaver & Fraley 1993
Cutthroat trout	L		25		50				4	Weaver & Fraley 1993
Kokanee salmon	L	0[4]	0[5]		0				95	Irving and Bjornn 1984
Kokanee salmon	L	0.7[4]	1.8[5]		9				93.2	Irving and Bjornn 1984
Kokanee salmon	L	1.4[4]	3.5[5]		18				87.4	Irving and Bjornn 1984
Kokanee salmon	L	2[4]	5.2[5]		26.9				83.3	Irving and Bjornn 1984
Kokanee salmon	L	2.7[4]	7[5]		35.9				87.4	Irving and Bjornn 1984
Kokanee salmon	L	3.5[4]	5.7[5]		14.1				93	Irving and Bjornn 1984
Kokanee salmon	L	3.9[4]	5.9[5]		9.9				99.7	Irving and Bjornn 1984
Kokanee salmon	L	5.8[4]	9.5[5]		23.4				79.8	Irving and Bjornn 1984
Kokanee salmon	L	7.8[4]	11.7[5]		19.8				64.5	Irving and Bjornn 1984
Kokanee salmon	L	8.2[4]	13.3[5]		32.8				50.1	Irving and Bjornn 1984
Kokanee salmon	L	10.5[4]	17.1[5]		42.2				23.5	Irving and Bjornn 1984
Kokanee salmon	L	11.7[4]	17.6[5]		29.6				32.8	Irving and Bjornn 1984
Kokanee salmon	L	12.9[4]	20.9[5]		51.6				13.7	Irving and Bjornn 1984
Kokanee salmon	L	15.6[4]	23.4(5}		39.5				16.1	Irving and Bjornn 1984
Kokanee salmon	L	19.5[4]	29.3[5]		49.4				10.6	Irving and Bjornn 1984
Rainbow trout	L	0[4]	0[5]		0				95	Irving and Bjornn 1984
Rainbow trout	L	0.7[4]	1.8[5]		9				96.3	Irving and Bjornn 1984
Rainbow trout	L	1.4[4]	3.5[5]		18				102	Irving and Bjornn 1984
Rainbow trout	L	2[4]	5.2[5]		26.9				91.2	Irving and Bjornn 1984
Rainbow trout	L	2.7[4]	7[5]		35.9				69.5	Irving and Bjornn 1984
Rainbow trout	L	3.5[4]	5.7[5]		14.I				96.5	Irving and Bjornn 1984
Rainbow trout	L	3.9[4]	5.9[5]		9.9				92.5	Irving and Bjornn 1984

Species	Study Type	PF (<1.00 mm)	PF (<2.00 mm)	PF (<3.00 mm)	PF (<6.35 mm)	Dg (mm)	Dg/De	Fredle Number	Percent Survival to Emergence	Reference
Rainbow trout	L	5.8[(4)]	9.5[(5)]		23.4				64.2	Irving and Bjornn 1984
Rainbow trout	L	7.8[(4)]	11.7[(5)]		19.8				33.2	Irving and Bjornn 1984
Rainbow trout	L	8.2[(4)]	13.3[(5)]		32.8				50.2	Irving and Bjornn 1984
Rainbow trout	L	10.5[(4)]	11.1[(5)]		42.2				13.2	Irving and Bjornn 1984
Rainbow trout	L	11.7[(4)]	17.6[(5)]		29.6				7.2	Irving and Bjornn 1984
Rainbow trout	L	12.9[(4)]	20.95[(5)]		51.6				8.5	Irving and Bjornn 1984
Rainbow trout	L	15.6[(4)]	23.4[(5)]		39.5				3	Irving and Bjornn 1984
Rainbow trout	L	19.5[(4)]	29.3[(5)]		49.4				10.2	Irving and Bjornn 1984
Sockeye salmon	L					2.5	0.53		3	Cooper 1965
Sockeye salmon	L					11	2.32		74	Cooper 1965
Sockeye salmon	L					14.5	3.05		90	Cooper 1965
Steelhead	L		2[(3)]						100	Hall and Lantz 1969
Steelhead	L		11[(3)]						100	Hall and Lantz 1969
Steelhead	L		20[(3)]						70	Hall and Lantz 1969
Steel head	L		29[(3)]						53	Hall and Lantz 1969
Steelhead	L		33[(3)]						50	Hall and Lantz 1969
Steelhead	L		49[(3)]						33	Hall and Lantz 1969
Steelhead	L		53[(3)]						30	Hall and Lantz 1969
Steelhead	L		68[(3)]						20	Hall and Lantz 1969
Steelhead	L				2				90	Bjornn 1968
Steelhead	L				12				98	Bjornn 1968
Steelhead	L				17				82	Bjornn 1968
Steelhead	L				30				81	Bjornn 1968
Steel head	L				36				79	Bjornn 1968
Steelhead	L				47				50	Bjornn 1968
Steelhead	L				55				10	Bjornn 1968
Steelhead	L				62				3	Bjornn 1968
Steelhead	L					4	1		17	Phillips et al. 1975*
Steelhead	L					5	1.25		29	Phillips et al. 1975*

Species	Study Type	PF (<1.00 mm)	PF (<2.00 mm)	PF (<3.00 mm)	PF (<6.35 mm)	Dg (mm)	Dg/De	Fredle Number	Percent Survival to Emergence	Reference
Steelhead	L					6.5	1.63		32	Phillips et al. 1975*
Steelhead	L					8	2		49	Phillips et al. 1975*
Steelhead	L					11	2.75		53	Phillips et al. 1975*
Steelhead	L					15	3.75		70	Phillips et al. 1975*
Steelhead	L					20	5		99	Phillips et al. 1975*
Steelhead	L			0					99	Phillips et al. 1975
Steelhead	L			10					99	Phillips et al. 1975
Steelhead	L			20					70	Phillips et al. 1975
Steelhead	L			40					50	Phillips et al. 1975
Steelhead	L			50					32	Phillips et al. 1975
Steelhead	L			60					30	Phillips et al. 1975
Steelhead	L			70					15	Phillips et al. 1975
Steelhead	L		0[5]		0	21.5	5.38	17.6	93	Tappel 1981
Steelhead	L		1.8[5]		9	19.1	4.78	14.8	94	Tappel 1981
Steelhead	L		3.5[5]		18	13.9	3.48	9.6	93	Tappel 1981
Steelhead	L		5.2[5]		26.9	10.7	2.68	6	95	Tappel 1981
Steelhead	L		5.7[5]		14.1	16.4	4.1	12.5	92	Tappel 1981
Steelhead	L		5.9[5]		9.9	19.1	4.78	14.8	87	Tappel 1981
Steelhead	L		7[5]		35.9	9.1	2.28	4.5	90	Tappel 1981
Steelhead	L		9.5[5]		23.4	10.4	2.6	6.5	91	Tappel 1981
Steelhead	L		11.7[5]		19.8	11.8	2.95	8.7	86	Tappel 1981
Steelhead	L		13.3[5]		32.8	7.6	1.90	3.5	67	Tappel 1981
Steelhead	L		17.1[5]		42.2	6.1	1.53	2.4	59	Tappel 1981
Steelhead	L		17.6[5]		29.6	6.6	1.65	3	59	Tappel 1981
Steelhead	L		20.9[5]		51.6	5	1.25	1.7	30	Tappel 1981
Steelhead	L		23.4[5]		39.5	4.7	1.18	1.6	14	Tappel 1981
Steelhead	L		29.3[5]		49.4	4	1	1.1	10	Tappel 1981

