APPENDIX 4



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# Channel Suspended Sediment and Fisheries: A Synthesis for Quantitative Assessment of Risk and Impact

CHARLES P. NEWCOMBE

*Habitat Protection Branch, Ministry of Environment, Lands and Parks*
*Victoria, British Columbia V8V 1X4, Canada*

JORGEN O. T. JENSEN

*Science Branch, Pacific Biological Station, Department of Fisheries and Oceans*
*Nanaimo, British Columbia V9R 5K6, Canada*

*Abstract.*—Our meta-analysis of 80 published and adequately documented reports on fish responses to suspended sediment in streams and estuaries has yielded six empirical equations that relate biological response to duration of exposure and suspended sediment concentration. These equations answer an important need in fisheries management; quantifying the response of fishes to suspended sediment pollution of streams and estuaries has been difficult historically, and the lack of a reliable metric has hindered assessment for risk and impact for fishes subjected to excess sedimentation. The six equations address various taxonomic groups of lotic, lentic, and estuarine fishes, life stages of species within those groups, and particle sizes of suspended sediments. The equations all have the form

$$z = a + b(\log_e x) + c(\log_e y);$$

$z$ is severity of ill effect, $x$ is duration of exposure (h), $y$ is concentration of suspended sediment (mg SS/L), $a$ is the intercept, and $b$ and $c$ are slope coefficients. The severity of ill effect ($z$) is delineated semiquantitatively along a 15-point scale on which is superimposed four "decision" categories ranging from no effect through behavioral and sublethal effects to lethal consequences (a category that also includes a range of paralethal effects such as reduced growth rate, reduced fish density, reduced fish population size, and habitat damage). The study also provided best available estimates of the onset of sublethal and lethal effects, and it supported the hypothesis that susceptible individuals are affected by sediment doses (concentration × exposure duration) lower than those at which population responses can be detected. Some species and life stages show "ultrasensitivity" to suspended sediment. When tested against data not included in the analysis, the equations were robust. They demonstrate that meta-analysis can be an important tool in habitat impact assessment.

While it is now generally accepted that the severity of effect of suspended sediment pollution on fish increases as a function of sediment concentration and duration of exposure, or dose (the product of concentration and exposure time), attempts to document the dose–response relationship for sediment and aquatic organisms have been limited in several ways. First, initial analyses were based on pooled data (Newcombe 1986; Newcombe and MacDonald 1991). Second, the database available for those analyses embraced a wide taxonomic range from phytoplankton to fish. Third, the database contained little information about particular species and life stages. The resulting dose–response model for aquatic ecosystems (Newcombe 1986; Newcombe and MacDonald 1991) established a general principle, but this model was held to be too imprecise to help fishery and habitat managers address local sediment problems (Gregory et al. 1993).

In an effort to refine the general dose–response model, MacDonald and Newcombe (1993) extracted and analyzed data for juvenile salmon from the recent literature. These data yielded an equation similar to the one for pooled data, but the two curves differed in important ways. This finding established a need to revisit the dose–response database so that models could be tailored to particular groups of fishes as functions of taxonomic group, natural history, life history phase, and predominant sizes of the sediment particles responsible for ill effects (Newcombe 1994). We have endeavored to meet this need and present a meta-analytic synthesis of dose–response data in this paper. Insofar as this research provides new understanding of channel sediment impacts, it leads to discussion of potential changes in the methods and goals of quantitative impact assessment. Specifically, the results (i) suggest the need to change the methods of data collection for environmental law enforcement, (ii) demonstrate the value of meta-analysis as a research method in fisheries habitat impact assessment, and (iii) prompt an expression of concern about land use practices and protection of instream, riparian, and upland zones.

TABLE 1.—Scale of the severity (SEV) of ill effects associated with excess suspended sediment.

| SEV | Description of effect |
|---|---|
| | **Nil effect** |
| 0 | No behavioral effects |
| | **Behavioral effects** |
| 1 | Alarm reaction |
| 2 | Abandonment of cover |
| 3 | Avoidance response |
| | **Sublethal effects** |
| 4 | Short-term reduction in feeding rates; short-term reduction in feeding success |
| 5 | Minor physiological stress; increase in rate of coughing; increased respiration rate |
| 6 | Moderate physiological stress |
| 7 | Moderate habitat degradation; impaired homing |
| 8 | Indications of major physiological stress; long-term reduction in feeding rate; long-term reduction in feeding success; poor condition |
| | **Lethal and paralethal effects** |
| 9 | Reduced growth rate; delayed hatching; reduced fish density |
| 10 | 0–20% mortality; increased predation; moderate to severe habitat degradation |
| 11 | >20–40% mortality |
| 12 | >40–60% mortality |
| 13 | >60–80% mortality |
| 14 | >80–100% mortality |

## Methods

This study is based on 264 data triplets consisting of (i) suspended sediment concentration, (ii) duration of exposure, and (iii) severity of ill effect for fishes. These data were taken from a comprehensive literature review (Newcombe 1994; Newcombe et al. 1995). Supporting data extracted from the review included taxonomic group, species of fish, natural history, life history phase, and sediment particle size range.

We define dose as concentration of suspended sediment (SS) times duration of exposure; dose has the units mg $SS \cdot h \cdot L^{-1}$. The natural logarithm of dose is termed the stress index (Newcombe 1986, 1994; Newcombe and MacDonald 1991; MacDonald and Newcombe 1993). Response is the severity of ill effect, described below. The dose–response matrix, which is the basis of data presentation in this report, encompasses all combinations of sediment concentration (1–500,000 mg SS/L) and exposure duration (1–35,000 h). Except when it refers specifically to duration, we use "exposure" broadly to include dose, particle size, and other potential contributors to stress on fishes. In most cases, data on particle shape and roughness and on water temperature were lacking.

### *Severity-of-Ill-Effect Scale*

As before (MacDonald and Newcombe 1993; Newcombe 1994) and in a nearly identical way, we scored qualitative response data along a semi-quantitative ranking scale (Table 1). Superimposed on a 15-point scale (0–14) were four major classes of effect: (i) nil effect, (ii) behavioral effects, (iii) sublethal effects (a category that also includes effects such as short-term reduction in feeding success), and (iv) lethal effects (direct mortality, or its paralethal surrogates—reduced growth, reduced fish density, habitat damage such as reduced porosity of spawning gravel, delayed hatching, and reduction in population size). When these various effects could be compared directly, pollution episodes associated with sublethal or lethal effects



also degraded habitat and reduced population siz which is why these seemingly disparate ill effec are grouped together in the hierarchy. For ever between the extremes of nil effect and 100% mo tality, we assumed for modeling purposes that t severity-of-ill effects (SEV for "severity") sc represents proportional differences in true effec

We now incorporate all feeding reductions the class of sublethal effects, and we set the boun ary between short-term and long-term reductic in feeding success at 2 h. In practice, reports long-term disruption of feeding rates encomp 800 h and more. We consider all feeding reductic to be sublethal effects (unless feeding reductic can be linked to slow growth when we t as paralethal effects) because they reflec change in fish behavior than reduced availabi of food and reduced visual hunting range.

Along the SEV scale, habitat damage ran from moderate to severe. Habitat damage can characterized in biological or physical terms both of these in conjunction. Biological manif tations of habitat damage include underutilizat of stream habitat (Birtwell et al. 1984), aband ment of traditional spawning habitat (Hamil 1961), displacement of fish from their hab (McLeay et al. 1987), and avoidance of hab (Swenson 1978). Physical manifestations incl degradation of spawning habitat (Slaney et 1977b; Cederholm et al. 1981), damage to hab structure (Newcomb and Flagg 1983; Menze al. 1984), and loss of habitat (Menzel et al. 19 Coats et al. 1985). Biophysical manifestation excess SS are reported (in one typical example habitat degradation that reduces the relative cess of one or more fish species that depend low siltation rates and silt-free (<3% silt) ri (Berkmann and Rabeni 1987).

Habitat degradation can be inferred by (i) dence of increased mortality at any stage in a f life cycle (egg-to-fry survival may decrease result of increased sedimentation: J. LaPerr University of Alaska, personal communicat (ii) avoidance behavior by fishes (Suchanek 1984a, 1984b), (iii) reduced abundance of in and reduced quality of rearing habitat (Slane al. 1977b), (iv) decreased size of zoobenthic ulations (Gammon 1970; Rosenberg and S 1977), (v) reduced utility of spawni (Hamilton 1961), (vi) delayed hatching and Wang 1973), and (vii) disruption of ho behavior and home water preference (Brann al. 1981; Whitman et al. 1982).

Relative severity of habitat damage is a co also degraded habitat and reduced population size, which is why these seemingly disparate ill effects are grouped together in the hierarchy. For events between the extremes of nil effect and 100% mortality, we assumed for modeling purposes that the severity-of-ill effects (SEV for "severity") scale represents proportional differences in true effects.

We now incorporate all feeding reductions in the class of sublethal effects, and we set the boundary between short-term and long-term reductions in feeding success at 2 h. In practice, reports of long-term disruption of feeding rates encompass 800 h and more. We consider all feeding reductions to be sublethal effects (unless feeding reductions can be linked to slow growth when we treat them as paralethal effects) because they reflect less a change in fish behavior than reduced availability of food and reduced visual hunting range.

Along the SEV scale, habitat damage ranges from moderate to severe. Habitat damage can be characterized in biological or physical terms or both of these in conjunction. Biological manifestations of habitat damage include underutilization of stream habitat (Birtwell et al. 1984), abandonment of traditional spawning habitat (Hamilton 1961), displacement of fish from their habitat (McLeay et al. 1987), and avoidance of habitat (Swenson 1978). Physical manifestations include degradation of spawning habitat (Slaney et al. 1977b; Cederholm et al. 1981), damage to habitat structure (Newcomb and Flagg 1983; Menzel et al. 1984), and loss of habitat (Menzel et al. 1984; Coats et al. 1985). Biophysical manifestations of excess SS are reported (in one typical example) as habitat degradation that reduces the relative success of one or more fish species that depend on low siltation rates and silt-free (<3% silt) riffles (Berkmann and Rabeni 1987).

Habitat degradation can be inferred by (i) evidence of increased mortality at any stage in a fish's life cycle (egg-to-fry survival may decrease as a result of increased sedimentation: J. LaPerriere, University of Alaska, personal communication), (ii) avoidance behavior by fishes (Suchanek et al. 1984a, 1984b), (iii) reduced abundance of insects and reduced quality of rearing habitat (Slaney et al. 1977b), (iv) decreased size of zoobenthic populations (Gammon 1970; Rosenberg and Snow 1977), (v) reduced utility of spawning habitat (Hamilton 1961), (vi) delayed hatching (Schubel and Wang 1973), and (vii) disruption of homing behavior and home water preference (Brannon et al. 1981; Whitman et al. 1982).

Relative severity of habitat damage is a continuum on a two-dimensional plane (SS concentration × duration of SS exposure) in which an event may be minor (ephemeral or low SS concentration or both), or major (long term or high SS concentration or both), or anywhere between these extremes. Severe habitat damage has been described by various authors, some of whom used aquatic invertebrates as indicators (Herbert and Richards 1963; Vaughan 1979; Vaughan et al. 1982; Menzel et al. 1984; Wagener and LaPerriere 1985). Severity of habitat damage caused by excess SS sometimes has been reported in terms of the length of time required for the stream to return to its natural state—sometimes as long as 15–20 years (estimated) after extensive coal mining (Vaughan et al. 1982).

The distinction between moderate and severe habitat damage is a matter of degree that still has not been delineated exactly. Severe habitat damage can be characterized in its extreme by the absence of fish where fish normally are found or by substantial reduction in fish popultion size, as was documented for brown trout by Herbert et al. (1961). (Scientific names of fish species are given in Table 2.) A pollution event that results in the deposition of suspended sediment in or on spawning habitat during egg incubation might be considered "moderately severe" if the area affected were a small portion of the total available. On the other hand, chronic or acute SS pollution that causes substantial reduction in the size of riverine fish populations (Herbert et al. 1961; Stober et al. 1981) should be considered to represent "severe" habitat damage. Likewise, major SS pollution that results in extensive deposition of sediment on spawning grounds should be characterized as severe habitat damage because its effects could reduce the strength of an entire year-class.

Habitat damage is a valid description of the harm caused by SS pollution, but it is probably an abstraction insofar as ill effects operate on one or more life stages of a fish's life cycle. Age-specific morbidity and mortality rates are fundamental to the notion of habitat damage. For example, habitat damage may manifest itself as foregone opportunity for fish to use a portion of a stream. Reduced suitability of habitat could result in increased age-specific morbidity and mortality rates, or both, depending on the focus and methods of a study. Habitat damage, therefore, should be seen as an accumulative measure of numerous (potentially undocumented) ill effects at various stages in a fish's life cycle. It is a unique phenomenon in that it can only be studied in the field (in contrast to direct effects—age-specific morbidity and mortality, for example—that can be studied in the laboratory as well as in the field). Thus the documented harm caused by excess *SS*—especially when it is not known by direct observation to have caused an increase in morbidity or mortality rates—can reasonably be characterized in more general terms as habitat damage.

## Model Formulation

From the expanded database (see Appendix Table A.1), six groupings of fish data were identified for which sample sizes were large enough to support modeling. The six groupings arose from various combinations of four attributes: taxonomic group, life stage, life history, and particle size of suspended sediment.

*Taxonomy.*—Salmonids (family Salmonidae) were distinguished from nonsalmonids, although several groupings were not exclusively one or the other.

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

*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.*—The predominant sizes of suspended sediment particles reported in the database literature ranged up to 250 μm. We collated sizes into two categories separated at 75 μm. *Fine* particles were smaller than 75 μm, small enough to pass through gill membranes into interlamellar spaces of gill tissue. This category includes clay, silt, and very fine sand particles (Agriculture Canada 1974). *Coarse* particles were 75–250 μm in diameter, large enough to cause mechanical abrasion of gills. This size range includes very fine to fine sand particles.

The six data groups for which we developed models follow. Species in each group are listed in Table 2.

*Group 1: juvenile and adult salmonids; particle sizes 0.5–250 μm.*—Group 1 ($N$ = 171 studies or experimental units) includes Atlantic and Pacific salmon, trout, Arctic grayling, mountain whitefish, and rainbow smelt (a nonsalmonid). Some studies dealt with fine sediment as categorized above, some with coarse sediment, and some with both.

TABLE 2.—Common and scientific names of fish species and other taxa mentioned in this paper and the sediment effects model(s) to which they contributed. Species without a model number were not used in any model.

| Common name | Scientific name | Model |
|---|---|---|
| Anchovy (bay) | *Anchoa mitchilli* | 5[a] |
| Bass (largemouth) | *Micropterus salmoides* | 6 |
| Bass (smallmouth) | *Micropterus dolomieu* | |
| Bass (striped) | *Morone saxatilis* | 4,5 |
| Bluegill | *Lepomis macrochirus* | 6 |
| Carp (common) | *Cyprinus carpio* | 6 |
| Cunner | *Tautogolabrus adspersus* | 5 |
| Darters | Percidae; includes *Semotilus atromaculatus*[b] | 6 |
| Fish | (Genus and species obscure) | 5 |
| Fish (warmwater) | (Genus and species obscure) | 5,6 |
| Goldfish | *Carassius auratus* | 6 |
| Grayling (Arctic) | *Thymallus arcticus* | 1–4 |
| Herring (Atlantic) | *Clupea harengus* | 4,5[a] |
| Herring (lake) | *Coregonus artedi* | 4 |
| Herring (Pacific) | *Clupea pallasi* | 4 |
| Hogchoker | *Trinectes maculatus* | 5 |
| Killifish (striped) | *Fundulus majalis* | 5 |
| Menhaden (Atlantic) | *Brevoortia tyrannus* | 5[a] |
| Minnow (sheepshead) | *Cyprinodon variegatus* | 5[a] |
| Mummichog | *Fundulus heteroclitus* | 5 |
| Perch (white) | *Morone americana* | 4,5 |
| Perch (yellow) | *Perca flavescens* | 4 |
| Rasbora (harlequin) | *Rasbora heteromorpha* | 5 |
| Salmon | (Genus and species obscure) | 1,2,4 |
| Salmon (Atlantic) | *Salmo salar* | 1,2 |
| Salmon (chinook) | *Oncorhynchus tshawytscha* | 1–3 |
| Salmon (chum) | *Oncorhynchus keta* | 1,3,4 |
| Salmon (coho) | *Oncorhynchus kisutch* | 1,3,4 |
| Salmon (Pacific) | *Oncorhynchus* spp. | 1,2 |
| Salmon (sockeye) | *Oncorhynchus nerka* | 1–3 |
| Shad (American) | *Alosa sapidissima* | 4,5 |
| Silverside (Atlantic) | *Menidia menidia* | 5[a] |
| Smelt (rainbow) | *Osmerus mordax* | 1,2 |
| Spot | *Leiostomus xanthurus* | 5[a] |
| Steelhead | *Oncorhynchus mykiss* (anadromous) | 1–4 |
| Stickleback (fourspine) | *Apeltes quadracus* | 5[a] |
| Stickleback (threespine) | *Gasterosteus aculeatus* | 5 |
| Sunfish (green) | *Lepomis cyanellus* | 6 |
| Sunfish (redear) | *Lepomis microlophus* | 6 |
| Toadfish (oyster) | *Opsanus tau* | 5 |
| Trout | (Genus and species obscure) | 1,2,4 |
| Trout (brook) | *Salvelinus fontinalis* | 1–3 |
| Trout (brown) | *Salmo trutta* | 1,2 |
| Trout (cutthroat) | *Oncorhynchus clarki* | 1,2 |
| Trout (lake) | *Salvelinus namaycush* | 1,2 |
| Trout (rainbow) | *Oncorhynchus mykiss* | 1–4 |
| Trout (sea) | (Genus and species obscure) | 1,2 |
| Whitefish (lake) | *Coregonus clupeaformis* | 1,2 |
| Whitefish (mountain) | *Prosopium williamsoni* | 1,2 |

[a] A relatively sensitive species used in the empirical model for estuarine species.

