ESTIMATION OF EFFECT THRESHOLDS FOR THE DEVELOPMENT OF WATER QUALITY CRITERIA

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Abstract: Biological and ecological effect thresholds can be used for determining safe levels of nontraditional stressors. The U.S. EPA Framework for Developing Suspended and Bedded Sediments (SABS) Water Quality Criteria (WQC) [36] uses a risk assessment approach to estimate effect thresholds for unacceptable levels of SABS in water bodies. Sources of SABS include:

1. Erosion from agricultural, construction, forestry practices, and stream banks
2. Resuspension of deposited sediment
3. Direct discharge from municipal, industrial, and agricultural sources

Excessive levels of SABS can destroy habitat for plants and animals, reduce the quality of drinking water, impair the quality and safety of recreational waters, increase the costs associated with irrigation and navigation, and decrease aesthetics. The SABS Framework is intended as a guide to the development of water quality criteria (WQC) and restoration targets. The *SABS Framework* uses an eco-epidemiological perspective to incorporate information from field observations with data from controlled laboratory experiments. The combined information is used to develop relationships that estimate the levels of SABS that will impair aquatic life or pollute sources

intended for drinking water. The *SABS Framework* uses several statistical procedures to compare the estimated effects levels derived from field and laboratory data. Protective levels and restoration goals are recommended based on scientific precedent, logical argument, and statistical resolution. The risk estimates that result from using this approach are readily applicable for use in future emergency situations.

1. Introduction

Any substance or agent has the potential to cause environmental harm. The detrimental effects of a limited number of substances are characterized in criteria documents and existing, completed risk assessments [42]. Based on these prior assessments, risk managers are able to develop possible actions for protecting and restoring environmental conditions. These actions can include controlling releases or limiting exposure to waste streams or other media. Proposed releases can also be evaluated to determine whether the actual releases are acceptable in the environment or if they need to be regulated in some way. If the substance to be released is well studied, assessors can adapt existing assessments to evaluate the new situation [13, 10, 27]. When the release is a mixture of known compounds or substances having similar properties and suspected modes of action, assessors can reapply stressor-response relationships found in existing assessments to address the new situation. Information and lessons learned from completed assessments can also contribute to the development of emergency response plans with standard operating procedures. Applying accumulated knowledge ensures an efficient, reliable reaction process that restricts the spread of a pollutant and reduces exposure or harm from the unexpected releases. This knowledge also helps the assessor and manager later, when evaluating the release, to select a remedial action that minimizes unacceptable exposures or harm from the release and from the remediation process itself.

Access to completed assessments and a mechanism for applying them to new situations are essential for emergency preparedness. For aquatic systems, this has been accomplished by agencies in the U.S. and other countries that have adopted criteria for the protection of drinking water sources, recreational waters, wildlife, and other designated uses [8, 14, 20, 42]. Regulations that require setting acceptable levels of pollutants and that require monitoring to ensure that designated uses are retained have been enormously successful in improving or maintaining water quality despite allowing permitted discharges [40]. However, many pollutants enter the waterways from overland flow or from unregulated discharges, also referred to as pollution. In the U.S., programs instituted to reduce damage from unregulated discharges of a wide range of physical, chemical, and biological agents include the U.S. Department of Agriculture's incentive programs and the U.S. EPA's nonpoint source program [32] and total maximum daily load (TMDL) program [34]. Guidance for addressing chemical agents with toxicological modes of action dates back to the early years of environmental protection but is still evolving. Guidance for determining acceptable levels of agents with physical and biological modes of action have only recently been developed and applied. One of the most recent is the *U.S. EPA Framework for the Development of Suspended and Bedded Sediments* (*SABS Framework*) [36].

The U.S. EPA specifically developed the *SABS Framework* for uncontaminated sediment; however, assessors can adapt the overall process to any stressor and thereby develop WQC or set restoration goals. The foundation for the development of WQC was originally limited to controlled laboratory toxicity tests using fish, invertebrate, and plant species [26]. More recently, the criteria values have been fine-tuned by interpreting causal relationships developed from toxicity tests in the context of body burdens and wildlife exposures [28–31, 33]. The *SABS Framework* recommends using these methods but also encourages assessors to use knowledge from causal associations developed from field studies.

This more inclusive approach retains laboratory-derived knowledge about exposure-response relationships that is independent from other influences while also evaluating more types of effects than are practicable in controlled laboratory experiments alone. Field studies can include routine seasonal biological surveys or observations of field manipulations, such as changes following restoration. Because interventions have already achieved environmental goals in other places, using stressor-response relationships observed from previous field manipulations increases confidence that criteria or restoration goals will protect and improve aquatic resources. When the agent is already in the environment, an adaptive management approach can use monitoring results to inform and improve the assessment and the resulting criteria or restoration goals.