L = laboratory test; Dg = geometric mean diameter growth; Dg/De = geometric mean diameter divided by mean egg diameter; PF = percent fines.
Egg diameters: Sockeye salmon = 4.75 mm; Chinook salmon = 6.5 mm; Coho salmon = 5.25 mm; Cutthroat trout = 4.7 mm; Steelhead trout = 4.0 mm; Bull trout = 5.2 mm; Brown trout = 4.5 mm.
[1] = PF less than 0.83; [2] = PF less than 3.327; [3] = PF 1 – 3 mm; [4] = PF less than 0.85; [5] = PF less than 1.7.
* data on Dg and Fredle Index were reported from Shirazi and Seim 1979.

Appendix 4. Summary of the available information on biotic responses to streambed substrate composition under field conditions.

Species	Study Type	PF (<1.00 mm)	PF (< 2.00 mm)	PF (< 3.00 mm)	PF (<6.35 mm)	Dg (mm)	Dg/De	Fredle Number	Percent Survival to Emergence	Reference
Atlantic salmon	L		17.7						82.9	Marty et al. 1986
Atlantic salmon	L		18.6						94.1	Marty et al. 1986
Atlantic salmon	L		23.7						86.1	Marty et al. 1986
Atlantic salmon	L		23.6						86.3	Marty et al. 1986
Atlantic salmon	L		32.9						71.6	Marty et al. 1986
Atlantic salmon	L		37.4						75.7	Marty et al. 1986
Atlantic salmon	L		32.4						67.2	Marty et al. 1986
Atlantic salmon	L		58.9						32.2	Marty et al. 1986
Chum salmon	F			2[2]					72	Koski 1972
Chum salmon	F			4[2]					57	Koski 1972
Chum salmon	F			8[2]					27	Koski 1972
Churn salmon	F			8.5[2])					70	Koski 1972
Chum salmon	F			9[2]					75	Koski 1972
Chum salmon	F			11[2]					68	Koski 1972
Chum salmon	F			11[2]					79	Koski 1972
Chum salmon	F			14[2]					75	Koski 1972
Chum salmon	F			15[2]					74	Koski 1972
Chum salmon	F			15[2]					64	Koski 1972
Chum salmon	F			17.5[2]					70	Koski 1972
Chum salmon	F			17.5[2]					98	Koski 1972
Chum salmon	F			22[2]					75	Koski 1972
Chum salmon	F			23[2]					90	Koski 1972
Chum salmon	F			25[2]					55	Koski 1972
Chum salmon	F			28[2]					13	Koski 1972
Chum salmon	F			35[2]					22	Koski 1972
Chum salmon	F			37[2]					42	Koski 1972
Chum salmon	F			37[2]					53	Koski 1972
Chum salmon	F			41[2]					13	Koski 1972
Chum salmon	F			44[2]					16	Koski 1972
Chum salmon	F			45[2]					38	Koski 1972
Chum salmon	F			43[2]					13	Koski 1972

Species	Study Type	PF (<1.00 mm)	PF (< 2.00 mm)	PF (< 3.00 mm)	PF (<6.35 mm)	Dg (mm)	Dg/De	Fredle Number	Percent Survival to Emergence	Reference
Chum salmon	F			50[2]					33	Koski 1972
Coho salmon	F					4.8	0.91		3	Koski 1966
Coho salmon	F					7	1.33		22	Koski 1966
Coho salmon	F					9	1.71		32	Koski 1966
Coho salmon	F					9.5	1.81		51	Koski 1966
Coho salmon	F					17	3.24		68	Koski 1966
Coho salmon	F	17[1]							47	Hall and Campbell 1968
Coho salmon	F	18[1]							57	Hall and Campbell 1968
Coho salmon	F	20[1]							51	Hall and Campbell 1968
Coho salmon	F	20[1]							76.5	Hall and Campbell 1968
Coho salmon	F	21[1]							24	Hall and Campbell 1968
Coho salmon	F	21[1]							62	Hall and Campbell 1968
Coho salmon	F	22.5[1]							40	Hall and Campbell 1968
Coho salmon	F	23[1]							15	Hall and Campbell 1968
Coho salmon	F	23.5[1]							28	Hall and Campbell 1968
Coho salmon	F	23.5[1]							61	Hall and Campbell 1968
Coho salmon	F	25.5[1]								Hall and Campbell 1968
Coho salmon	F	25.5[1]							41	Hall and Campbell 1968
Coho salmon	F	26[1]							1	Hall and Campbell 1968
Coho salmon	F	26[1]							62	Hall and Campbell 1968
Coho salmon	F	27[1]							7	Hall and Campbell 1968
Coho salmon	F	27[1]							31	Hall and Campbell 1968
Coho salmon	F	28[1]							40	Hall and Campbell 1968
Coho salmon	F	30[1]							1	Hall and Campbell 1968
Coho salmon	F	31[1]							1	Hall and Campbell 1968
Coho salmon	F	22.5		32[2]					55	Koski 1972
Coho salmon	F	23.5		37[2]					28	Koski 1972
Coho salmon	F	23.5		36.5[2]					41	Koski 1972
Coho salmon	F	26		38.5[2]					42	Koski 1972
Coho salmon	F	26.5		42[2]					24	Koski 1972