[b] Creek chubs are included with darters here because the relevant study (Vaughan et al. 1978) referred to reduced fish abundance in streams where chubs and darters were reported to live.



TABLE 3.—Attributes, slopes and coefficients, and statis (z, 15-point scale) to duration of exposure (x, h) and conce + $b(\log_e x)$ + $c(\log_e y)$.

| Term | 1 | 2 |
|---|---|---|
| | | Attri |
| Taxon[a] | S | S |
| Life stage[b] | J + A | A |
| Life history[c] | FW | FW |
| Sediment particle size[d] | F to C | F to C |
| | | Slopes and |
| Intercept ($a$) | 1.0642 | 1.6814 |
| Slope of $\log_e x$ ($b$) | 0.6068 | 0.4769 |
| Slope of $\log_e y$ ($c$) | 0.7384 | 0.7565 |
| | | Statis |
| Coefficient of determination[e] ($r^2$) | 0.6009 | 0.6173 |
| $F$-statistic | 130.28 | 52.37 |
| Probability ($P$) | <0.01 | <0.01 |
| Sample size ($N$) | 171 | 63 |

[a] S = salmonids (predominantly); N = nonsalmonids.

[b] A = adults; J = juveniles; L = larvae; E = eggs.

[c] FW = freshwater and anadromous; ES = estuarine.

[d] F = fine (predominantly <75 μm); C = coarse (75–250 μm).

[e] Corrected for degrees of freedom.

*Group 2: adult salmonids; particle sizes 0.5–250 μm.*—Group 2 ($N$ = 63) is a subset of group 1.

*Group 3: juvenile salmonids; particle sizes 0.5–75 μm.*—Group 3 ($N$ = 108) is a subset of group 1. In a few cases, sediment sizes were as large as 150 μm.

*Group 4: eggs and larvae of salmonids and nonsalmonids; particle sizes 0.5–75 μm.*—Group 4 ($N$ = 43) includes salmonids that do not bury their eggs. Nonsalmonids comprise species that spawn in rivers, lakes, and estuaries. Sediment sizes exceeded 75 μm in a few studies.

*Group 5: adult estuarine nonsalmonids; particle sizes 0.5–75 μm.*—Group 5 ($N$ = 28) includes several species believed to be particularly sensitive to the effects of suspended sediment; these are footnoted in Table 2. Some test sediments exceeded 75 μm.

*Group 6: adult freshwater nonsalmonids; particle sizes 0.5–75 μm.*—Group 6 ($N$ = 22) includes both lentic and lotic species. Particle sizes exceeded 75 μm in some cases.

For each group, the severity of effect (SEV, 15-point scale, 0–14) was regressed on suspended sediment dose (exposure duration [ED, h] and suspended sediment concentration [mg SS/L]). Preliminary analyses indicated that logarithmic transformations of ED and concentration provided suitably linear relations of the form

TABLE 3.—Attributes, slopes and coefficients, and statistics of six models that relate severity of ill effect on fishes ($z$, 15-point scale) to duration of exposure ($x$, h) and concentration of suspended sediment ($y$, mg/L) in the form $z = a + b(\log_e x) + c(\log_e y)$.

| Term | Model 1 | Model 2 | Model 3 | Model 4 | Model 5 | Model 6 |
|---|---|---|---|---|---|---|
| | | | **Attributes** | | | |
| Taxon[a] | S | S | S | S + N | N | N |
| Life stage[b] | J + A | A | J | E + L | A | A |
| Life history[c] | FW | FW | FW | FW + ES | ES | FW |
| Sediment particle size[d] | F to C | F to C | F | F | F | F |
| | | | ***Slopes and coefficients*** | | | |
| Intercept ($a$) | 1.0642 | 1.6814 | 0.7262 | 3.7466 | 3.4969 | 4.0815 |
| Slope of $\log_e x$ ($b$) | 0.6068 | 0.4769 | 0.7034 | 1.0946 | 1.9647 | 0.7126 |
| Slope of $\log_e y$ ($c$) | 0.7384 | 0.7565 | 0.7144 | 0.3117 | 0.2669 | 0.2829 |
| | | | **Statistics** | | | |
| Coefficient of determination[e] ($r^2$) | 0.6009 | 0.6173 | 0.5984 | 0.5516 | 0.6200 | 0.6998 |
| $F$-statistic | 130.28 | 52.37 | 82.00 | 28.03 | 24.50 | 27.42 |
| Probability ($P$) | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 | <0.01 |
| Sample size ($N$) | 171 | 63 | 108 | 43 | 28 | 22 |

[a] S = salmonids (predominantly); N = nonsalmonids.
[b] A = adults; J = juveniles; L = larvae; E = eggs.
[c] FW = freshwater and anadromous; ES = estuarine.
[d] F = fine (predominantly <75 μm); C = coarse (75–250 μm).
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*Group 2: adult salmonids; particle sizes 0.5–250 μm.*—Group 2 ($N$ = 63) is a subset of group 1.

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For each group, the severity of effect (SEV, 15-point scale, 0–14) was regressed on suspended sediment dose (exposure duration [ED, h] and suspended sediment concentration [mg SS/L]). Preliminary analyses indicated that logarithmic transformations of ED and concentration provided suitably linear relations of the form

$$\text{SEV} = a + b(\log_e \text{ED}) + c(\log_e \text{mg SS/L});$$

intercepts ($a$) and slope coefficients ($b$ and $c$) emerged from the fitting exercise. Commercial software was used for the regressions (TableCurve 3D; Jandel Scientific). Coefficients of determination ($r^2$) were adjusted for degrees of freedom ($r^2$ = 1 − [sum of squares due to error]/[sum of squares around the mean]). The software also generated $F$-statistics, $P$-values, and 95% confidence intervals around the SEVs. Although arithmetic values for exposure duration and concentration are also given in the Results and in the Appendix, the models we present are based on logarithmic transformations.

The regressions, having been fitted to the data, become predictive models of the form

$$z = a + b(\log_e x) = c(\log_e y),$$

for which $z$ is calculated severity of ill effect (SEV), $x$ is an estimate of exposure duration (ED), and $y$ is the concentration of the (estimated) predominant suspended sediment size (mg SS/L). These predictive models are numbered 1–6 to correspond with the data groupings already described. Because of scatter even in the fitted data, the predictive equations can yield severity-of-ill-effect ($z$) values greater than 14, which already includes the

Juvenile and Adult Salmonids

Duration of exposure to SS ($\log_e$ hours)

(A) Average severity-of-ill-effect scores (empirical)

| Concentration (mg SS/L) | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | ($\log_e$ mg SS/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 162755 | 14 | - | - | 14 | - | - | - | - | - | - | - | 12 |
| 59874 | - | - | 12 | 10 | 14 | 12 | - | - | - | - | - | 11 |
| 22026 | | - | - | - | - | 11 | - | - | - | - | - | 10 |
| 8103 | 3 | | - | 10 | 12 | 10 | - | 11 | - | 14 | - | 9 |
| 2981 | 8 | | | 9 | 11 | 8 | 9 | 13 | - | - | - | 8 |
| 1097 | 4 | - | | | 10 | 9 | 11 | 8 | - | 14 | 8 | 7 |
| 403 | 4 | 5 | 8 | | | | 10 | 9 | 10 | - | - | 6 |
| 148 | 4 | - | 6 | | | | 9 | 9 | - | - | - | 5 |
| 55 | | 4 | 5 | - | - | | 10 | 9 | 10 | - | - | 4 |
| 20 | 3 | 4 | - | 4 | - | - | | 10 | - | 9 | - | 3 |
| 7 | - | | | | | 3 | | 9 | - | 9 | - | 2 |
| 3 | - | - | - | - | | 5 | - | | | | - | 1 |
| 1 | 3 | - | - | - | - | - | - | - | - | - | - | 0 |
| | 1 | 3 | 7 | 1 | 2 | 6 | 2 | 7 | 4 | 11 | 30 | |
| | Hours | | | Days | | | Weeks | | Months | | | |

FIGURE 1.—(A) Average empirical severity-of-ill-effect scores for juvenile and adult salmonids (freshwater, group 1) in the matrix of suspended sediment (*SS*) concentration and duration of exposure. Both matrix axes are expressed in logarithmic and absolute terms. Dashes mean "no data." Shaded bands denote inferred (by manual interpolation) thresholds of sublethal effects (shading without a border) and lethal effects (shading with a border; see Table 1 for criteria). (B, upper matrix) Severity-of-effect scores calculated by model (1) (Table 3). Severity-of-ill-effect calculations are based on the logarithmic values shown on the axes of the matrix. Shaded areas represent extrapolations beyond empirical data; extrapolations have been capped at 14 (upper limit of the effects scale: Table 1), although higher values are possible. Diagonal terraced lines denote thresholds of sublethal effects (lower left) and lethal effects (middle diagonal) delineated by the model with reference to Table 1. (B, lower matrix) Half-95% confidence intervals around calculated severity-of-effect scores. Shaded areas denote half-intervals greater than 1.0.

most serious effects to be measured (100% mortality; catastrophic habitat degradation).

### Data Presentation

*Empirical data.*—Severity-of-ill-effect values for each of the six data groups are presented as rounded averages in the cells of dose matrixes whose axes are concentration of suspended sediment and duration of exposure (panel A of the figure for each group). Maximum possible duration of exposure in the matrix is 48 months ($\log_e$[hours] = 10.4999). All but one of the matrixes show a maximum possible suspended sediment concentration of 268,337 mg/L ($\log_e$[mg SS/L] = 12.4999). The exception—adult estuarine fishes—has a maximum possible concentration of 729,416 mg SS/L ($\log_e$[mg SS/L] = 13.4999).

Displayed logarithmic values of duration and concentration are the midrange values. Thus the range of logarithmic values represented by a row or a column in the figures is approximately the value ± 0.4999 in logarithmic units (take antilogarithms for absolute values and their ranges). The accompanying confidence values are one-half the 95% confidence intervals around *z*.

Cells of a matrix that contain data form a cluster of "populated" cells. The imaginary "tight-string" polygon that encompasses all the populated cells in a matrix is the "data envelope." Typically, some cells within a data envelope are unpopulated. For predictive purposes, values are assigned to these cells by interpolation. Empty cells outside the envelope are given values by extrapolation. Interpolations are considered to have greater intrinsic reliability than extrapolations because they can be compared more easily with known data.



Juvenile and

Duration of expos

(B) Average severity-of-ill-

| Concentration (mg SS/L) | 0 | 1 | 2 | 3 | 4 |
|---|---|---|---|---|---|
| 162755 | 10 | 11 | 11 | 12 | [illegible] |
| 59874 | 9 | 10 | 10 | 11 | 12 |
| 22026 | 8 | 9 | 10 | 10 | 11 |
| 8103 | 8 | 8 | 9 | 10 | 10 |
| 2981 | 7 | 8 | 8 | 9 | |
| 1097 | 6 | 7 | 7 | 8 | |
| 403 | 5 | 6 | 7 | 7 | 8 |
| 148 | 5 | 5 | 6 | 7 | 7 |
| 55 | 4 | 5 | 5 | 6 | 6 |
| 20 | 3 | 4 | 4 | 5 | 6 |
| 7 | 3 | 3 | 4 | 4 | 5 |
| 3 | 2 | 2 | 3 | 4 | 4 |
| 1 | 1 | | | | |
| | 1 | 3 | 7 | 1 | 2 |
| | Hours | | | Days | |

Half-95% confide

around calculated severity-

| | | | | | | |
|---|---|---|---|---|---|---|
| 162755 | 0.9 | 0.8 | 0.8 | 0.8 | 0.7 | 0.7 |
| 59874 | 0.8 | 0.7 | 0.7 | 0.6 | 0.6 | 0.6 |
| 22026 | 0.7 | 0.6 | 0.6 | 0.5 | 0.5 | 0.5 |
| 8103 | 0.6 | 0.6 | 0.5 | 0.4 | 0.4 | 0.4 |
| 2981 | 0.6 | 0.5 | 0.4 | 0.4 | 0.4 | 0.4 |
| 1097 | 0.6 | 0.5 | 0.4 | 0.3 | 0.3 | 0.3 |
| 403 | 0.6 | 0.5 | 0.4 | 0.3 | 0.3 | 0.3 |
| 148 | 0.6 | 0.5 | 0.4 | 0.4 | 0.4 | 0.4 |
| 55 | 0.6 | 0.6 | 0.5 | 0.5 | 0.5 | 0.5 |
| 20 | 0.7 | 0.7 | 0.6 | 0.6 | 0.6 | 0.6 |
| 7 | 0.8 | 0.7 | 0.7 | 0.7 | 0.7 | 0.7 |
| 3 | 0.9 | 0.8 | 0.8 | 0.8 | 0.8 | 0.8 |
| 1 | 1.0 | 1.0 | 0.9 | 0.9 | 0.9 | 0.9 |
| | 1 | 3 | 7 | 1 | 2 | 6 |
| | Hours | | | Days | | |

FIGURE 1.—Co

*Thresholds of ill effect.*—Display of empiri- severity-of-effect scores in the dose matrix perm estimation of the minimum concentrations and durations that trigger sublethal and lethal effects (panel A of the figure for each group). For this purpose, unpopulated cells within the data envelope are assigned values by manual interpolation. Thresholds thus estimated from empirical data of-

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Juvenile and Adult Salmonids

Duration of exposure to SS ($\log_e$ hours)

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

| Concentration (mg SS/L) | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | ($\log_e$ mg SS/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 162755 | 10 | 11 | 11 | 12 | 12 | 13 | 14 | 14 | [illegible] | [illegible] | [illegible] | 12 |
| 59874 | 9 | 10 | 10 | 11 | 12 | 12 | 13 | 13 | 14 | [illegible] | [illegible] | 11 |
| 22026 | 8 | 9 | 10 | 10 | 11 | 11 | 12 | 13 | 13 | 14 | [illegible] | 10 |
| 8103 | 8 | 8 | 9 | 10 | 10 | 11 | 11 | 12 | 13 | 13 | 14 | 9 |
| 2981 | 7 | 8 | 8 | 9 | 9 | 10 | 11 | 11 | 12 | 12 | 13 | 8 |
| 1097 | 6 | 7 | 7 | 8 | 9 | 9 | 10 | 10 | 11 | 12 | 12 | 7 |
| 403 | 5 | 6 | 7 | 7 | 8 | 9 | 9 | 10 | 10 | 11 | 12 | 6 |
| 148 | 5 | 5 | 6 | 7 | 7 | 8 | 8 | 9 | 10 | 10 | 11 | 5 |
| 55 | 4 | 5 | 5 | 6 | 6 | 7 | 8 | 8 | 9 | 9 | 10 | 4 |
| 20 | 3 | 4 | 4 | 5 | 6 | 6 | 7 | 8 | 8 | 9 | 9 | 3 |
| 7 | 3 | 3 | 4 | 4 | 5 | 6 | 6 | 7 | 7 | 8 | 9 | 2 |
| 3 | 2 | 2 | 3 | 4 | 4 | 5 | 5 | 6 | 7 | 7 | 8 | 1 |
| 1 | 1 | 2 | 2 | 3 | 3 | 4 | 5 | 5 | 6 | 7 | 7 | 0 |
| | 1 | 3 | 7 | 1 | 2 | 6 | 2 | 7 | 4 | 11 | 30 | |
| | Hours | | | Days | | | Weeks | | Months | | | |