Species	Study Type	PF (<1.00 mm)	PF (< 2.00 mm)	PF (< 3.00 mm)	PF (<6.35 mm)	Dg (mm)	Dg/De	Fredle Number	Percent Survival to Emergence	Reference
Coho salmon	F	27		40(2)					27	Koski 1972
Coho salmon	F	27.5		42(2)					22	Koski 1972
Coho salmon	F	28		42.5(2)					21	Koski 1972
Coho salmon	F	28		43.2(2)					12	Koski 1972
Coho salmon	F	28		43.5(2)					13	Koski 1972
Coho salmon	F	28		43(2)					17	Koski 1972
Coho salmon	F	30		44.5(2)					18	Koski 1972
Coho salmon	F					6	1.14		15	Tagart 1976
Coho salmon	F					9	1.71		24	Tagart 1976
Coho salmon	F					9.5	1.81		37	Tagart 1976
Coho salmon	F					10.5	2		42	Tagart 1976
Cutthroat trout	F					16	3.4		85	Cederholm & Lestelle 1974
Rainbow trout	F					1.0	0.25	0.4	4.3	Sowden and Power 1985
Rainbow trout	F					1.8	0.45	0.8	0.3	Sowden and Power 1985
Rainbow trout	F					2.5	0.63	0.9	0	Sowden and Power 1985
Rainbow trout	F					2.7	0.68	1.4	0.4	Sowden and Power 1985
Rainbow trout	F					3.1	0.78	1.3	0	Sowden and Power 1985
Rainbow trout	F					3.2	0.80	1.8	0.9	Sowden and Power 1985
Rainbow trout	F					3.6	0.90	1.6	9.6	Sowden and Power 1985
Rainbow trout	F					3.7	0.93	1.5	0	Sowden and Power 1985
Rainbow trout	F					3.7	0.93	1.5	0	Sowden and Power 1985
Rainbow trout	F					5.3	1.33	2.0	0.6	Sowden and Power 1985
Rainbow trout	F					6.0	1.50	2.6	I.I	Sowden and Power 1985
Rainbow trout	F					6.1	1.53	2.5	22.1	Sowden and Power 1985
Rainbow trout	F					6.3	1.58	2.8	21.3	Sowden and Power 1985
Rainbow trout	F					6.3	1.58	2.8	21.5	Sowden and Power 1985
Rainbow trout	F					7.1	1.78	3.4	0	Sowden and Power 1985
Rainbow trout	F					7.6	1.9	3.7	5.6	Sowden and Power 1985
Rainbow trout	F					7.9	1.98	3.5	43.5	Sowden and Power 1985
Rainbow trout	F					9.2	2.3	4.4	13.4	Sowden and Power 1985

Species	Study Type	PF (<1.00 mm)	PF (< 2.00 mm)	PF (< 3.00 mm)	PF (<6.35 mm)	Dg (mm)	Dg/De	Fredle Number	Percent Survival to Emergence	Reference
Rainbow trout	F					10.2	2.55	5.0	0	Sowden and Power 1985
Steelhead trout	F					13	3.25		68	Cederholm & Lestelle 1974

F = field test; Dg = geometric mean diameter growth; Dg/De = geometric mean diameter divided by mean egg diameter; PF= percent fines.

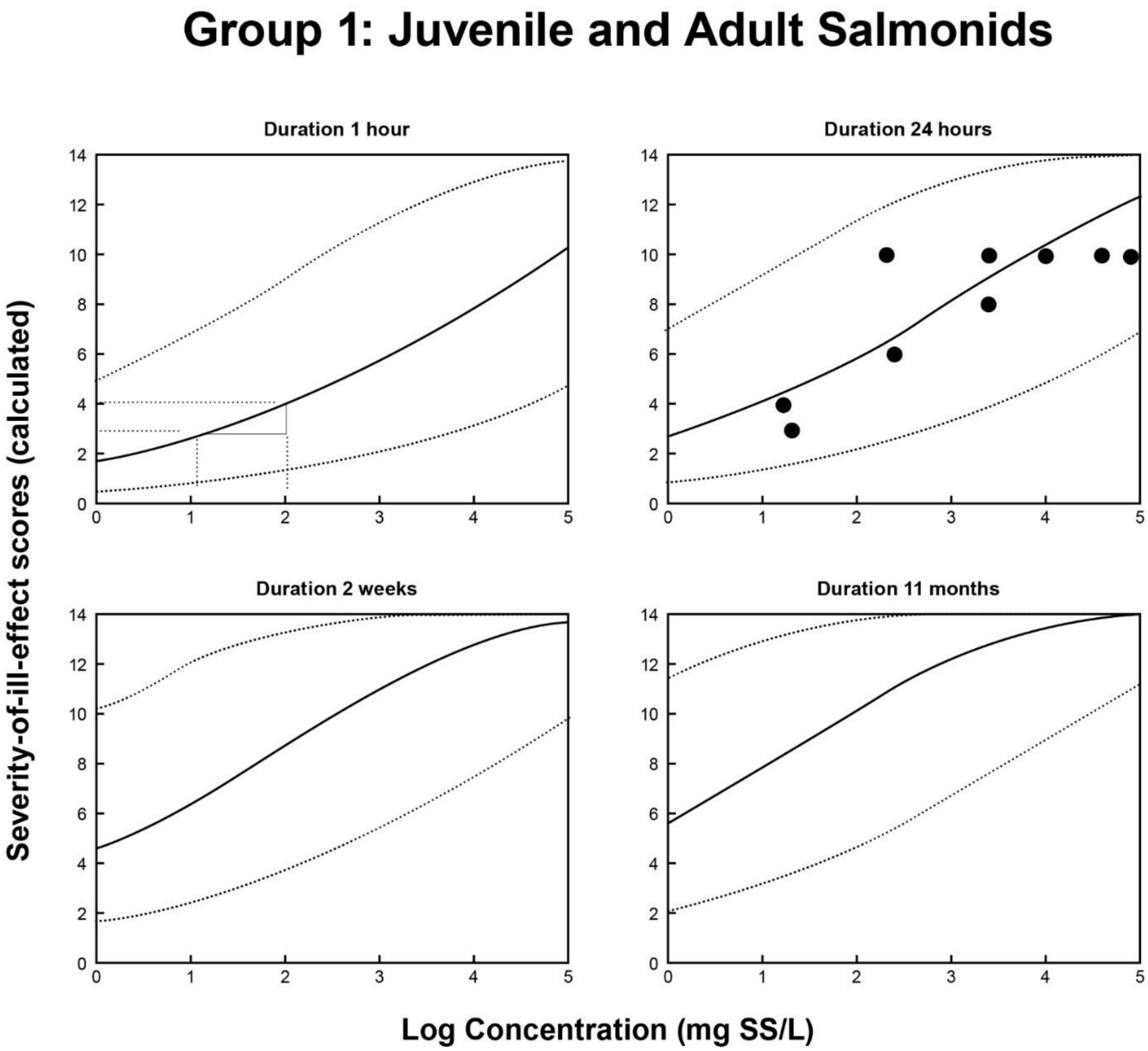
Egg diameters:
Sockeye salmon= 4.7S mm; Chinook salmon = 6.5 mm; Coho salmon = 5.25 mm; Cutthroat trout = 4.7 mm; Steelhead trout= 4.0 mm;
Bull trout = 5.2 mm; and, Brown trout = 4.5 mm (Scot and Crossman 1973).

(1) = PF less than 0.83; (2) = PF less than 3.327; (3) = PF 1-3 mm; (4) = PF less than 0.85; (5) = PF less than 1.7/
* data on Dg and Fredle Index were reported from Shirazi and Seim 1979.

Appendix 5. 95% Confidence Intervals parameters for polynomial equation (created by SAS® statistical package).

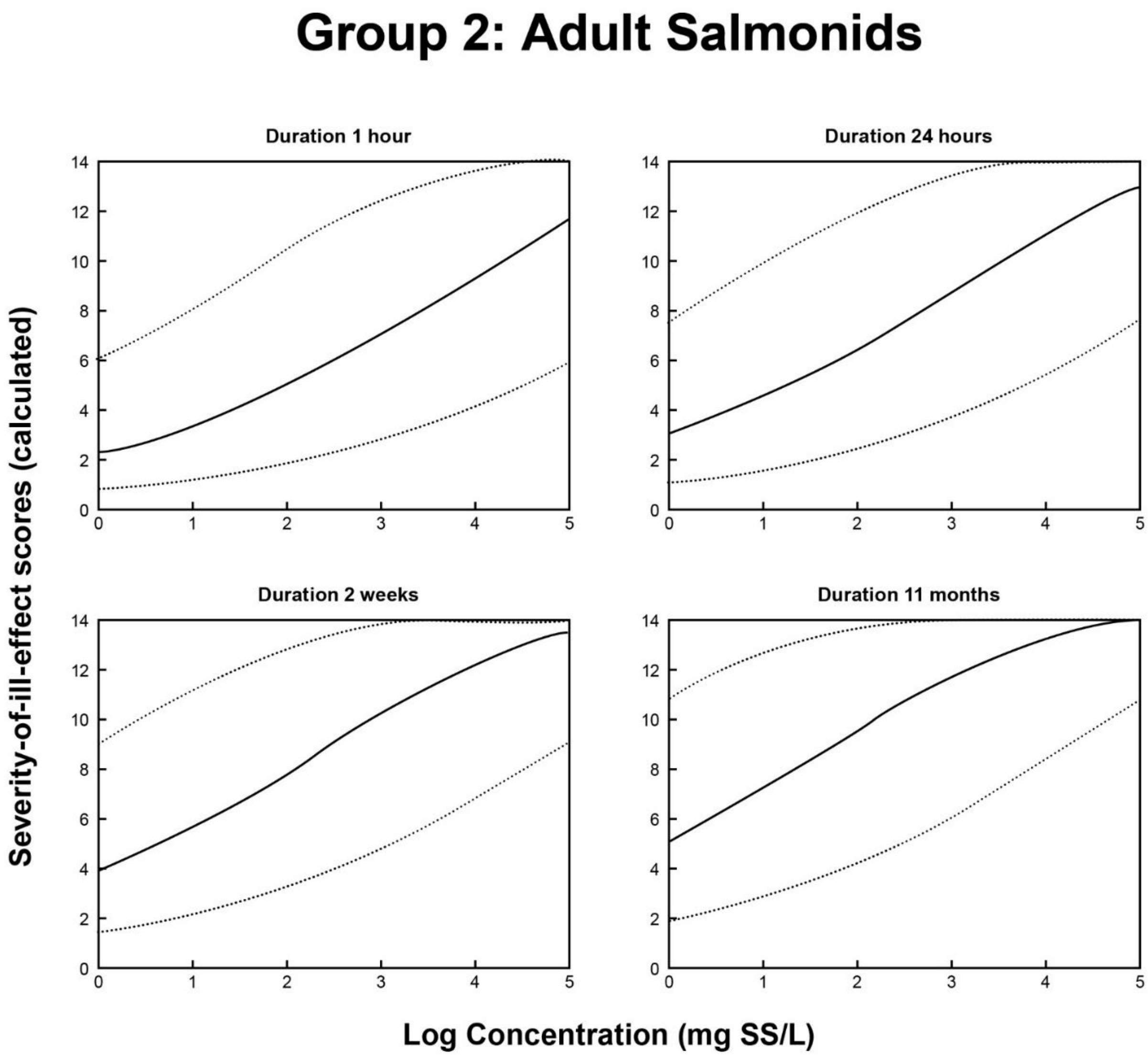
Lower Confidence Limit								
Group	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Intercept	1.8012	2.0139	1.6171	2.4893	1.8158	2.4896	3.0187	2.9494
LNC	0.2073	0.2179	0.2123	0.1188	0.1295	0.1017	0.1366	0.1507
LND	0.1561	0.12	0.1874	0.32	0.6493	0.1922	0.1767	0.1851
LNC^2	-0.0005	-0.001	-0.0012	-0.0042	-0.0041	-0.0022	-0.0014	-0.0015
LND^2	-0.0003	-0.001	-0.0004	-0.0027	-0.0262	-0.0018	-0.0004	-0.0004
Upper Confidence Limit								
Intercept	4.1493	4.383	4.0456	4.8017	5.5852	4.7832	4.8193	4.7276
LNC	0.1927	0.1923	0.1798	0.2957	0.0125	0.044	0.1087	0.1215
LND	0.1494	0.1046	0.1765	0.2674	0.3614	0.1541	0.1699	0.1782
LNC^2	0.0005	0.001	0.0011	0.0042	0.0041	0.0022	0.0014	0.0015
LND^2	0.0005	0.001	0.001	0.0028	0.0262	0.0018	0.0004	0.0004