Half-95% confidence intervals (±)
around calculated severity-of-ill-effect scores (above)

| Concentration (mg SS/L) | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | ($\log_e$ mg SS/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 162755 | 0.9 | 0.8 | 0.8 | 0.8 | 0.7 | 0.7 | 0.8 | 0.8 | - | - | - | 12 |
| 59874 | 0.8 | 0.7 | 0.7 | 0.6 | 0.6 | 0.6 | 0.6 | 0.7 | 0.7 | - | - | 11 |
| 22026 | 0.7 | 0.6 | 0.6 | 0.5 | 0.5 | 0.5 | 0.6 | 0.6 | 0.7 | 0.7 | - | 10 |
| 8103 | 0.6 | 0.6 | 0.5 | 0.4 | 0.4 | 0.4 | 0.5 | 0.5 | 0.6 | 0.7 | 0.8 | 9 |
| 2981 | 0.6 | 0.5 | 0.4 | 0.4 | 0.4 | 0.4 | 0.4 | 0.5 | 0.5 | 0.6 | 0.7 | 8 |
| 1097 | 0.6 | 0.5 | 0.4 | 0.3 | 0.3 | 0.3 | 0.4 | 0.4 | 0.5 | 0.6 | 0.7 | 7 |
| 403 | 0.6 | 0.5 | 0.4 | 0.3 | 0.3 | 0.3 | 0.4 | 0.5 | 0.5 | 0.6 | 0.7 | 6 |
| 148 | 0.6 | 0.5 | 0.4 | 0.4 | 0.4 | 0.4 | 0.4 | 0.5 | 0.6 | 0.7 | 0.8 | 5 |
| 55 | 0.6 | 0.6 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.6 | 0.6 | 0.7 | 0.8 | 4 |
| 20 | 0.7 | 0.7 | 0.6 | 0.6 | 0.6 | 0.6 | 0.6 | 0.7 | 0.7 | 0.8 | 0.9 | 3 |
| 7 | 0.8 | 0.7 | 0.7 | 0.7 | 0.7 | 0.7 | 0.7 | 0.8 | 0.8 | 0.9 | 1.0 | 2 |
| 3 | 0.9 | 0.8 | 0.8 | 0.8 | 0.8 | 0.8 | 0.8 | 0.9 | 0.9 | 1.0 | 1.0 | 1 |
| 1 | 1.0 | 1.0 | 0.9 | 0.9 | 0.9 | 0.9 | 0.9 | 1.0 | 1.0 | 1.1 | 1.1 | 0 |
| | 1 | 3 | 7 | 1 | 2 | 6 | 2 | 7 | 4 | 11 | 30 | |
| | Hours | | | Days | | | Weeks | | Months | | | |

FIGURE 1.—Continued.

*Thresholds of ill effect.*—Display of empirical severity-of-effect scores in the dose matrix permits estimation of the minimum concentrations and durations that trigger sublethal and lethal effects (panel A of the figure for each group). For this purpose, unpopulated cells within the data envelope are assigned values by manual interpolation. Thresholds thus estimated from empirical data often are lower than thresholds predicted by regressions fit to meta-analytical data. We interpret "empirical thresholds" as an approximated response of the more "sensitive" individuals within a species group.

*Predictions of ill effect.*—The regression equation fitted to each of the six data groups provides predictions of response within the matrix of con-

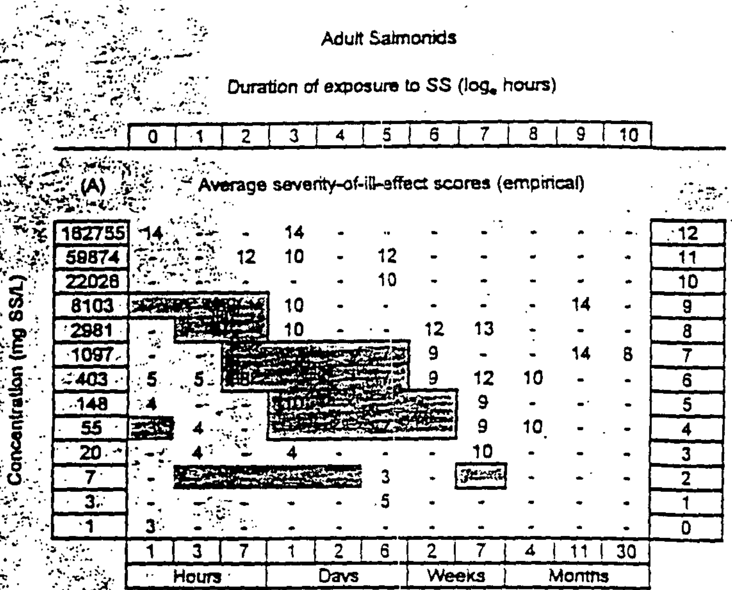


FIGURE 2.—Empirical severity-of-ill-effect scores for adult salmonids (freshwater, group 2) and scores (with half-95% confidence intervals) predicted by model (2). Conventions are those of Figure 1.

centration and duration of exposure (panel B of the figure for each group). Each prediction is accompanied by half-95% confidence intervals.

Each prediction matrix is divided into a maximum of three zones by terraced lines separating behavioral, sublethal, and lethal responses. We compare these modeled thresholds to empirical ones to discern responses of "sensitive" individuals within each species group.

## Results

Dose–response models fitted to the empirical data groups were all highly significant ($P < 0.01$) and accounted for 55–70% of the variances (Table 3). Averaged empirical data on which the models are based are displayed in panel A of Figures 1–6. Panel B of Figures 1–6 gives the model-generated responses (and confidence intervals) for each cell of the dose–response matrixes. These panels provide a set of "look-up tables" suitable for field use in impact assessment. Superimposed on them are predicted thresholds of sublethal and lethal effects based on the response categories in Table 1. Response surfaces resulting from the models are shown in Figures 7–12. Data are derived from sources listed in the Appendix.

*Group 1: Juvenile and Adult Salmonids*

Average empirical severity-of-ill-effect data for group 1 fill 56 of the 143 available cells (Figure 1A). Data are widely distributed, but thresholds for the onset of sublethal and lethal ill effects can be inferred within broad limits, based on manual interpolations within the data envelope (see gray-shaded zones without and with borders).

The full matrix array of severity scores predicted by model 1 (Table 3, Figure 1B) shows regular increases of response intensity with sediment dose, as expected. Predicted thresholds of sublethal and lethal effects (terraced diagonals) have similar orientations to those inferred from empirical data, but they generally occur at higher sediment doses.

*Group 2: Adult Salmonids*

Group 2 data fill 36 widely scattered cells of the 143 available in the empirical matrix (Figure 2A). The thresholds of lethal effect predicted by model 2 (Table 3; Figure 2B) are similar to the empirically inferred threshold (Figure 2A), but predicted sublethal effects emerge at slightly lower sediment doses than implied by empirical data.



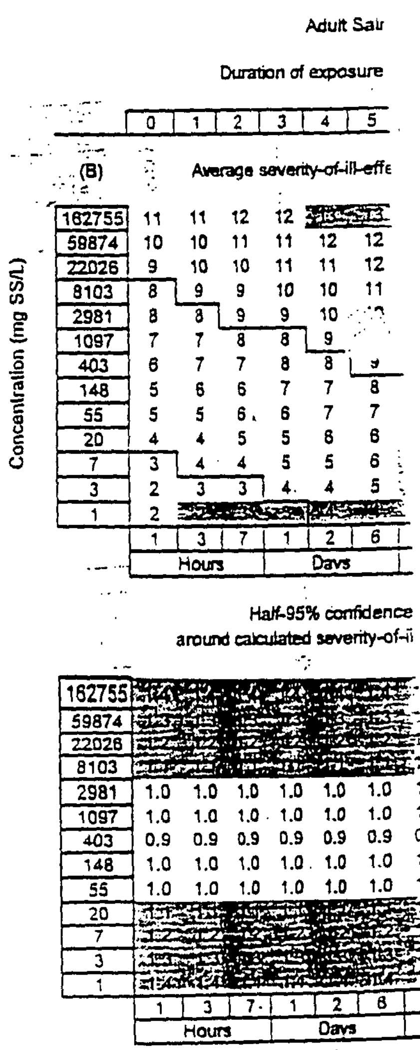


FIGURE 2.—Conti

*Group 3: Juvenile Salmonids*

Average severity-of-effect scores for group 3 fill 37 cells, most of them clustered at exposure durations of 1 h and 2 d to 7 weeks (Figure 3A). As for adult salmonids, predicted thresholds (model 3: Table 3; Figure 3B) were similar to empirical thresholds for lethal effects but lower than empirical ones for sublethal effects.

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Adult Salmonids

Duration of exposure to SS ($\log_e$ hours)

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

| Concentration (mg SS/L) | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | ($\log_e$ mg SS/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 162755 | 11 | 11 | 12 | 12 | 13 | 13 | 14 | 14 | [illegible] | [illegible] | [illegible] | 12 |
| 59874 | 10 | 10 | 11 | 11 | 12 | 12 | 13 | 13 | 14 | 14 | [illegible] | 11 |
| 22026 | 9 | 10 | 10 | 11 | 11 | 12 | 12 | 13 | 13 | 14 | 14 | 10 |
| 8103 | 8 | 9 | 9 | 10 | 10 | 11 | 11 | 12 | 12 | 13 | 13 | 9 |
| 2981 | 8 | 8 | 9 | 9 | 10 | 10 | 11 | 11 | 12 | 12 | 13 | 8 |
| 1097 | 7 | 7 | 8 | 8 | 9 | 9 | 10 | 10 | 11 | 11 | 12 | 7 |
| 403 | 6 | 7 | 7 | 8 | 8 | 9 | 9 | 10 | 10 | 11 | 11 | 6 |
| 148 | 5 | 6 | 6 | 7 | 7 | 8 | 8 | 9 | 9 | 10 | 10 | 5 |
| 55 | 5 | 5 | 6 | 6 | 7 | 7 | 8 | 8 | 9 | 10 | 9 | 4 |
| 20 | 4 | 4 | 5 | 5 | 6 | 6 | 7 | 7 | 8 | 8 | 9 | 3 |
| 7 | 3 | 4 | 4 | 5 | 5 | 6 | 6 | 7 | 7 | 7 | 8 | 2 |
| 3 | 2 | 3 | 3 | 4 | 4 | 5 | 5 | 6 | 6 | 7 | 7 | 1 |
| 1 | 2 | 2 | 3 | 3 | 4 | 4 | 5 | 5 | 5 | 6 | 6 | 0 |
| | 1 | 3 | 7 | 1 | 2 | 6 | 2 | 7 | 4 | 11 | 30 | |
| | Hours | | | Days | | | Weeks | | Months | | | |

Half-95% confidence intervals (±)
around calculated severity-of-ill-effect scores (above)

| Concentration (mg SS/L) | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | ($\log_e$ mg SS/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 162755 | 1.4 | 1.4 | 1.4 | 1.4 | 1.4 | 1.4 | 1.4 | 1.4 | [illegible] | [illegible] | [illegible] | 12 |
| 59874 | 1.3 | 1.3 | 1.3 | 1.3 | 1.3 | 1.3 | 1.3 | 1.3 | 1.3 | 1.3 | [illegible] | 11 |
| 22026 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.3 | 10 |
| 8103 | [illegible] | 1.1 | 1.1 | [illegible] | [illegible] | 1.0 | 1.1 | 1.1 | 1.1 | 1.1 | 1.2 | 9 |
| 2981 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.1 | 8 |
| 1097 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.1 | 7 |
| 403 | 0.9 | 0.9 | 0.9 | 0.9 | 0.9 | 0.9 | 0.9 | 0.9 | 0.9 | 0.9 | 1.0 | 6 |
| 148 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.1 | 5 |
| 55 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.1 | 4 |
| 20 | 1.1 | 1.1 | 1.1 | 1.1 | 1.1 | 1.1 | 1.1 | 1.1 | 1.1 | 1.1 | 1.2 | 3 |
| 7 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.3 | 2 |
| 3 | 1.3 | 1.3 | 1.3 | 1.3 | 1.3 | 1.3 | 1.3 | 1.3 | 1.3 | 1.3 | 1.4 | 1 |
| 1 | 1.4 | 1.4 | 1.4 | 1.4 | 1.4 | 1.4 | 1.4 | 1.4 | 1.4 | 1.4 | 1.5 | 0 |
| | 1 | 3 | 7 | 1 | 2 | 6 | 2 | 7 | 4 | 11 | 30 | |
| | Hours | | | Days | | | Weeks | | Months | | | |

FIGURE 2.—Continued.

*Group 3: Juvenile Salmonids*

Average severity-of-effect scores for group 3 fill 37 cells, most of them clustered at exposure durations of 1 h and 2 d to 7 weeks (Figure 3A). As for adult salmonids, predicted thresholds (model 3; Table 3; Figure 3B) were similar to empirical thresholds for lethal effects but lower than empirical ones for sublethal effects.

*Group 4: Eggs and Larvae of Salmonids and Nonsalmonids*

Average severity scores for eggs and larvae of salmonids and freshwater and estuarine nonsalmonids fill 23 cells (Figure 4A). Most data are clustered in the exposure interval of 1 d to 7 weeks. Sublethal effects thresholds were estimated empirically, but they were not recognized by model

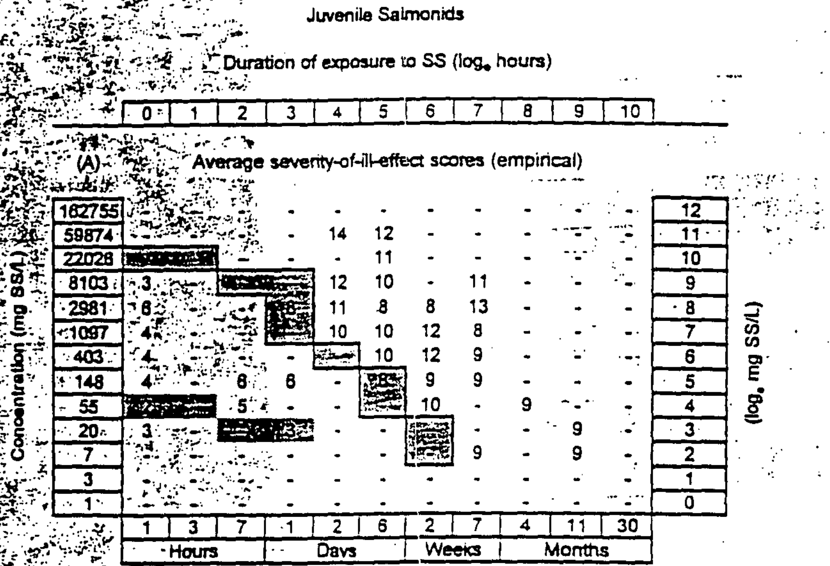


FIGURE 3.—Empirical severity-of-ill-effect scores for juvenile salmonids (freshwater, group 3) and scores (with half-95% confidence intervals) predicted by model (3). Conventions are those of Figure 1.

4 (Table 3; Figure 4B), which generated no severity score lower than 4. Empirical and predicted thresholds of lethal effect agreed well and occurred at relatively low doses.

*Group 5: Adult Estuarine Nonsalmonids*

Average severity-of-effect scores for at least 15 species of estuarine fishes filled 23 of the available 154 matrix cells (Figure 5A). Most of the data represent 1–6-d exposures.

Model 5 (Table 3) was developed for only the seven species represented by adequate data. These seven are believed to be relatively more sensitive to the ill effects of suspended sediment than the other species in the database (Table 2). Predicted thresholds of lethal effect (Figure 5B) tracked empirical thresholds well for exposure durations less than 1 d; both estimates indicated that lethal effects on those sensitive species result from short exposures to a wide range of sediment concentrations. Sublethal effect thresholds were considerably closer the origin in the predictive matrix than in the empirical matrix.

*Group 6: Adult Freshwater Nonsalmonids*

A relatively small sample of stream and stillwater fishes in cold, temperate, and warmwater environments provided average severity scores for 15 scattered matrix cells of the 143 available (Figure 6A). Model 6 (Table 3) generated lethal effects thresholds that agreed well with interpolations of empirical data for exposures of 7 d to 7 weeks (Figure 6B). Although sublethal thresholds could be inferred from empirical data, the model indicated that they lay beyond the matrix—below concentrations of 1 mg/L, exposure durations of 1 h, or both.

*Response Surfaces*

Dose–response surfaces based on models 1–6 are shown in Figures 7–12. We think it important to emphasize that only models (1), (3), and (4) address early life stages in some form. Many studies have shown that early stages (some stages of egg development through young juveniles) are more susceptible to toxicants and other pollutants than older juveniles and adults. The response surfaces (and prediction matrixes) should be judged by the data available to develop them.