Group 1: Juvenile and adult salmonids
Group 2: Adult salmonids
Group 3: Juvenile salmonids
Group 4: Eggs and larvae of salmonids and non-salmonids
Group 5: Adult estuarine non-salmonids
Group 6: Adult freshwater non-salmonids
Group 7: Aquatic invertebrates
Group 8: Aquatic invertebrates and aquatic flora



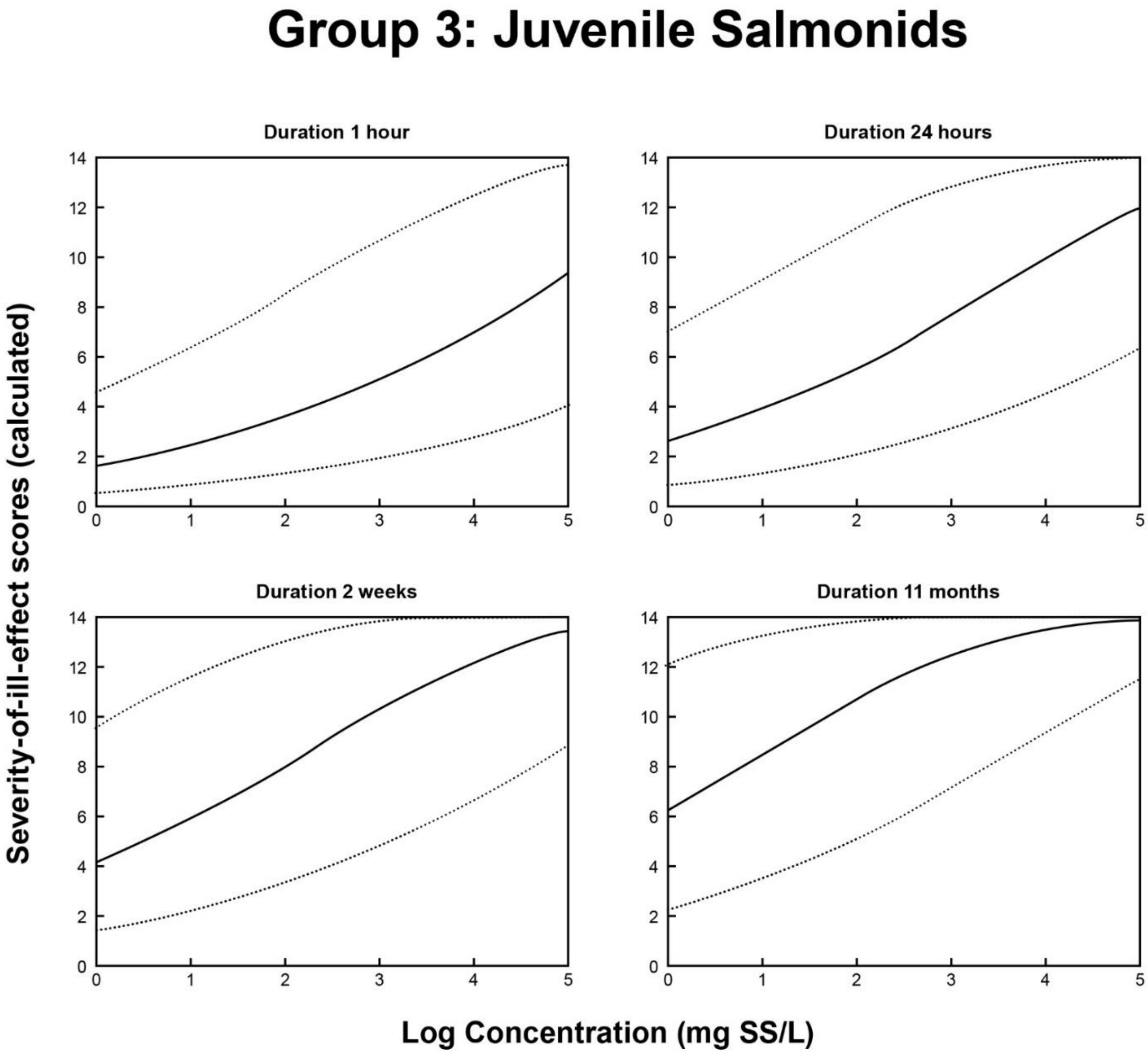
Appendix 6. Group 1: Juvenile and adult salmonids severity-of-ill-effect score vs. concentration at four separate durations. 95% prediction confidence intervals shown, n = 171.