## Discussion

Fisheries biologists, habitat protection specialists, and enforcement officers in many parts of the world may find that the dose–response equations



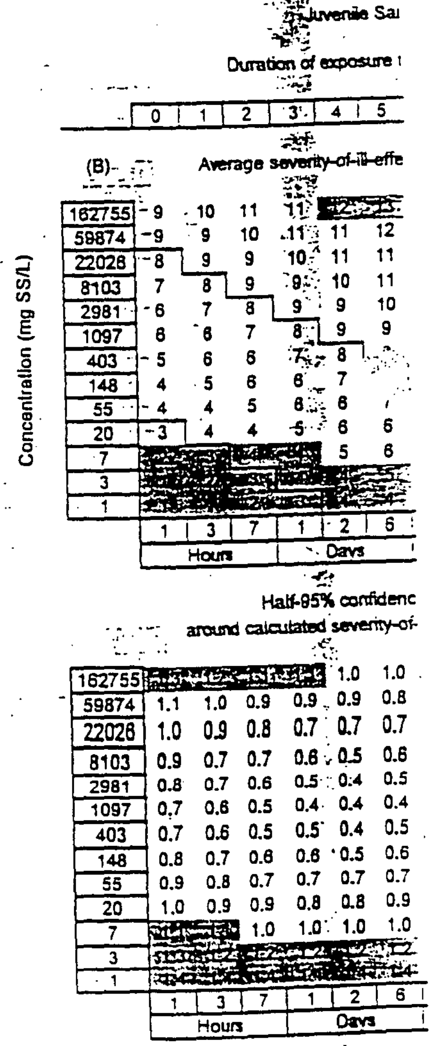


FIGURE 3.—Con

generated in this study are useful additions to their daily work. The discussion below focuses on (i) validation of the models, (ii) the dose–response patterns of ultrasensitive species and life stages, (iii) potential new options in environmental law enforcement, (iv) the role of meta-analysis in the findings of this study, (v) possible directions of

Juvenile Salmonids

Duration of exposure to SS ($\log_e$ hours)

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

| Concentration (mg SS/L) | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | ($\log_e$ mg SS/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 162755 | 9 | 10 | 11 | 11 | 12 | 13 | 14 | 14 | [illegible] | [illegible] | [illegible] | 12 |
| 59874 | 9 | 9 | 10 | 11 | 11 | 12 | 13 | 14 | 14 | [illegible] | [illegible] | 11 |
| 22026 | 8 | 9 | 9 | 10 | 11 | 11 | 12 | 13 | 13 | 14 | [illegible] | 10 |
| 8103 | 7 | 8 | 9 | 9 | 10 | 11 | 11 | 12 | 13 | 13 | 14 | 9 |
| 2981 | 6 | 7 | 8 | 9 | 9 | 10 | 11 | 11 | 12 | 13 | 13 | 8 |
| 1097 | 6 | 6 | 7 | 8 | 9 | 9 | 10 | 11 | 11 | 12 | 13 | 7 |
| 403 | 5 | 6 | 6 | 7 | 8 | 9 | 9 | 10 | 11 | 11 | 12 | 6 |
| 148 | 4 | 5 | 6 | 6 | 7 | 8 | 9 | 9 | 10 | 11 | 11 | 5 |
| 55 | 4 | 4 | 5 | 6 | 6 | 7 | 8 | 8 | 9 | 10 | 11 | 4 |
| 20 | 3 | 4 | 4 | 5 | 6 | 6 | 7 | 8 | 8 | 9 | 10 | 3 |
| 7 | 2 | 3 | 4 | 4 | 5 | 6 | 6 | 7 | 8 | 8 | 9 | 2 |
| 3 | 1 | 2 | 3 | 4 | 4 | 5 | 6 | 6 | 7 | 8 | 8 | 1 |
| 1 | 1 | 1 | 2 | 3 | 4 | [illegible] | 5 | 6 | 6 | 7 | 8 | 0 |
| | 1 Hours | 3 Hours | 7 Hours | 1 Days | 2 Days | 6 Days | 2 Weeks | 7 Weeks | 4 Months | 11 Months | 30 Months | |

Half-95% confidence intervals (±) around calculated severity-of-ill-effect scores (above)

| Concentration (mg SS/L) | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | ($\log_e$ mg SS/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 162755 | 1.3 | 1.2 | 1.1 | 1.1 | 1.0 | 1.0 | 1.0 | 1.1 | [illegible] | [illegible] | [illegible] | 12 |
| 59874 | 1.1 | 1.0 | 0.9 | 0.9 | 0.9 | 0.8 | 0.9 | 0.9 | 1.0 | [illegible] | [illegible] | 11 |
| 22026 | 1.0 | 0.9 | 0.8 | 0.7 | 0.7 | 0.7 | 0.7 | 0.8 | 0.9 | 1.0 | [illegible] | 10 |
| 8103 | 0.9 | 0.7 | 0.7 | 0.6 | 0.5 | 0.6 | 0.6 | 0.7 | 0.8 | 0.9 | 1.0 | 9 |
| 2981 | 0.8 | 0.7 | 0.6 | 0.5 | 0.4 | 0.5 | 0.5 | 0.6 | 0.7 | 0.8 | 1.0 | 8 |
| 1097 | 0.7 | 0.6 | 0.5 | 0.4 | 0.4 | 0.4 | 0.5 | 0.6 | 0.7 | 0.8 | 1.0 | 7 |
| 403 | 0.7 | 0.6 | 0.5 | 0.5 | 0.4 | 0.5 | 0.5 | 0.6 | 0.7 | 0.9 | 1.0 | 6 |
| 148 | 0.8 | 0.7 | 0.6 | 0.6 | 0.5 | 0.6 | 0.6 | 0.7 | 0.8 | 1.0 | [illegible] | 5 |
| 55 | 0.9 | 0.8 | 0.7 | 0.7 | 0.7 | 0.7 | 0.8 | 0.9 | 0.9 | [illegible] | 1.2 | 4 |
| 20 | 1.0 | 0.9 | 0.9 | 0.8 | 0.8 | 0.9 | 0.9 | 1.0 | 1.1 | 1.2 | 1.3 | 3 |
| 7 | 1.1 | 1.1 | 1.0 | 1.0 | 1.0 | 1.0 | 1.1 | 1.2 | 1.2 | 1.3 | 1.4 | 2 |
| 3 | 1.3 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.3 | 1.3 | 1.4 | 1.5 | 1.6 | 1 |
| 1 | 1.4 | 1.4 | 1.4 | 1.4 | 1.4 | 1.4 | 1.4 | 1.5 | 1.6 | 1.6 | 1.7 | 0 |
| | 1 Hours | 3 Hours | 7 Hours | 1 Days | 2 Days | 6 Days | 2 Weeks | 7 Weeks | 4 Months | 11 Months | 30 Months | |

FIGURE 3.—Continued.

generated in this study are useful additions to their daily work. The discussion below focuses on (i) validation of the models, (ii) the dose–response patterns of ultrasensitive species and life stages, (iii) potential new options in environmental law enforcement, (iv) the role of meta-analysis in the findings of this study, (v) possible directions of future research, and (vi) implications of this study for ecosystem assessment.

### *Validation of the Models*

Validation of the models in this study will rely on new studies that add to the data now available. Creation of new data—in sufficient volume for

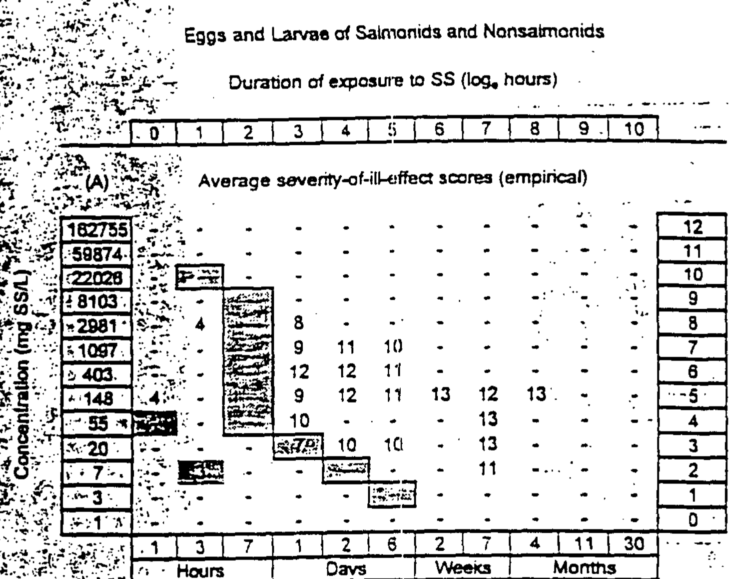


FIGURE 4.—Empirical severity-of-ill-effect scores for eggs and larvae of salmonids and nonsalmonids (freshwater and estuarine, group 4) and scores (with half-95% confidence intervals) predicted by model (4). Conventions are those of Figure 1, except the model (B, upper matrix) recognized no threshold of sublethal effects.

testing and refinement of these models—is bound to be a slow process. However, in the brief time since the conclusion of the data-gathering phase of this study, some new data have emerged.

First, coho salmon fry (mean weight, 1.95 g; $N$ = 10 fish), when exposed to suspended sediment at a concentration of 5,471 mg SS/L for 96 h, sustained a mortality rate of 10% after they had been held in water at 18.7°C and 9.7 mg $O_2$/L (J.O.T.J., unpublished data). This mortality rate expressed as a severity of ill effect (with reference to Table 1) is SEV = 10. Severity of ill effect as predicted by model 1, (SEV = 0.7262 − 0.7034log$_e$[96 h] + 0.7144(log$_e$5,471 mg SS/L]) is 10.09. These values agree closely and tend to validate this model. Steelhead ($N$ = 10), similarly exposed, had 0% mortality. This result too is consistent with the predictions of the model, because SEV = 10 represents 0–20% mortality, and the test fish exhibited behaviors of severe sublethal stress.

Second, a recent laboratory study of effects of suspended bentonite clay (1–5-μm diameters) on larval nonsalmonid fishes (smallmouth bass, largemouth bass, and bluegill) in warm water (20–25°C) has produced several sets of morbidity data (reduced growth rate) and mortality data that are highly consistent with the predictions of model (4) (J. Sweeten, Asherwood Environmental Learning Centre, personal communication).

Third, an inverse relationship has been documented between sediment concentrations in streams and maximum salmonid densities in fluvial habitats in British Columbia (Ptolemy 1993; R. A. Ptolemy, British Columbia Ministry of Environment, Lands and Parks, personal communication). For example, the density (number of fish per unit area) of juvenile chinook salmon and steelhead that rear in the turbid main stem of the Bella Coola River (British Columbia) is lower than would be expected in clear water. Rearing occurs in June, July, and August. During this time, turbidity averages 21 nephelometric units, suspended sediment concentration averages 61 mg SS/L, particle sizes are smaller than 75 μm, and the temperature range is 8–12°C). Reduced fish density is consistent with the range of ill effects—low paralethal rankings—predicted by the models. These results tacitly acknowledge the role of excess sediment exposure—particularly concentration and duration—as a factor in the productivity of salmon streams. Two extenuating factors—relatively



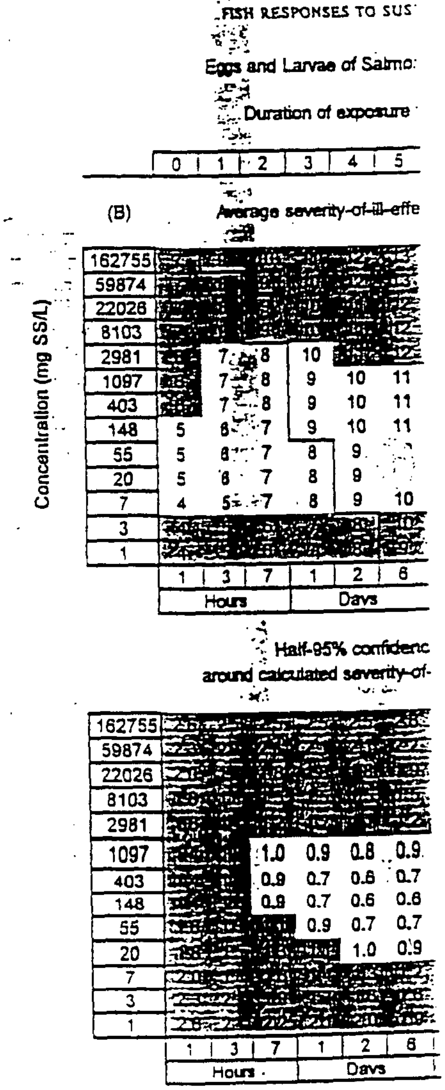


FIGURE 4.—Cor

small particle size and relatively cool water—could explain the absence of direct lethality in the Bella Coola.

Fourth, juvenile salmonids (chinook salmon, rainbow trout, and mountain whitefish) are thoug[illegible] to seek refuge—an average of 9 d for age-0 w[illegible] chinook salmon—in a small nonnatal tributary o[illegible] the upper Fraser River, perhaps to avoid unsuitable

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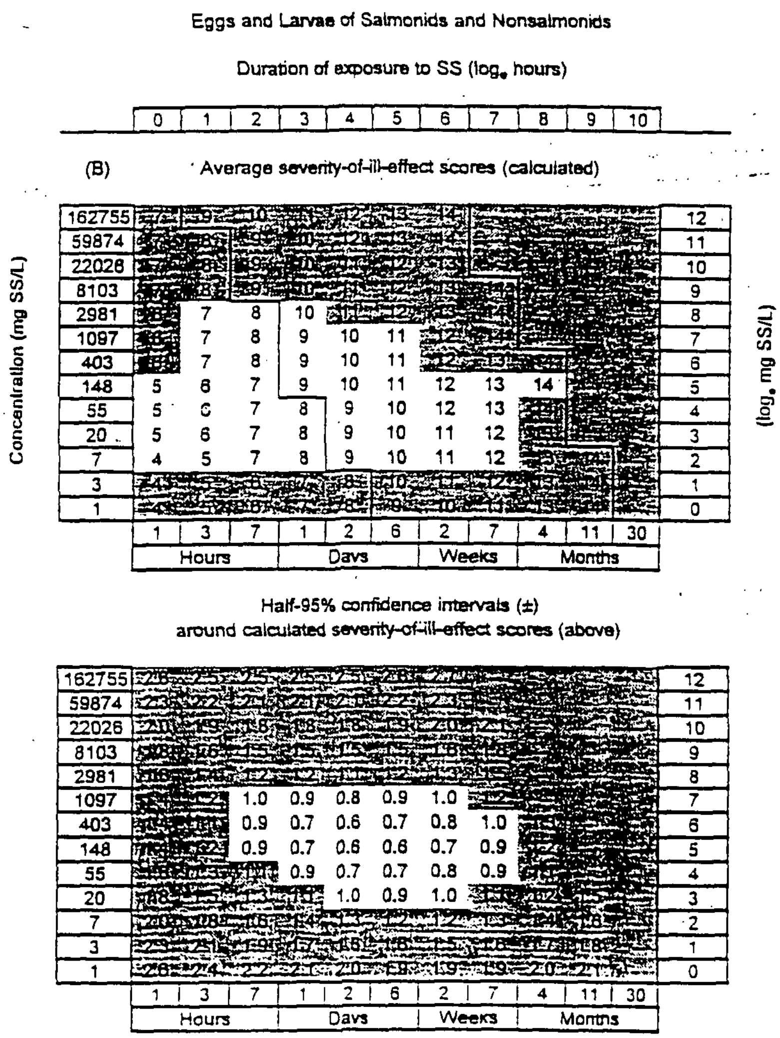


FIGURE 4.—Continued.

small particle size and relatively cool water—could explain the absence of direct lethality in the Bella Coola.

Fourth, juvenile salmonids (chinook salmon, rainbow trout, and mountain whitefish) are thought to seek refuge—an average of 9 d for age-0 wild chinook salmon—in a small nonnatal tributary of the upper Fraser River, perhaps to avoid unsuitable rearing conditions created by high, naturally occurring sediment loads found in the main stem (Scrivener et al. 1993).

Although these recent findings tend to support the predictions of the models, the well-documented good health (as indicated by acceptable rates of growth and survival) among salmon juveniles in turbid estuarine waters remains unexplained.

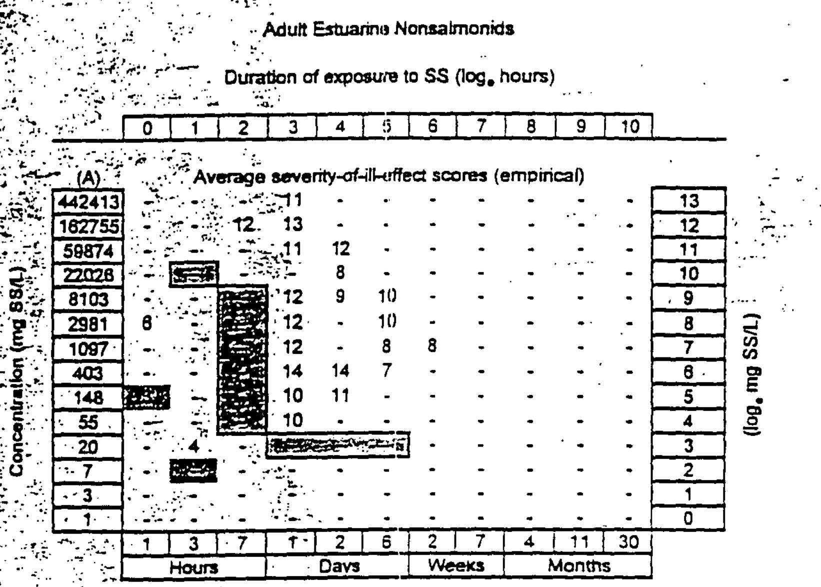


FIGURE 5.—Empirical severity-of-ill-effect scores for adult nonsalmonids (estuarine, group 5) and scores (with half-95% confidence intervals) predicted by model (5). Conventions are those of Figure 1.

Considerations relevant to this "anomaly" include (i) the extremely fine texture of suspended sediment (generally much smaller than 75 μm); (ii) the relatively cold water temperatures; (iii) the potential for favorable physicochemical effects such as flocculation, which could be enhanced by the chemistry of brackish water; (iv) beneficial behavioral adaptations of juvenile salmonids; and (v) the suitability of reedy habitat, where average sediment concentrations and average particle size may be further reduced below those found in traditional sampling sites.

### *Ultrasensitivity of Some Species and Life Stages*

Rapid escalation of ill effects on eggs, larvae, and fry (Figures 4, 10) and on some adult fishes of the estuary (Figures 5, 11) as duration of sediment exposure increases suggests that the mechanisms of self-preservation in at least some estuarine fishes are easily overwhelmed by the presence of suspended sediment. This pattern implies the existence of an abrupt threshold concentration of suspended sediment leading to ill effects in ultrasensitive species and life stages.

If this inference is correct, these dose–response patterns might be explained in terms of the time required to reach an end point (e.g., lethality), and might indicate that the physiological and physical processes involved in homeostasis are more sensitive to exposure time than to suspended sediment concentrations. It is reasonable to speculate further that the sequence of events leading to a lethal end point (for example, severely abraded gill tissue and associated loss of capacity for ion regulation), once triggered, would not easily be halted or reversed.

### *Environmental Enforcement Issues*

Fisheries biologists and enforcement personnel can, as part of an investigation, document the sediment concentration and duration of exposure, and they can use these data to infer the most probable severity of impact. The dose–response equations alone are sufficient for this task. But the "look-up" tables (here, Figures 1–6, panels B) simplify the task even more; they are based on the equations, and they supply ranges of interpolation and extrapolation and confidence intervals. They make it possible for field workers readily to distinguish between minor and major events in the broad context established by the dose–response matrixes. This knowledge can contribute to decisions about



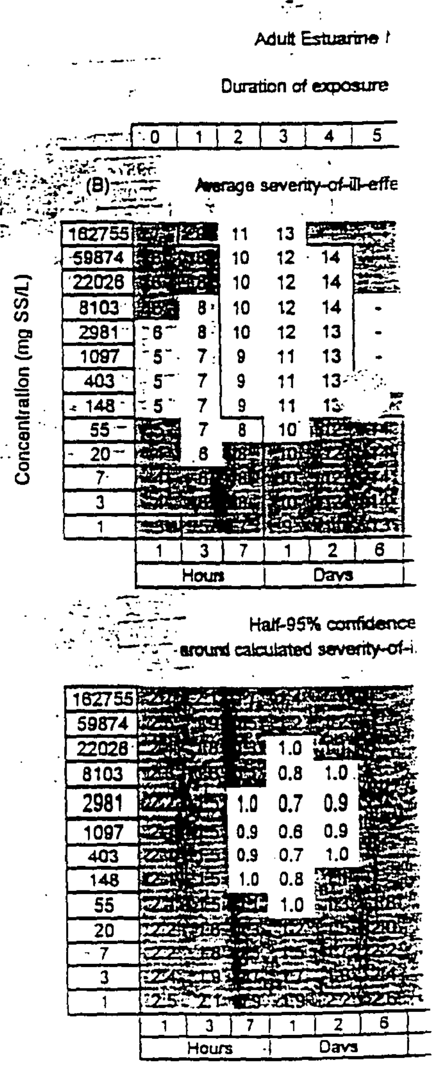


FIGURE 5.—Contin

the need for additional field work by which to gather physical evidence about the nature and severity of the ill effects. This new capacity to make inferences—an unprecedented development in the field of channel sediment impacts—might also influence the goals of a prosecution.

Impacts on fish populations exposed to episodes of excess sediment may vary according to the cir-

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Adult Estuarine Nonsalmonids

Duration of exposure to SS ($\log_e$ hours)

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

| Concentration (mg SS/L) | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | ($\log_e$ mg SS/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 162755 | 7 | 9 | 11 | 13 | | | | | | | | 12 |
| 59874 | 6 | 8 | 10 | 12 | 14 | | | | | | | 11 |
| 22026 | 6 | 8 | 10 | 12 | 14 | | | | | | | 10 |
| 8103 | 6 | 8 | 10 | 12 | 14 | - | | | | | | 9 |
| 2981 | 6 | 8 | 10 | 12 | 13 | - | | | | | | 8 |
| 1097 | 5 | 7 | 9 | 11 | 13 | - | - | | | | | 7 |
| 403 | 5 | 7 | 9 | 11 | 13 | - | | | | | | 6 |
| 148 | 5 | 7 | 9 | 11 | 13 | | | | | | | 5 |
| 55 | 5 | 7 | 8 | 10 | 12 | 14 | | | | | | 4 |
| 20 | 4 | 6 | 8 | 10 | 12 | 14 | | | | | | 3 |
| 7 | 4 | 6 | 8 | 10 | 12 | 14 | | | | | | 2 |
| 3 | 4 | 6 | 8 | 10 | 12 | 14 | | | | | | 1 |
| 1 | 3 | 5 | 7 | 9 | 11 | 13 | | | | | | 0 |
| | 1 | 3 | 7 | 1 | 2 | 6 | 2 | 7 | 4 | 11 | 30 | |
| | Hours | | | Days | | | Weeks | | Months | | | |

Half-95% confidence intervals (±) around calculated severity-of-ill-effect scores (above)

| Concentration (mg SS/L) | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | ($\log_e$ mg SS/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 162755 | 2.7 | 2.1 | 1.7 | 1.4 | | | | | | | | 12 |
| 59874 | 2.5 | 1.9 | 1.5 | 1.2 | 1.2 | | | | | | | 11 |
| 22026 | 2.4 | 1.8 | 1.3 | 1.0 | 1.1 | | | | | | | 10 |
| 8103 | 2.3 | 1.8 | 1.1 | 0.8 | 1.0 | | | | | | | 9 |
| 2981 | 2.2 | 1.5 | 1.0 | 0.7 | 0.9 | | | | | | | 8 |
| 1097 | 2.1 | 1.5 | 0.9 | 0.6 | 0.9 | | | | | | | 7 |
| 403 | 2.1 | 1.5 | 0.9 | 0.7 | 1.0 | | | | | | | 6 |
| 148 | 2.1 | 1.5 | 1.0 | 0.8 | 1.1 | | | | | | | 5 |
| 55 | 2.1 | 1.5 | 1.3 | 1.0 | 1.3 | 1.8 | | | | | | 4 |
| 20 | 2.2 | 1.8 | 1.3 | 1.2 | 1.5 | 2.0 | | | | | | 3 |
| 7 | 2.2 | 1.8 | 1.5 | 1.5 | 1.7 | 2.2 | | | | | | 2 |
| 3 | 2.4 | 1.9 | 1.7 | 1.7 | 1.9 | 2.4 | | | | | | 1 |
| 1 | 2.5 | 2.1 | 1.9 | 1.9 | 2.2 | 2.6 | | | | | | 0 |
| | 1 | 3 | 7 | 1 | 2 | 6 | 2 | 7 | 4 | 11 | 30 | |
| | Hours | | | Days | | | Weeks | | Months | | | |

FIGURE 5.—Continued.

the need for additional field work by which to gather physical evidence about the nature and severity of the ill effects. This new capacity to make inferences—an unprecedented development in the field of channel sediment impacts—might also influence the goals of a prosecution.

Impacts on fish populations exposed to episodes of excess sediment may vary according to the circumstances of the event. For example, fish tend to avoid high concentrations of suspended sediment when possible. Thus, a pollution episode capable of causing high mortality (e.g., of sac fry) or gill damage or starvation or slowed maturation (e.g., of age-0 fingerlings and age-2 juveniles) among caged fish (Reynolds et al. 1989) might not cause any of these direct effects in a wild population that

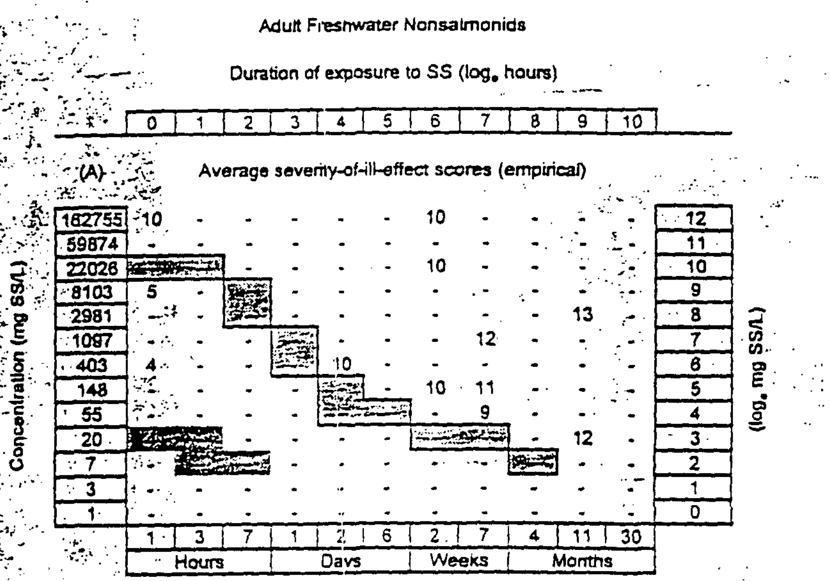


FIGURE 6.—Empirical severity-of-ill-effect scores for adult nonsalmonids (freshwater, group 6) and scores (with half-95% confidence intervals) predicted by model (6). Conventions are those of Figure 1, except the model (B, upper matrix) recognized no threshold of sublethal effects.

is free to move elsewhere in the stream system. Absence of dead fish (notwithstanding reduced egg-to-fry survival) is, however, not necessarily an indication of absence of harm. Indirect effects of sedimentation—loss of summer habitat for feeding and reproduction—may outweigh the direct effects seen in caged fish (Reynolds et al. 1989). This dichotomy has practical implications for enforcement. An investigation during a pollution event should attempt to document suspended sediment concentrations and durations for possible use with the models given here.

However, in the aftermath of a sediment pollution event, the investigation should switch its focus and gather evidence of sediment deposition. Changes in streambed composition resulting from excess sediment are usually manifested as changes in particle size composition. Subjective methods for assessing the extent of sedimentation exist. Objective methods are being developed (Kondolf and Li 1992; Kondolf and Wolman 1993; Potyondy and Hardy 1995) and could be used in place of or in conjunction with the traditional methods. Photographic and videographic records are invaluable regardless of the streambed survey methods chosen.

Four provisions of existing legislation and four potential goals of prosecution are convictions, fines, compensatory damages, and remediation. When the state's purpose is to secure a conviction, a single water sample may be the only evidence required. In some jurisdictions, water quality criteria may be used to identify potential episodes of SS pollution by a tandem system of thresholds. Typically these guidelines state that SS concentrations should not exceed background by more than 10 mg SS/L when background is less than 100 mg SS/L and not more than 10% when background is equal to or greater than 100 mg SS/L (Singleton 1985a, 1985b). This tandem system of thresholds—based on literature reviews specifically intended to document the nature and severity of ill effect under these conditions—is commendable because it recognizes the seasonal patterns in suspended sediment load of natural streams. However, these guidelines do not purport to deal with the inherent nature of sediment as a deleterious substance in aquatic ecosystems as defined by an act of legislation. Nor do they purport to detect the least change in concentration capable of causing ill effects. Various researchers report ill effects when concentrations exceed



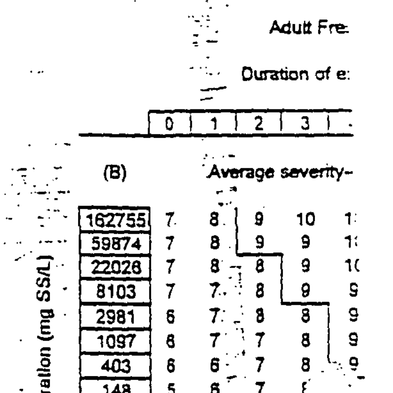


FIGURE 6

background levels by small amounts (see Lawrence and Scherer 1974; Swenson 1978; Gradall and Swenson 1982).

Prosecution based on these rules has bee cessful because the increased concentration known to harm aquatic life. Such evidence abounds, but pertains largely to invertebrate populations (fish food) and primary production (phy-

Adult Freshwater Nonsalmonids

Duration of exposure to SS ($\log_e$ hours)

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

| Concentration (mg SS/L) | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | ($\log_e$ mg SS/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 162755 | 7 | 8 | 9 | 10 | 10 | 11 | 12 | 12 | 13 | 14 | [illegible] | 12 |
| 59874 | 7 | 8 | 9 | 9 | 10 | 11 | 11 | 12 | 13 | [illegible] | [illegible] | 11 |
| 22026 | 7 | 8 | 8 | 9 | 10 | 10 | 11 | 12 | 13 | 13 | [illegible] | 10 |
| 8103 | 7 | 7 | 8 | 9 | 9 | 10 | 11 | 12 | 12 | 13 | [illegible] | 9 |
| 2981 | 6 | 7 | 8 | 8 | 9 | 10 | 11 | 11 | 12 | 13 | [illegible] | 8 |
| 1097 | 6 | 7 | 7 | 8 | 9 | 10 | 10 | 11 | 12 | 12 | [illegible] | 7 |
| 403 | 6 | 6 | 7 | 8 | 9 | 9 | 10 | 11 | 11 | 12 | [illegible] | 6 |
| 148 | 5 | 6 | 7 | 8 | 8 | 9 | 10 | 10 | 11 | 12 | 13 | 5 |
| 55 | 5 | 6 | 7 | 7 | 8 | 9 | 9 | 10 | 11 | 12 | 12 | 4 |
| 20 | 5 | 6 | 6 | 7 | 8 | 8 | 9 | 10 | 11 | 11 | 12 | 3 |
| 7 | [illegible] | 6 | 6 | 7 | 7 | 8 | 9 | 10 | 10 | 11 | 12 | 2 |
| 3 | [illegible] | 5 | 6 | 7 | 7 | 8 | 9 | 9 | 10 | [illegible] | [illegible] | 1 |
| 1 | [illegible] | [illegible] | 5 | 6 | 7 | 8 | 8 | 8 | 10 | 10 | [illegible] | 0 |
| | 1 | 3 | 7 | 1 | 2 | 6 | 2 | 7 | 4 | 11 | 30 | |
| | Hours | | | Days | | | Weeks | | Months | | | |

Half-95% confidence intervals (±)
around calculated severity-of-ill-effect scores (above)

| Concentration (mg SS/L) | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | ($\log_e$ mg SS/L) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 162755 | [illegible] | 1.6 | 1.5 | 1.5 | 1.4 | 1.5 | 1.5 | 1.8 | 1.7 | 1.8 | [illegible] | 12 |
| 59874 | [illegible] | [illegible] | 1.5 | [illegible] | 1.2 | 1.3 | [illegible] | 1.4 | [illegible] | 1.8 | [illegible] | 11 |
| 22026 | [illegible] | [illegible] | [illegible] | [illegible] | 1.0 | [illegible] | [illegible] | 1.2 | 1.3 | 1.4 | 1.5 | 10 |
| 8103 | [illegible] | [illegible] | 1.0 | 0.9 | 0.9 | 0.9 | 0.9 | 1.0 | 1.1 | 1.2 | 1.4 | 9 |
| 2981 | [illegible] | [illegible] | 0.9 | 0.8 | 0.7 | 0.7 | 0.8 | 0.8 | 1.0 | 1.1 | 1.3 | 8 |
| 1097 | [illegible] | [illegible] | 0.9 | 0.8 | 0.7 | 0.6 | 0.7 | 0.7 | 0.9 | 1.0 | 1.2 | 7 |
| 403 | [illegible] | [illegible] | 0.9 | 0.8 | 0.7 | 0.6 | 0.6 | 0.7 | 0.8 | 1.0 | 1.1 | 6 |
| 148 | [illegible] | [illegible] | 1.0 | 0.9 | 0.8 | 0.7 | 0.7 | 0.7 | 0.8 | 1.0 | 1.1 | 5 |
| 55 | [illegible] | [illegible] | 1.2 | 1.0 | 0.9 | 0.9 | 0.8 | 0.9 | 0.9 | 1.0 | 1.2 | 4 |
| 20 | [illegible] | [illegible] | [illegible] | [illegible] | [illegible] | 1.0 | 1.0 | 1.0 | [illegible] | 1.2 | 1.3 | 3 |
| 7 | [illegible] | [illegible] | 1.5 | [illegible] | 1.3 | 1.2 | 1.2 | 1.2 | 1.2 | 1.3 | 1.4 | 2 |
| 3 | [illegible] | [illegible] | [illegible] | [illegible] | 1.5 | 1.4 | 1.4 | 1.4 | 1.4 | 1.5 | 1.5 | 1 |
| 1 | [illegible] | 2.0 | 1.9 | 1.8 | 1.7 | 1.7 | 1.6 | 1.6 | 1.6 | 1.7 | 1.7 | 0 |
| | 1 | 3 | 7 | 1 | 2 | 6 | 2 | 7 | 4 | 11 | 30 | |
| | Hours | | | Days | | | Weeks | | Months | | | |

FIGURE 6.—Continued.

background levels by small amounts (see Lawrence and Scherer 1974; Swenson 1978; Gradall and Swenson 1982).