Appendix 7



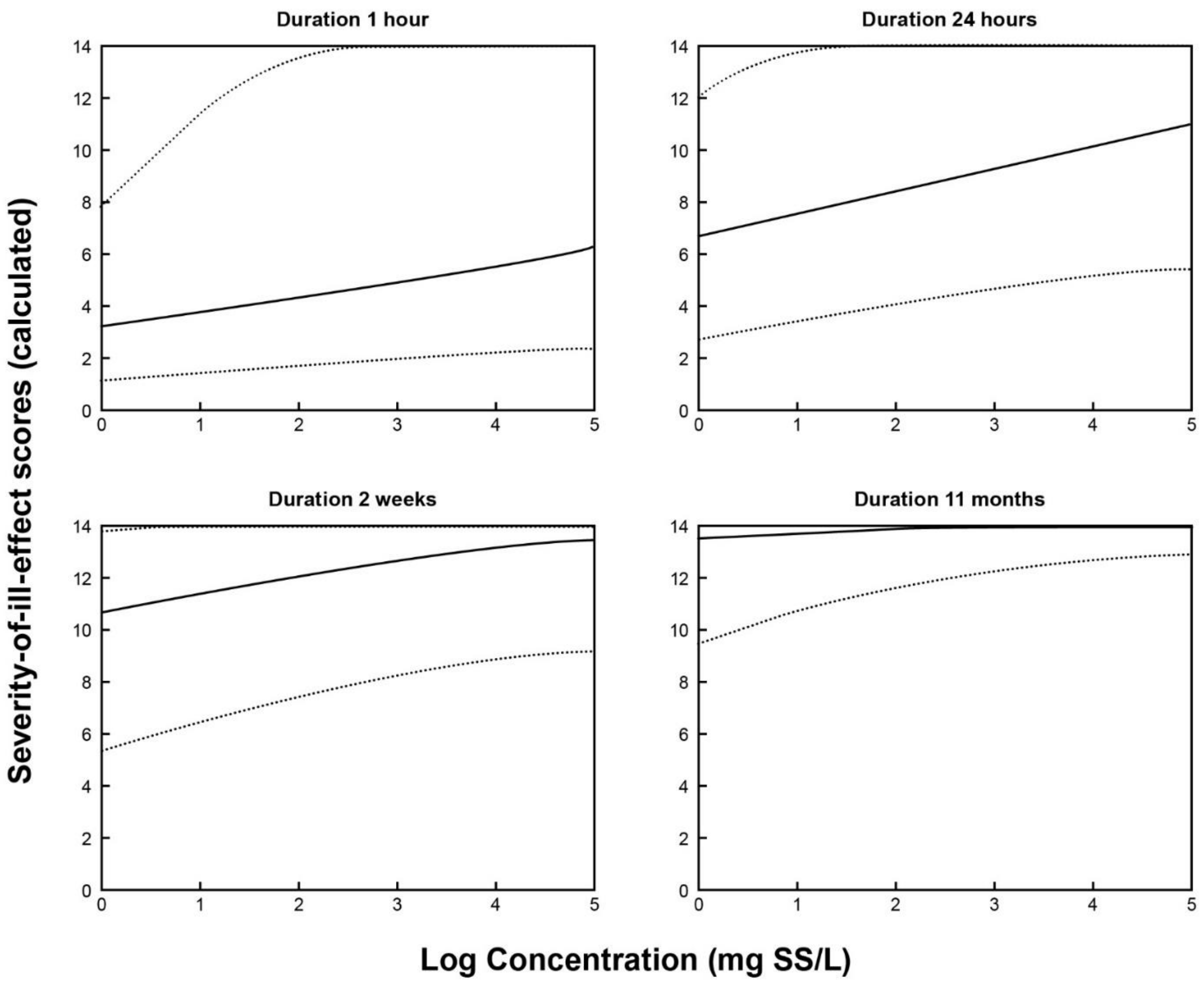
Appendix 7. Group 2: Adult salmonids severity-of-ill-effect score vs. concentration at four separate durations. 95% prediction confidence intervals shown, n = 63.

Appendix 8



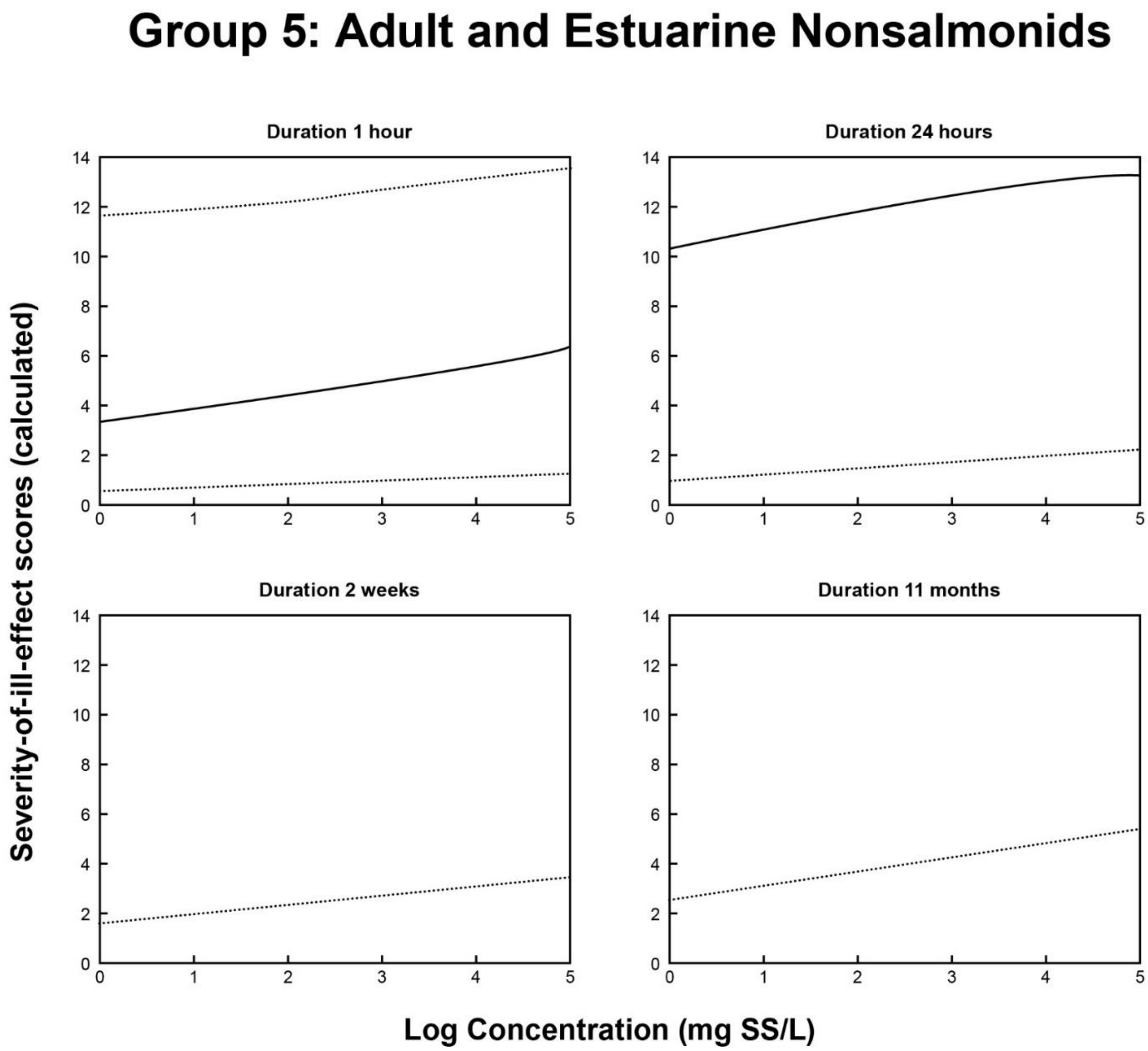
Appendix 8. Group 3: Juvenile salmonids severity-of-ill-effect score vs. concentration at four separate durations. 95% prediction confidence intervals shown, n = 108.