Prosecution based on these rules has been successful because the increased concentrations are known to harm aquatic life. Such evidence abounds, but pertains largely to invertebrate populations (fish food) and primary production (phytoplankton and periphyton, the source of energy on which invertebrates may depend) (Newcombe 1994).

However, to the extent that legislation emphasizes the existence of an impact, or the probability of an impact, its primary goal is to secure a conviction. Scope for additional penalty—fines, compensatory damages, and remediation—depends on

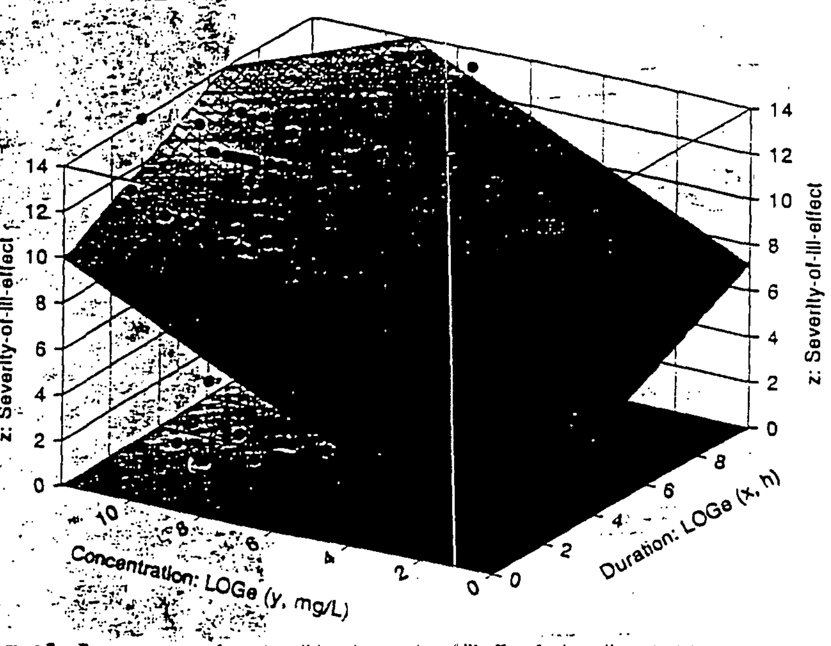


FIGURE 7.—Dose-response surfaces describing the severity of ill effect for juvenile and adult salmonids (freshwater, group 1) as a function of suspended sediment concentration and duration of exposure (model 1): $z = 1.0642 + 0.6068(\log_e x) + 0.7384(\log_e y)$.

an ability to demonstrate harmful effects. Dose-response models enhance this capability.

It is difficult to overstate the value of time series water quality data, but there are some kinds of pollution episodes in which other evidence might take precedence. These instances could be classed as catastrophic events in which one or more of the following conditions prevail: (i) the pollution damage is severe, or extensive and highly visible—blanketing by silt, for example; (ii) the extent of harm is to be confirmed by field studies designed and conducted for the purpose (especially relevant for streams on which previous work has been done); or (iii) the pollution event is detected after the fact, in which case the option to sample suspended sediment is foregone already. Notwithstanding these exceptions, efforts to collect sequential water samples during a pollution episode may be the most cost-effective option, especially when court fines, compensation, and remediation are high-priority goals.

In short, the dose-response equations proposed in this report make it possible not only to identify the existence of a pollution event—this information alone being sufficient to secure a conviction—but also to document the severity of ill effect in support of additional penalties.

*Meta-analysis*

No single researcher could have aspired to conduct all the field work represented in our database. However, the collective works have value beyond anything the original authors could have envisaged. To the extent that this synthesis informs the science, it demonstrates the utility of meta-analysis as a way to shed new light on old problems by using existing data. Limitations of the database can be overcome with further study.

*Future Research*

The dose-response models in this synthesis are only a beginning. Many gaps remain. Gaps are



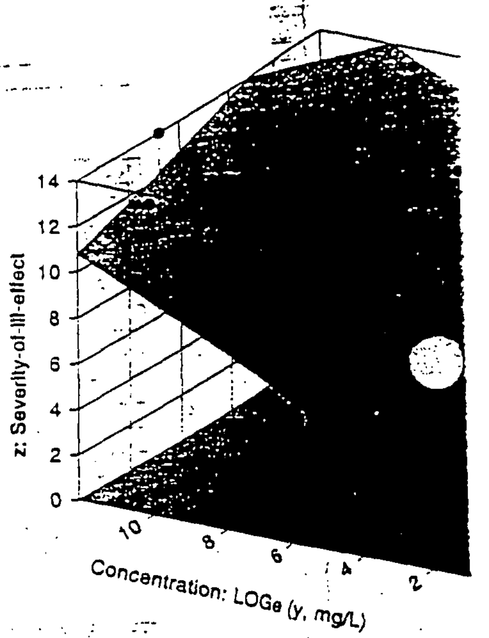


FIGURE 8.—Dose-response surface describing the severity of i as a function of suspended sediment concentration and duration o + 0.7565($\log_e y$).

especially conspicuous for the youngest age-classes (eggs through young juveniles). The pooling of life stages required for these models—eggs with larvae, young with old juveniles—doubtless masks important thresholds of susceptibility to suspended sediment. Each developmental stage should be identified and treated separately for the purpose of developing uniquely age-specific and size-specific dose-response profiles.

There are practical reasons to make such distinctions. For example, artificial spawning channels must be cleaned annually. Gravel cleaning, which raises a plume of silty water, therefore must be carefully timed to minimize the potential ill effects. Susceptibilities of resident life stages to sediment must be known.

Thresholds of sublethal and lethal effects must be known more precisely. Our analysis has shown, in particular, that sublethal effects thresholds are poorly delineated for most groups. Finding useable data is a challenge; we rejected many studies be-

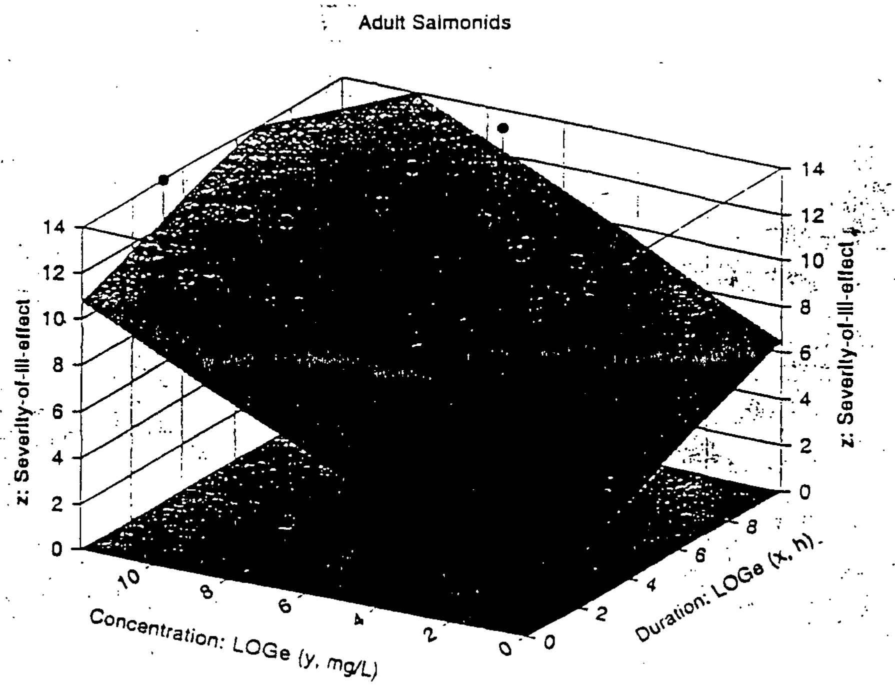


FIGURE 8.—Dose–response surface describing the severity of ill effect for adult salmonids (freshwater, group 2) as a function of suspended sediment concentration and duration of exposure (model 2): $z = 1.6814 + 0.4769(\log_e x) + 0.7565(\log_e y)$.

especially conspicuous for the youngest age-classes (eggs through young juveniles). The pooling of life stages required for these models—eggs with larvae, young with old juveniles—doubtless masks important thresholds of susceptibility to suspended sediment. Each developmental stage should be identified and treated separately for the purpose of developing uniquely age-specific and size-specific dose–response profiles.

There are practical reasons to make such distinctions. For example, artificial spawning channels must be cleaned annually. Gravel cleaning, which raises a plume of silty water, therefore must be carefully timed to minimize the potential ill effects. Susceptibilities of resident life stages to sediment must be known.

Thresholds of sublethal and lethal effects must be known more precisely. Our analysis has shown, in particular, that sublethal effects thresholds are poorly delineated for most groups. Finding useable data is a challenge; we rejected many studies because they were too vague about sediment concentration, duration of exposure, or the exact nature of the ill effect. We undoubtedly overlooked some reports, but more directed research is warranted. Research is especially needed into particle quality (particle size, angularity, and mineralogy), particle toxicity (toxicants in and adsorbed on sediments), and temperature effects.

*Particle quality and toxicology.*—Ill effects increase as a function of increasing particle size (if other variables are kept constant). Pollution events often subject fish to particle sizes to which they are not normally exposed. Newcombe et al. (1995) documented that rainbow trout died rapidly when exposed to a silty water discharge (mortality, 80–100%; concentration, ≈4,315 mg SS/L; duration, <57 h; particle sizes, 100–170 μm, water temperature, 10°C). These results differ from those from other pollution episodes in which the particle size was smaller; generally, the ill effects would be much less severe—on the order of 0–10% mor-

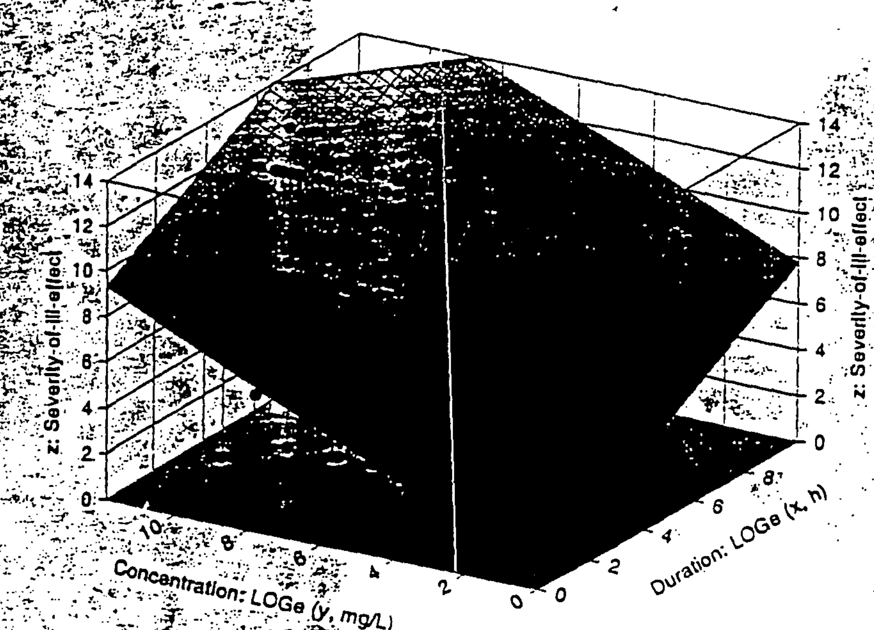


FIGURE 9.—Dose-response surface describing the severity of ill effect for juvenile salmonids (freshwater, group 3) as a function of suspended sediment concentration and duration of exposure (model 3): $z = 0.7262 + 0.7034(\log_e x) + 0.7144(\log_e y)$.

tality. Some research to quantify ill effect as a function of particle size has been done with several species of Pacific salmon (Servizi and Martens 1987, 1991, 1992). Further work should make it possible to create a set of dose-response models as functions of particle size range that are unique to each relevant life stage. The growing need to explore ill effects of suspended sediment as a function of particle size imposes an obligation among fisheries biologists to use a uniform nomenclature in reference to the particle grade scale. Suitable systems exist already so there is no need to invent a more specialized one. For example, soils scientists recognize three particle size-classes—sand, silt and clay (Agriculture Canada 1974)—with formalized subdivisions, names, and sizes as follows: very coarse sand, 2.0–1.0 mm; coarse sand, 1.0–0.5 mm; medium sand, 0.5–0.25 mm; fine sand, 0.25–0.10 mm; very fine sand, 0.10–0.05 mm; silt, 0.05–0.002 mm; and clay, ≤0.002 mm. Fisheries biologists would do well to adopt this or some similar particle grade scale.

The importance of particle angularity, especially in relation to gill abrasion, should be studied. The mineralogy of sediment particles may offer clues to the potential for toxicity and physiological effects. Likewise, the presence of innate or adsorbed toxicants may offer clues to latent effects on fish population health. Studies of the mineralogy and potential chemical activity of the particle itself, of particles in the colloidal size range capable of entering the fish's cells, and of particles with adsorbed toxicants may reveal common properties relating to fate and ill effect at the tissue and cellular level. If common properties do exist among these particular variables, there may be a unifying explanation in the phenomenon of phagocytosis.

Phagocytosis, the envelopment of fine particles by cells of the fish's gill and gut, transports the particles into the fish's body. Although these par-



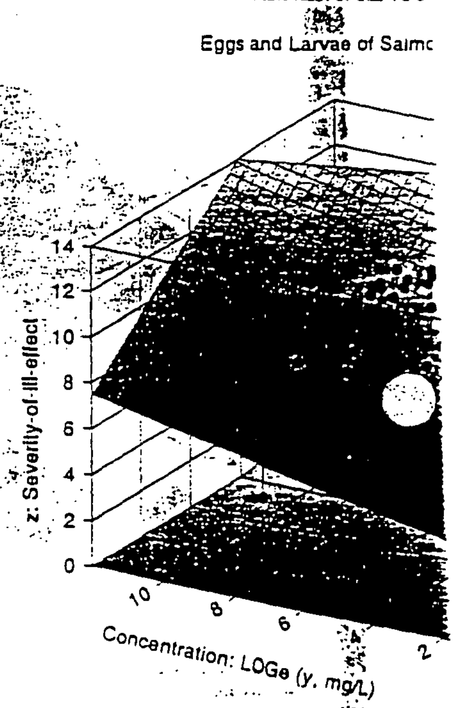


FIGURE 10.—Dose-response surface describing the sever nonsalmonids (freshwater and estuarine, group 4) as a functi of exposure (model 4): $z = 3.7466 + 1.0946(\log_e x) + 0.31$

ticles may end up in various tissues, the spleen is a major repository. The spleens of some fishes exposed to fine sediment become mineralized to the extent that the tissue damages the cutting edge of the glass microtome blades (Goldes 1983; S. Goldes, Malaspina College, personal communication). Thus, phagocytosis of fine suspended sediments could trigger a sequence of harmful events within the cells of a fish's body leading to ill effects that are only partially understood today. Invasive particles may be the biological equivalent of a Trojan horse: harmless when on the outside, devastating when on the inside. Tumorigenesis, especially among groundfish that dwell in harbors where sediments may be contaminated by stormwater runoff or by industrial effluent, may be one such latent ill effect yet to be linked to this phenomenon.

*Water temperature.*—Severity of ill effect as a function of ambient water temperature ought to be explored more fully. Ill effects are greater in sea-

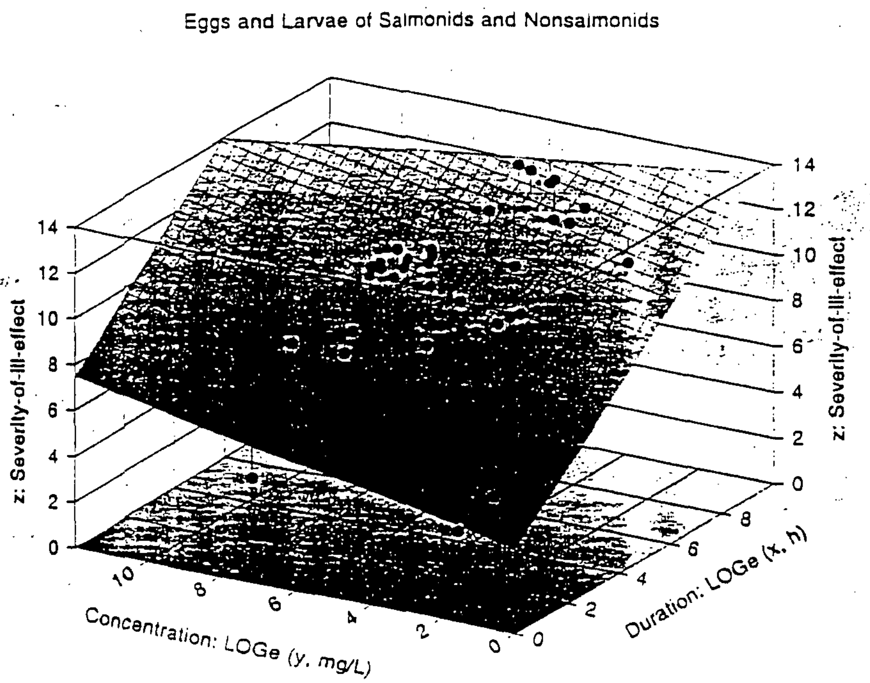


FIGURE 10.—Dose–response surface describing the severity of ill effect for eggs and larvae of salmonids and nonsalmonids (freshwater and estuarine, group 4) as a function of suspended sediment concentration and duration of exposure (model 4): $z = 3.7466 + 1.0946(\log_e x) + 0.3117(\log_e y)$.

ticles may end up in various tissues, the spleen is a major repository. The spleens of some fishes exposed to fine sediment become mineralized to the extent that the tissue damages the cutting edge of the glass microtome blades (Goldes 1983; S. Goldes, Malaspina College, personal communication). Thus, phagocytosis of fine suspended sediments could trigger a sequence of harmful events within the cells of a fish's body leading to ill effects that are only partially understood today. Invasive particles may be the biological equivalent of a Trojan horse: harmless when on the outside, devastating when on the inside. Tumorigenesis, especially among groundfish that dwell in harbors where sediments may be contaminated by stormwater runoff or by industrial effluent, may be one such latent ill effect yet to be linked to this phenomenon.

*Water temperature.*—Severity of ill effect as a function of ambient water temperature ought to be explored more fully. Ill effects are greater in seasonably warm water than would be the case for the same fishes in seasonably cold water. Mechanisms for this effect have not been systematically described. The dynamics of this variable probably have to do with the temperature-related patterns of oxygen saturation, respiration rate, and metabolic rate of fishes (slower in cool water, more rapid in warm)—all of which result in reduced risk of gill abrasion in cool water and increased risk in warm water. These mechanisms should be explored in the context of seasonal temperature ranges in a fish's natural habitat.

### *Ecosystem Considerations*

Broad-based ecosystem research supporting stream protection is under way, but it is a relatively new science. Stream protection requires, among other things, quantitative linkages between impacts of channel sediment and the land use practices that generate the sediment. Leadership in this area will come from many disciplines, as exem-

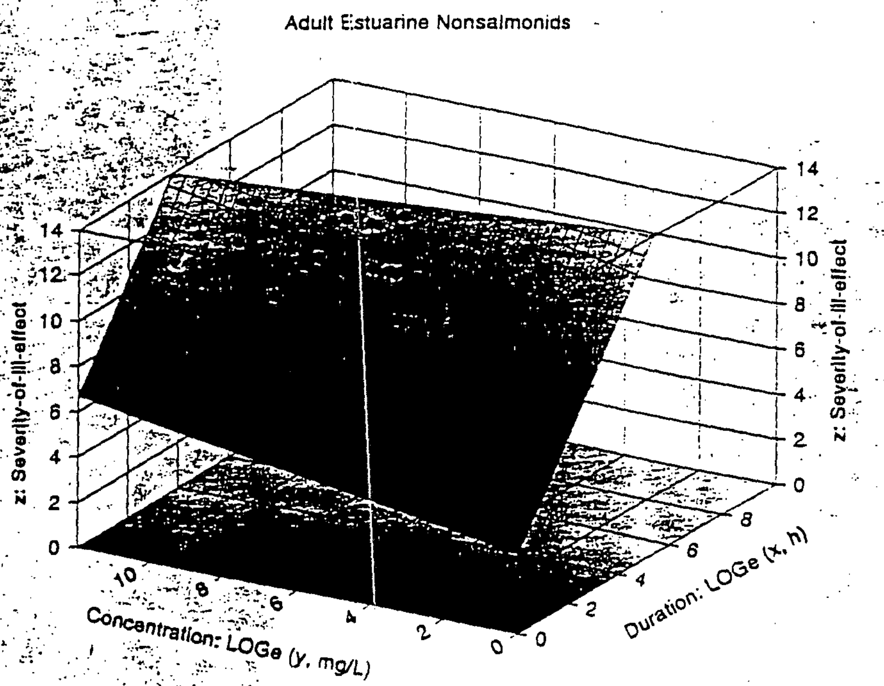


FIGURE 11.—Dose-response surface describing the severity of ill effect for adult nonsalmonids (estuarine, group 5) as a function of suspended sediment concentration and duration of exposure (model 5): $z = 3.4969 + 1.9647(\log_e x) + 0.2669(\log_e y)$.

plified by several important contributions dealing with water quality, resource roads, timber harvest, and channel sediment (Cederholm et al. 1981; Chamberlin 1988; Hartman 1988; Macdonald et al. 1992; Davies and Nelson 1993; Grayson et al. 1993; Macdonald 1994). This research emphasizes the consequences of land disturbance in the upland and riparian zones. It shows that the upland zone capable of impacts on stream quality may be much larger than previously supposed—especially in hilly terrain. The size of upland and riparian zones may be a function of the time scale used to view them. Latent impacts of land use practices—reduced slope stability, increased frequency and severity of flooding, more frequent and longer-lasting episodes of channel sediment pollution—may develop decades after the fact of land disturbance. Thus we should broaden our definition of the upland and riparian zones to accommodate latent ill effects from land disturbance. A broader definition, to the extent it is scientifically supported, can justify a wider legislated zone of protection that extends well into the upland, far away from the stream itself.

Suspended channel sediment is a major factor determining stream quality. Excess sediment is a serious but still underrated pollutant. Unless it is addressed, instream and riparian zones can not be reliably protected. Although the need for increased protection of instream environments might be publicly acceptable, the case for increased protection of upland and riparian areas in aid of stream protection has yet to be made.

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We are grateful to Harold Mundie (Nanaimo, British Columbia) for his sustained interest in this study and for his many thoughtful suggestions. We also thank Jacqueline LaPerriere (Alaska Cooperative Fisheries Research Unit, University of Alaska, Fairbanks), Ron Ptolemy (Fisheries Branch, Ministry of Environment, Lands and



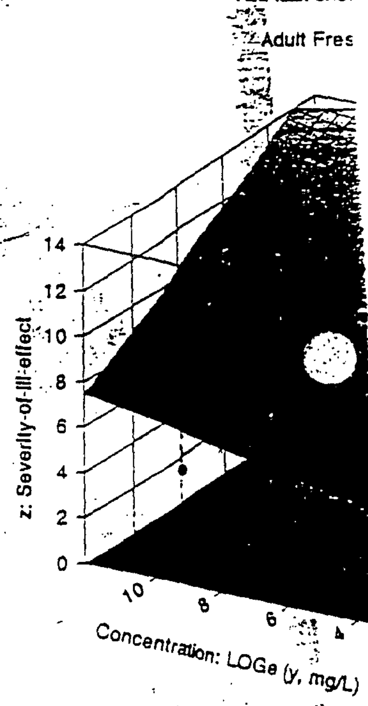


FIGURE 12.—Dose-response surface describing the
6) as a function of suspended sediment concentration a
+ 0.2829($\log_e y$).

Parks), and Jerry Sweeten (Asherwood Learnin
Center) for raw data; Sally Goldes (Fisherie
Branch, Malaspina College, Nanaimo) for infor
mation about fate and effects of small particles o
cells and tissues of fish; Mike Miles (Mike Mile
and Associates, Victoria, British Columbia), How
ard Singleton (Water Quality Branch, Ministry o
Environment, Lands and Parks), and Mark Labelle
(Institut Français de Recherche pour l'Exploitation
de la Mer, Nantes Cedex) for various suggestions.
Bill McLean (Qunisam River Hatchery, Campbel
River, British Columbia) for field-testing some o
the models; and American Fisheries Society re
viewers and staff for their numerous improv... ...nts
to the manuscript.

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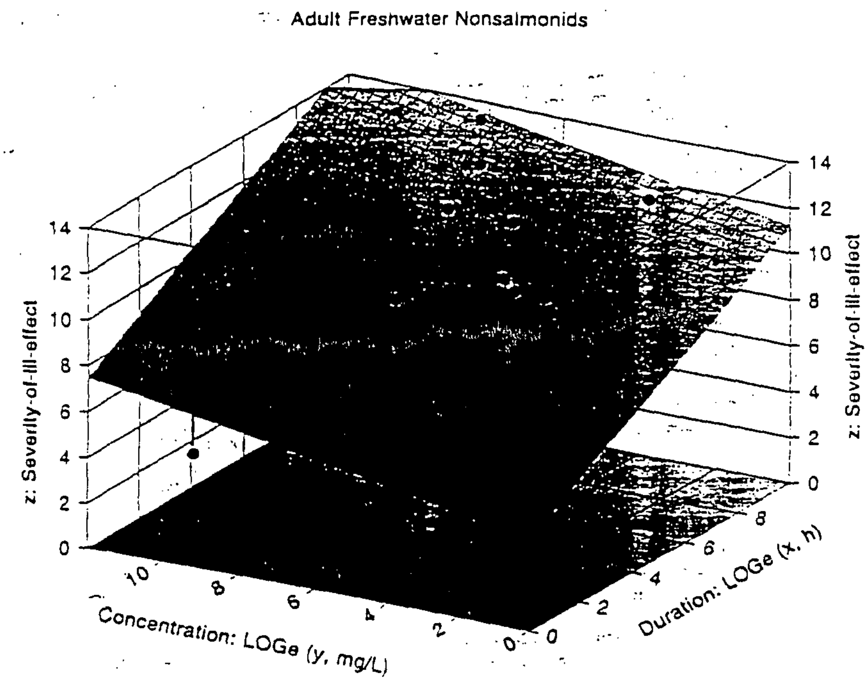


FIGURE 12.—Dose–response surface describing the severity of ill effect for adult nonsalmonids (freshwater, group 6) as a function of suspended sediment concentration and duration of exposure (model 6): $z = 4.0815 + 0.7126(\log_e x) + 0.2829(\log_e y)$.

Parks), and Jerry Sweeten (Asherwood Learning Center) for raw data; Sally Goldes (Fisheries Branch, Malaspina College, Nanaimo) for information about fate and effects of small particles on cells and tissues of fish; Mike Miles (Mike Miles and Associates, Victoria, British Columbia), Howard Singleton (Water Quality Branch, Ministry of Environment, Lands and Parks), and Mark Labelle (Institut Français de Recherche pour l'Exploitation de la Mer, Nantes Cedex) for various suggestions; Bill McLean (Qunisam River Hatchery, Campbell River, British Columbia) for field-testing some of the models; and American Fisheries Society reviewers and staff for their numerous improvements to the manuscript.

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**Appendix follows on page 720**

## Appendix: Dose–Response Database

TABLE A.1 Dose–response database for fishes exposed to suspended sediment.

| Species | Life stage[a] | Sediment dose: Exposure concentration (mg/L) | Sediment dose: Exposure duration (h) | Fish response: SEV[b] | Fish response: Description[c] | Reference |
|---|---|---|---|---|---|---|
| **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 | 16.5 | 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 | 0.17 | 7 | Home water preference disrupted | Whitman et al. (1982) |
| Salmon (chinook) | A | 650 | 168 | 7 | Homing behavior normal, but fewer test fish returned | Whitman et al. (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 | 3.5 | 168 | 7 | Increased vulnerability to predation | Swenson (1978) |
| Steelhead | A | 500 | 3 | 5 | Signs of sublethal stress (VA) | Redding and Schreck (1982) |
| Steelhead | A | 1,650 | 240 | 7 | Loss of habit caused by excessive sediment transport | Coats et al. (1985) |
| Steelhead | A | 500 | 9 | 8 | Blood cell count and blood chemistry change | Redding and Schreck (1982) |
| Trout | A | 16.5 | 24 | 4 | Feeding behavior apparently reduced | Townsend (1983); Ott (1984) |
| 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 | 4.5 | 168 | 3 | Fish more active and less dependent on cover | Gradall and Swenson (1982) |



TABLE A.1.—Continued.

| Species | Life stage[a] | Sediment dose: Exposure concentration (mg/L) | Sediment dose: Exposure duration (h) | SEV[b] | |
|---|---|---|---|---|---|
| Trout (brown) | A | 1,040 | 17,520 | 8 | Gill |
| Trout (brown) | A | 1,210 | 17,520 | 8 | Some (VF |
| Trout (brown) | A | 18 | 720 | 10 | Abunc |
| Trout (brown) | A | 100 | 720 | 11 | Popula |
| Trout (brown) | A | 1,040 | 8,760 | 14 | Popula expe |
| Trout (brown) | A | 5,838 | 8,760 | 14 | Fish or expe |
| Trout (cutthroat) | A | 35 | 2 | 4 | [illegible] |
| Trout (lake) | A | 3.5 | 168 | 3 | [illegible] |
| Trout (rainbow) | A | 66 | 1 | 3 | Avoida part c |
| Trout (rainbow) | A | 665 | 1 | 3 | Fish ac |
| Trout (rainbow) | A | 100 | 0.10 | 3 | Fish av (avoi |
| Trout (rainbow) | A | 100 | 0.25 | 5 | Rate of |
| Trout (rainbow) | A | 250 | 0.25 | 5 | Rate of |
| Trout (rainbow) | A | 810 | 504 | 8 | Gills of thicke |
| Trout (rainbow) | A | 17,500 | 168 | 8 | Fish surv prolife |
| Trout (rainbow) | A | 50 | 960 | 9 | Rate of (CWS |
| Trout (rainbow) | A | 50 | 960 | 9 | Rate of w |
| Trout (rainbow) | A | 810 | 504 | 10 | Some fish |
| Trout (rainbow) | A | 270 | 3,240 | 10 | Survival r |
| Trout (rainbow) | A | 200 | 24 | 10 | Test fish day (W |
| Trout (rainbow) | A | 80,000 | 24 | 10 | No mortal |
| Trout (rainbow) | A | 18 | 720 | 10 | Abundance |
| Trout (rainbow) | A | 59 | 2,232 | 10 | Habitat dar of grave |
| Trout (rainbow) | A | 4,250 | 588 | 12 | Mortality r |
| Trout (rainbow) | A | 49,838 | 96 | 12 | Mortality r |
| Trout (rainbow) | A | 3,500 | 1,488 | 13 | Catastroph populatio |
| Trout (rainbow) | A | 160,000 | 24 | 14 | Mortality r |
| Trout (sea) | A | 210 | 24 | 10 | Fish aband spawning |
| Whitefish (lake) | A | 0.66 | 1 | 3 | Swimming |
| Whitefish (lake) | A | 16,613 | 96 | 12 | Mortality ra |
| Whitefish (mountain) | A | 10,000 | 24 | 10 | Fish died; s |
| **Juvenile salmonids (freshwater, gr** | | | | | |
| Grayling (Arctic) | U | 20 | 24 | 3 | Fish avoide |
| Grayling (Arctic) | U | 10,000 | 96 | 3 | Fish swam |
| Grayling (Arctic) | J | 86 | 0.42 | 3 | 78% of fish (NTU, > |
| Grayling (Arctic) | U | 100 | 1 | 4 | C[illegible] |
| Grayling (Arctic) | U | 100 | 1 | 4 | Catch rate prey: tub |
| Grayling (Arctic) | U | 300 | 1 | 4 | Catch rate prey: dro |
| Grayling (Arctic) | U | 1,000 | 1 | 4 | Feeding rat prey: tub |