Group 4: Eggs and Larvae of Salmonids and Nonsalmonids



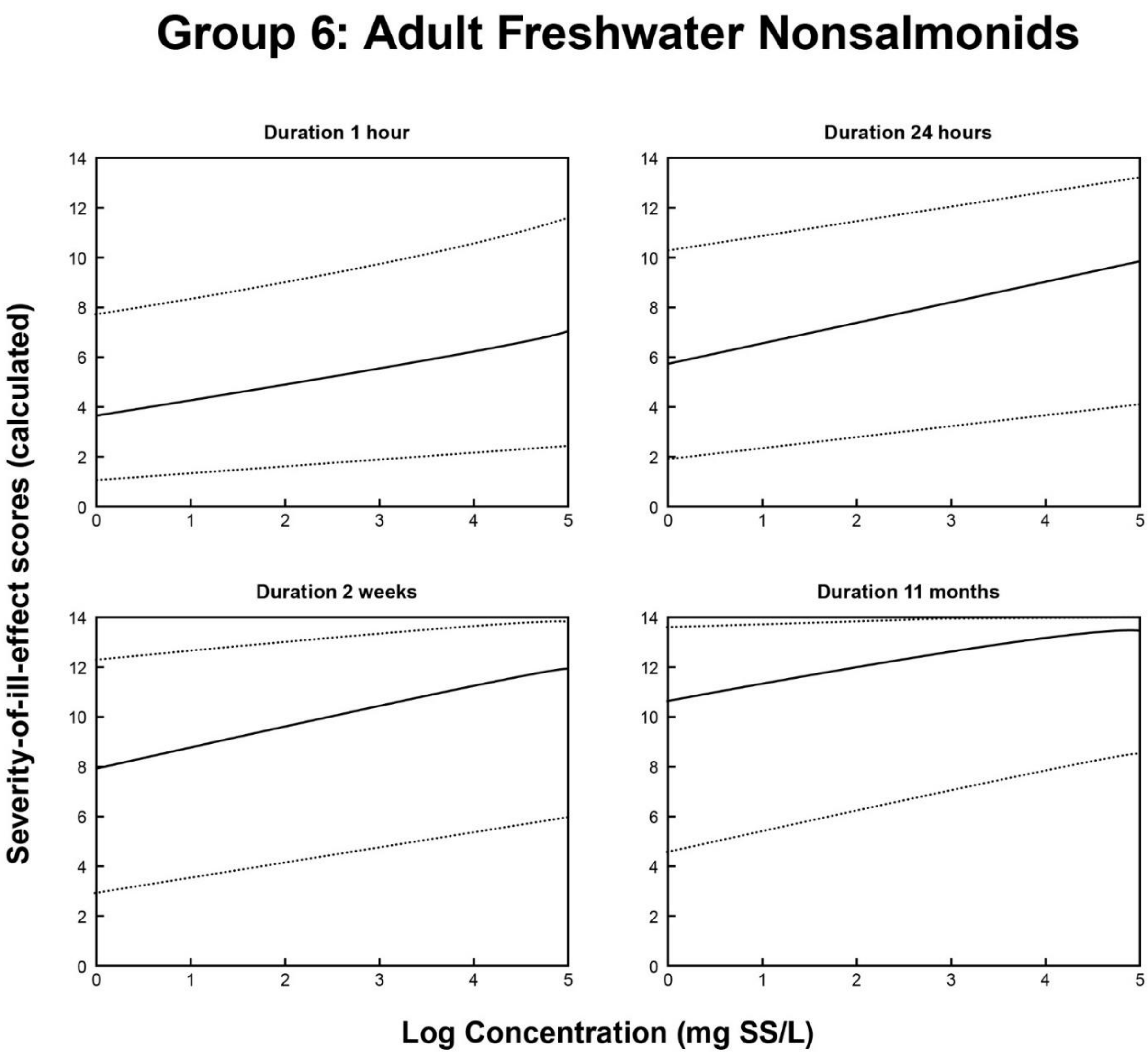
Appendix 9. Group 4: Eggs and larvae of salmonids and non-salmonids severity-of-ill-effect score vs. concentration at four separate durations. 95% prediction confidence intervals shown, n = 43.