TABLE A.1.—Continued.

| Species | Life stage[a] | Sediment dose: Exposure concentration (mg/L) | Sediment dose: Exposure duration (h) | Fish response: SEV[b] | Fish response: Description[c] | Reference |
|---|---|---|---|---|---|---|
| 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 became 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 | 3.5 | 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 avoided turbid water (avoidance behavior) | 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 (WF) | 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. Herbert, personal communication to Alabaster and Lloyd (1980) |
| 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% (DM) | 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. Herbert, personal communication to Alabaster and Lloyd (1980) |
| Trout (sea) | A | 210 | 24 | 10 | Fish abandoned traditional spawning habitat | Hamilton (1961) |
| Whitefish (lake) | A | 0.66 | 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 (1988) |
| 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: tubificids) | McLeay et al. (1987) |
| Grayling (Arctic) | U | 300 | 1 | 4 | Catch rate reduced (unfamiliar prey: drosophila) | McLeay et al. (1987) |
| Grayling (Arctic) | U | 1,000 | 1 | 4 | Feeding rate reduced (unfamiliar prey: tubificids) | McLeay et al. (1987) |

TABLE A.1.—Continued.

| | | Sediment dose | | Fish response | | |
|---|---|---|---|---|---|---|
| Species | Life stage[a] | Exposure concentration (mg/L) | Exposure duration (h) | SEV[b] | Description[c] | Reference |
| Trout (rainbow) | Y | 90 | 456 | 10 | Mortality rates 0–20% (DE) | Herbert and Merkens (1961) |
| Trout (rainbow) | Y | 90 | 456 | 10 | Mortality rates 0–15% (KC) | Herbert and Merkens (1961) |
| Trout (rainbow) | Y | 270 | 456 | 11 | Mortality rates 10–35% (KC) | Herbert and Merkens (1961) |
| Trout (rainbow) | Y | 810 | 456 | 12 | Mortality rates 35–85% (DE) | Herbert and Merkens (1961) |
| Trout (rainbow) | Y | 810 | 456 | 12 | Mortality rates 5–80% (KC) | Herbert and Merkens (1961) |
| Trout (rainbow) | Y | 270 | 456 | 12 | Mortality rates 25–80% (DE) | Herbert and Merkens (1961) |
| Trout (rainbow) | Y | 7,433 | 672 | 11 | Mortality rate 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 (personal communication) |
| Grayling (Arctic) | SF | 22.5 | 48 | 10 | Mortality rate 14.0% | J. LaPerriere (personal communication) |
| Grayling (Arctic) | SF | 65 | 24 | 10 | Mortality rate 15.0% | J. LaPerriere (personal communication) |
| Grayling (Arctic) | SF | 21.7 | 72 | 10 | Mortality rate 14.7% | J. LaPerriere (personal communication) |
| Grayling (Arctic) | SF | 20 | 96 | 10 | Mortality rate 13.4% | J. LaPerriere (personal communication) |
| Grayling (Arctic) | SF | 142.5 | 48 | 11 | Mortality rate 26% | J. LaPerriere (personal communication) |
| Grayling (Arctic) | SF | 185 | 72 | 12 | Mortality rate 41.3% | J. LaPerriere (personal communication) |
| Grayling (Arctic) | SF | 230 | 96 | 12 | Mortality rate of 47% | J. LaPerriere (personal communication) |
| 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 et al. (1981) |
| Trout (rainbow) | EE | 1,750 | 144 | 10 | Mortality rate greater than controls (controls, 6%) | Campbell (1954) |
| Trout (rainbow) | E | 6.6 | 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 rates 60–70% (controls, 38.6%) | Erman and Ligon (1988) |
| Trout (rainbow) | E | 20.8 | 1,152 | 13 | Mortality rate 72% | Slaney et al. (1977a) |
| Trout (rainbow) | E | 46.6 | 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) |
| **Nonsalmonid eggs and larvae (estuarine[d], 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) |

| SEV[b] | Fish response: Description[c] | Reference |
|---|---|---|
| 10 | Mortality rates 0–20% (DE) | Herbert and Merkens (1961) |
| 10 | Mortality rates 0–15% (KC) | Herbert and Merkens (1961) |
| 11 | Mortality rates 10–35% (KC) | Herbert and Merkens (1961) |
| 12 | Mortality rates 35–85% (DE) | Herbert and Merkens (1961) |
| 12 | Mortality rates 5–80% (KC) | Herbert and Merkens (1961) |
| 12 | Mortality rates 25–80% (DE) | Herbert and Merkens (1961) |
| 11 | Mortality rate 40% (CS) | Herbert and Wakeford (1962) |
| 12 | Mortality rate 50% | Herbert and Wakeford (1962) |
| 14 | Mortality rate 100% | Herbert and Wakeford (1962) |
| 14 | Mortality rate ~100% (CSS) | Newcombe et al. (1995) |
| d larvae (freshwater, group 4) | | |
| 10 | Mortality rate 5.7% | J. LaPerriere (personal communication) |
| 10 | Mortality rate 14.0% | J. LaPerriere (personal communication) |
| 10 | Mortality rate 15.0% | J. LaPerriere (personal communication) |
| 10 | Mortality rate 14.7% | J. LaPerriere (personal communication) |
| 10 | Mortality rate 13.4% | J. LaPerriere (personal communication) |
| 11 | Mortality rate 26% | J. LaPerriere (personal communication) |
| 12 | Mortality rate 41.3% | J. LaPerriere (personal communication) |
| 12 | Mortality rate of 47% | J. LaPerriere (personal communication) |
| 10 | Mortality; deterioration of spawning gravel | Cederholm et al. (1981) |
| 3 | Mortality rate 77% (controls, 6%) | Langer (1980) |
| 4 | Mortality rate 100% (controls, 16.2%) | Shaw and Maga (1943) |
| 2 | Hatching success 42% (controls, 63%) | Slaney et al. (1977b) |
| 0 | Mortality; deterioration of spawning gravel | Cederholm et al. (1981) |
| 0 | Mortality rate greater than controls (controls, 6%) | Campbell (1954) |
| 1 | Mortality rate 40% | Slaney et al. (1977b) |
| 2 | Mortality rate 47% (controls, 12%) | Slaney et al. (1977b) |
| | ortality rates 60–70% (controls, 38.6%) | Erman and Ligon (1988) |
| 1 | Mortality rate 72% | Slaney et al. (1977a) |
| [illegible] | Mortality rate 100% | Slaney et al. (1977b) |
| | Mortality rate 98% (controls, 14.6%) | Turnpenny and Williams (1980) |
| d larvae (estuarine[d], group 4) | | |
| | Feeding rate reduced 40% | Breitburg (1988) |
| | Development rate slowed significantly | Morgan et al. (1983) |
| | Hatching delayed | Schubel and Wang (1973) |
| | Reduced hatching success | Auld and Schubel (1978) |
| | Mortality rate 35% (controls, 16%) | Auld and Schubel (1978) |
| | Mortality rate 42% (controls, 17%) | Auld and Schubel (1978) |
| | Mortality rate 50% | Morgan et al. (1973) |



TABLE A.1.—Continued.

| Species | Life stage[a] | Sediment dose: Exposure concentration (mg/L) | Sediment dose: Exposure duration (h) | Fish response: SEV[b] | Fish response: Description[c] | Reference |
|---|---|---|---|---|---|---|
| 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; microridges less distinct | Boehlert (1984) |
| Perch (white) | 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 et al. (1973) |
| Perch (yellow) | L | 500 | 96 | 11 | Mortality rate 37% (controls, 7%) | Auld 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 36% (controls, 4%) | Auld and Schubel (1978) |
| Shad (American) | L | 1,000 | 96 | 11 | Mortality rate 34% (controls, 5%) | Auld and Schubel (1978) |
| Adult nonsalmonids (estuarine or riverine–estuarine, group 5) | | | | | | |
| 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 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 et al. (1975) |
| 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) |
| 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) |

TABLE A.1.—Continued.

| Species | Life stage[a] | Sediment dose: Exposure concentration (mg/L) | Sediment dose: Exposure duration (h) | Fish response: SEV[b] | Fish response: Description[c] | Reference |
|---|---|---|---|---|---|---|
| Perch (white) | A | 3,181 | 24 | 14 | Mortality rate 90% (FE) | Sherk et al. (1975) |
| Rasbora (harlequin) | A | 40,000 | 24 | 10 | Fish died (BC) | Alabaster and Lloyd (1980) |
| Rasbora (harlequin) | 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. (1983) |
| 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 et 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 (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 | 130,000 | 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) |
| Stickleback (threespine) | A | 28,000 | 96 | 10 | No mortality in test designed to identify lethal threshold | LeGore and DesVoigne (1973) |
| Toadfish (oyster) | A | 3,360 | 1 | 6 | Oxygen consumption more variable in prestressed fish | Neumann et al. (1975) |
| 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 nonsalmonids (freshwater, group 6) | |
| Bass (largemouth) | A | 62.5 | 720 | 9 | Weight gain reduced ~50% | Buck (1956) |
| Bass (largemouth) | A | 144.5 | 720 | 9 | Growth retarded | Buck (1956) |
| Bass (largemouth) | A | 144.5 | 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 | 144.5 | 720 | 9 | Growth retarded | Buck (1956) |
| Bluegill | A | 62.5 | 720 | 9 | Weight gain reduced ~50% | Buck (1956) |
| Bluegill | A | 144.5 | 720 | 12 | Fish unable to reproduce | Buck (1956) |
| Carp (common) | A | 25,000 | 336 | 10 | Some mortality (MC) | Wallen (1951) |
| Darters | A | 2,045 | 8,760 | 14 | Darters absent | Vaughan (1979); Vaughan et al. (1982) |
| Fish | A | 120 | 384 | 10 | Density of fish reduced | Erman and Ligon (1988) |
| Fish | A | 620 | 48 | 10 | Fish kills downstream from sediment source | Hesse and Newcomb (1982) |
| Fish | A | 900 | 720 | 12 | Fish absent or markedly reduced in abundance | Herbert and Richards (1963) |
| Fish | A | 2,045 | 8,760 | 12 | Habitat destruction: fish populations smaller than expected | Vaughan (1979); Vaughan et al. (1982) |
| Fish (warmwater) | A | 100,000 | 252 | 10 | Some fish died; most survived | Wallen (1951) |
| Fish (warmwater) | A | 200,000 | 1.125 | 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) |

| SEV[b] | Fish response: Description[c] | Reference |
|---|---|---|
| 14 | Mortality rate 90% (FE) | Sherk et al. (1975) |
| 10 | Fish died (BC) | Alabaster and Lloyd (1980) |
| 10 | No mortality | Alabaster and Lloyd (1980) |
| 3 | Change in preferred swimming depth | Dadswell et al. (1983) |
| 10 | Mortality rate 10% (FE) | Sherk et al. (1975) |
| 12 | Mortality rate 50% (FE) | Sherk et al. (1975) |
| 14 | Mortality rate 90% (FE) | Sherk et al. (1975) |
| 10 | Mortality rate 10% (FE) | Sherk et al. (1975) |
| 10 | Mortality rate 10% (FE) | Sherk et al. (1975) |
| 10 | Mortality rate 10% | Sherk et al. (1975) |
| 12 | Mortality rate 50% (FE) | Sherk et al. (1975) |
| 12 | Mortality rate 50% | Sherk et al. (1975) |
| 12 | Mortality rate 50% | Sherk et al. (1975) |
| 14 | Mortality rate 90% (FE) | Sherk et al. (1975) |
| 14 | Mortality rate 90% | Sherk et al. (1975) |
| 10 | Mortality rate <1% (IA) | Rogers (1969) |
| 10 | No mortality (KS; 10–12°C) | Rogers (1969) |
| 12 | Mortality rate ~50% (IA) | Rogers (1969) |
| 12 | Mortality rate 50% (15.0–16.0°C) | Rogers (1969) |
| 12 | Mortality rate 50% (KS) | Rogers (1969) |
| 12 | Mortality rate 50% (10–12°C) | Rogers (1969) |
| 12 | Mortality rate 50% (9.0–9.5°C) | Rogers (1969) |
| 14 | Mortality rate 100% | Rogers (1969) |
| 14 | Mortality rate 95% (KS) | Rogers (1969) |
| 10 | No mortality in test designed to identify lethal threshold | LeGore and DesVoigne (1973) |
| 6 | Oxygen consumption more variable in prestressed fish | Neumann et al. (1975) |
| 8 | Fish largely unaffected, but developed latent ill effects | Neumann et al. (1975) |
| 9 | Latent ill effects manifested in subsequent test at low SS | Neumann et al. (1975) |
| ...lmonids (freshwater, group 6) | | |
| 9 | Weight gain reduced ~50% | Buck (1956) |
| 9 | Growth retarded | Buck (1956) |
| 12 | Fish unable to reproduce | Buck (1956) |
| 4 | Rate of feeding reduced | Gardner (1981) |
| 4 | Reduced capacity to locate prey | Vinyard and O'Brien (1976) |
| 9 | Growth retarded | Buck (1956) |
| [illegible] | Weight gain reduced ~50% | Buck (1956) |
| [illegible] | Fish unable to reproduce | Buck (1956) |
| [illegible] | Some mortality (MC) | Wallen (1951) |
| 14 | Darters absent | Vaughan (1979); Vaughan et al. (1982) |
| 10 | Density of fish reduced | Erman and Ligon (1988) |
| 10 | Fish kills downstream from sediment source | Hesse and Newcomb (1982) |
| 12 | Fish absent or markedly reduced in abundance | Herbert and Richards (1963) |
| 12 | Habitat destruction; fish populations smaller than expected | Vaughan (1979); Vaughan et al. (1982) |
| 10 | Some fish died; most survived | Wallen (1951) |
| 10 | Fish died; opercular cavities and gill filaments clogged | Wallen (1951) |
| 12 | Fish populations destroyed | Menzel et al. (1984) |
| 10 | Some mortality (MC) | Wallen (1951) |



TABLE A.1.—Continued.

| Species | Life stage[a] | Sediment dose: Exposure concentration (mg/L) | Sediment dose: Exposure duration (h) | Fish response: SEV[b] | Fish response: Description[c] | Reference |
|---|---|---|---|---|---|---|
| Sunfish (green) | A | 9,600 | 1 | 5 | Rate of ventilation increased | Horkel and Pearson (1976) |
| Sunfish (redear) | A | 62.5 | 720 | 9 | Weight gain reduced ~50% compared to controls | Buck (1956) |
| Sunfish (redear) | A | 144.5 | 720 | 9 | Growth retarded | Buck (1956) |
| Sunfish (redear) | A | 144.5 | 720 | 12 | Fish unable to reproduce | Buck (1956) |

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

[b] Severity-of-ill-effect ranging from 0 (no detectible effect) to 14 (maximum effect; see Table 1).

[c] Full response annotations are in Newcombe (1994). Particle sizes of suspended sediment (SS) sometimes were given categorically in source documents. As abbreviated here, VFSS = very fine (<15 μm); FSS = fine (15–74 μm); MPSS = medium to fine (75–149 μm); MCSS = medium to coarse (150–290 μm); and CSS = coarse (180–740 μm). Usual "sediments" used: BC = bentonite clay; CS = calcium sulfate; CWS = coal washery solids; DE = diatomaceous earth; DM = drilling mud (nontoxic); FC = fire clay; FE = fuller's earth; IA = incinerator ash; KC = kaolin clay; KS = Kingston silt; LNFH = lime-neutralized ferric hydroxide; MC = montmorillonite clay; VA = volcanic ash; WF = wood fibers. Other abbreviation: NTU = nephelometric turbidity units.

[d] Lake herring larvae were tested in freshwater.