Appendix 10

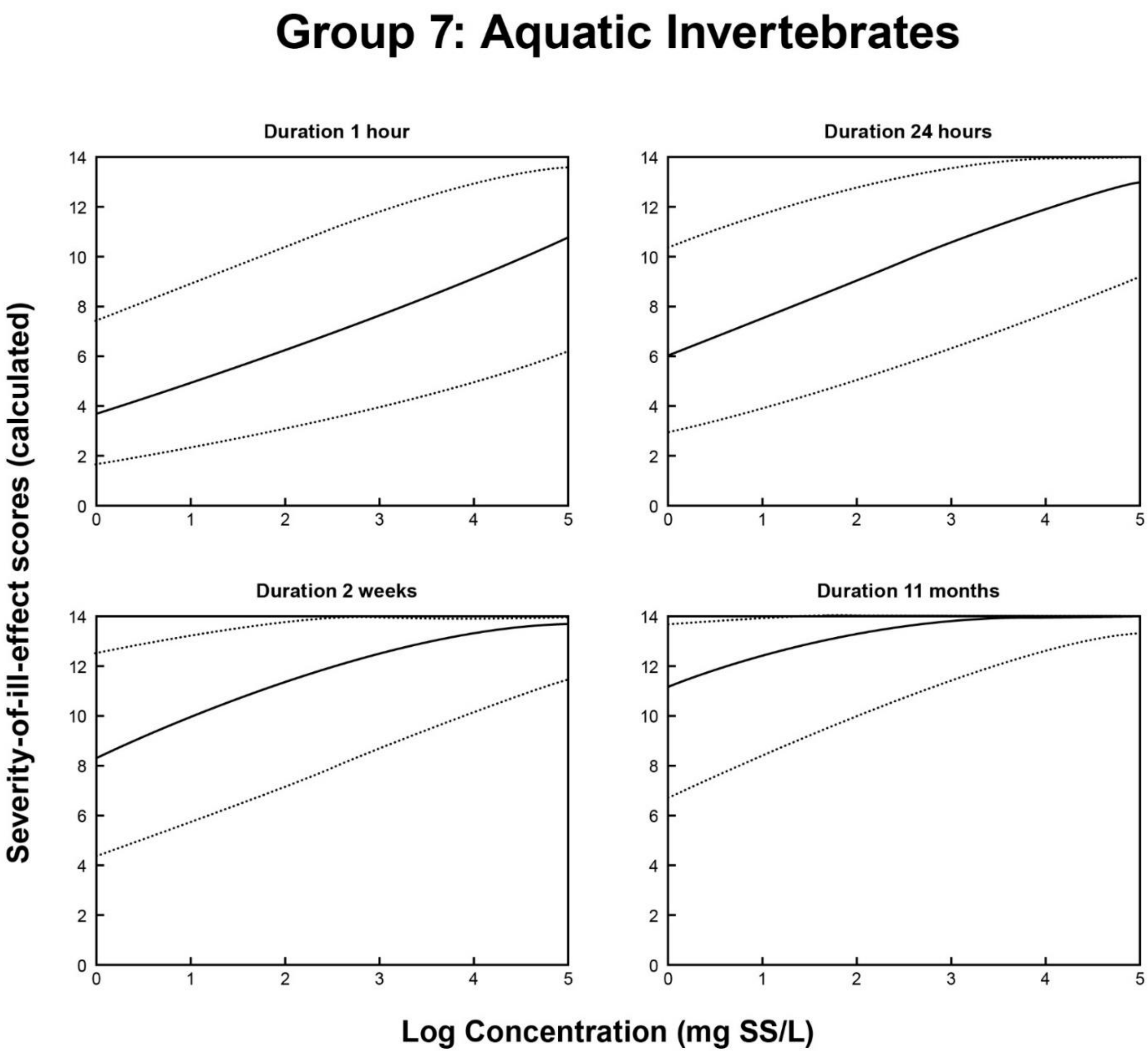


Appendix 10. Group 5: Adult estuarine non-salmonids severity-of-ill-effect score vs. concentration at four separate durations. 95% prediction confidence intervals shown, n = 28.

Appendix 11

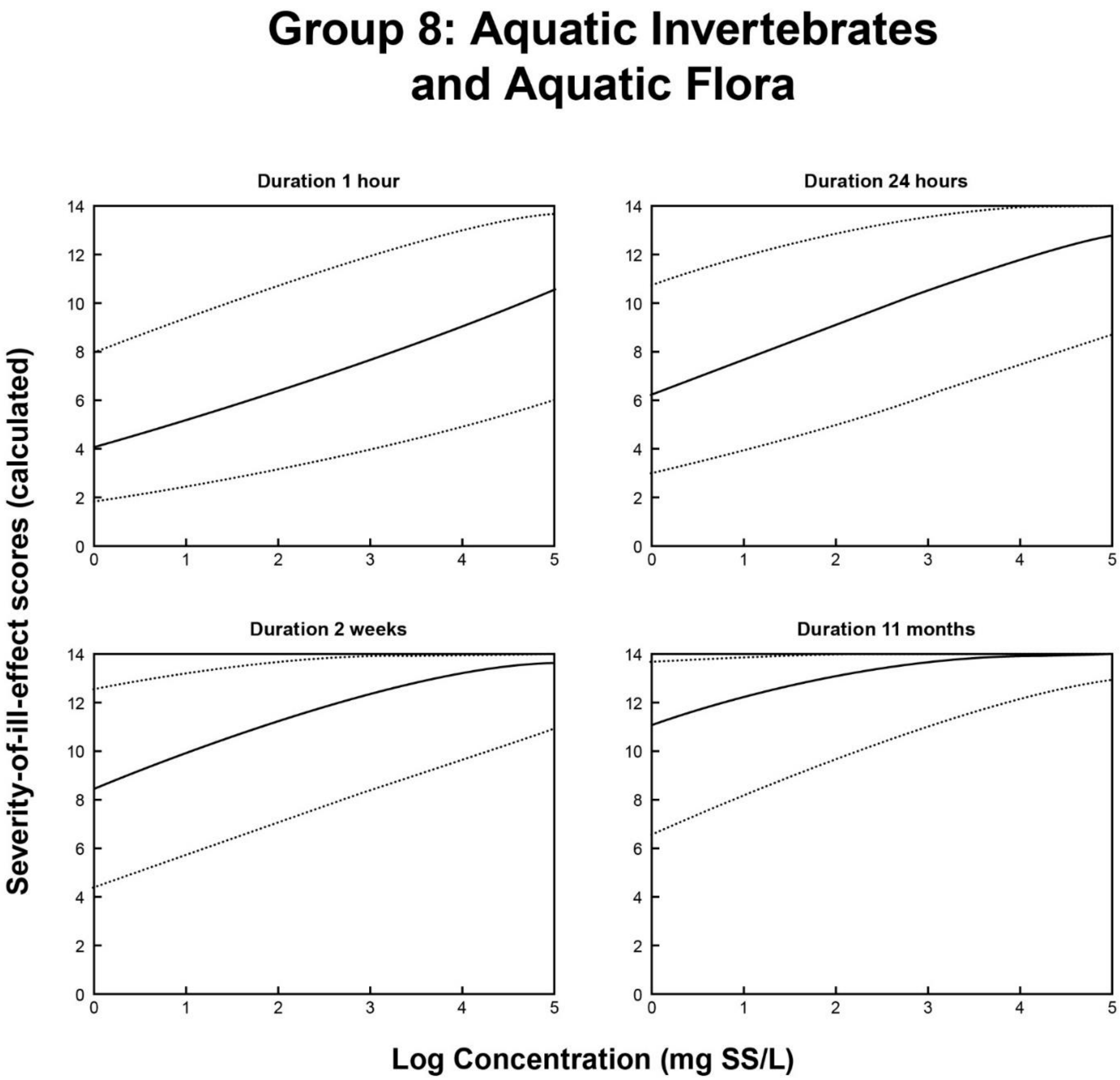


Appendix 11. Group 6: Adult freshwater non-salmonids severity-of-ill-effect score vs. concentration at four separate durations. 95% prediction confidence intervals shown, n = 22.



Appendix 12. Group 7: Aquatic invertebrates severity-of-ill-effect score vs. concentration at four separate durations. 95% prediction confidence intervals shown, n = 69.

Appendix 13



Appendix 13. Group 8: Aquatic invertebrates and flora severity-of-ill-effect score vs. concentration at four separate durations. 95% prediction confidence intervals shown, n = 61.