In order to combine different types of knowledge to evaluate options for criteria values or restoration goals, the SABS Framework recommends comparing results from several analytical methods applied to different datasets and endpoints. This approach is outlined below and can be considered a general method for developing criteria to be protective and restorative for any environmental resource subject to the detrimental effects of an agent. Then an abbreviated, hypothetical example (the development of WQC for sediments deposited on moderately steep-gradient streambeds with a gravel or cobble substrate) illustrates key steps and shows how that process can be applied.

Although sediment is a natural part of aquatic habitats, sediment quantity and characteristics can affect the physical, chemical, and biological integrity of streams, lakes, rivers, estuaries, wetlands, and coastal waters [2, 3, 36, 38, 43, 44]. Suspended sediments can impair a wide range of water uses:

- Suspended sediments clog filters that are used to finish drinking water and often reduce water clarity, thereby interfering with recreational uses.
- Decreased water clarity impairs visibility and affects many animal behaviors such as prey capture and predator avoidance, recognition of reproductive cues, and other behaviors that alter reproduction and survival [17, 18].
- At very high levels, suspended sediments can cause physical abrasion and clogging of filtration and respiratory organs [1].
- Suspended particles also decrease light penetration required for photosynthesis.

Excessive levels of suspended and bedded sediment and in some circumstances insufficient levels of those sediments can cause deleterious effects [25]. When sediments are contaminated, the combination of physical effects of sediment and toxic effect of contaminants are evaluated as distinct but related causes. However, because the development of chemical criteria for contaminated sediment already have well developed methodologies and applications [37], this chapter deals with only the physical effects of excess depositions of both inorganic and organic sediment to a stream bed (deposited and bedded sediment).

Sources of deposited and bedded sediments are soils and topsoil from land in the watershed or suspended sediment removed from stream banks and from the bed of an upstream channel. Some soils, such as volcanic ash, are more susceptible to movement. Generally, smaller, lighter particles move more readily and are easily resuspended. Slope, stream gradient, channel morphology, and other natural factors affect stream flow and, therefore, the ability to move sediments. Changes in watershed land cover may increase watershed erosion by increasing overland flow and the susceptibility of soil to movement. For example, during construction, vegetation is removed and soils are compacted, reducing permeability and increasing overland flow that carries disturbed soils from uncompacted areas into waterways [25].

2. Methodology

The SABS Framework [36] is a form of ecological risk assessment described in seven steps [20 21]. These seven steps (Figure 1) can be condensed into three phases: a Planning Phase, an Analysis Phase, and a final Synthesis Phase [5].

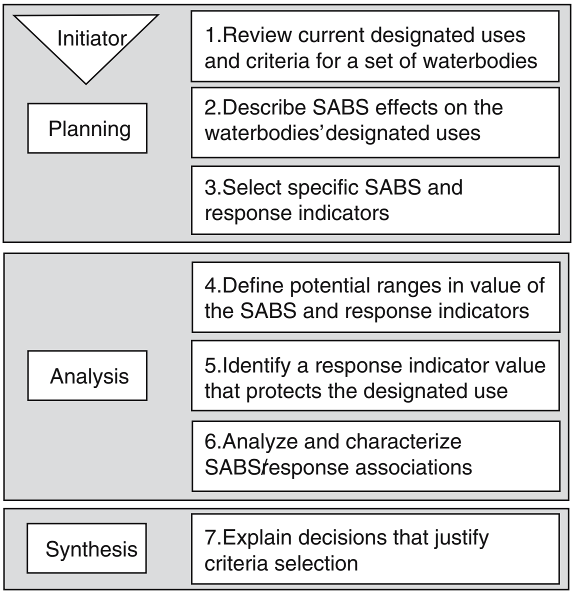

Figure 1. Phases of Assessment are Listed on the Left of the Seven Steps for Developing WQC [37].

The general process, as described here, primarily applies to the development of WQC but may also be considered a process to develop remediation goals.

Effect thresholds are selected based on scientific or legal precedent, stakeholder values, or other rationales. The effect threshold should protect the resource, retain its desired functions, and ensure safe conditions for wildlife and humans. The assessors should seek out readily available sources of information as well as datasets having the types of measurements that can be used to model stressor-response relationships. In some cases, new laboratory, field, or pilot studies may be necessary. Separate, independent studies are sought so that risk estimates can be compared and critiqued. For example, it is useful to compare results from different datasets, timeframes, or sub-samplings of datasets. The decisions of the planning phase are described in an analysis plan that guides the analysis phase. The plan should describe the objectives, datasets, and analytical approaches to be used. It should be appropriate for the environmental context of the assessment, the environmental value or use to be protected or remedied, the ecosystem type, and the measurements that represent the stressors and effects.

2.1. ANALYSIS PHASE

The objective of the Analysis Phase is to model the stressor-response relationship(s), develop an understanding of the mechanisms behind these relationships, and interpret their relevance to the environmental goals. To meet these goals, analysis results are used to answer questions like:

- What concentration of suspended sediment may occur without clogging filtration systems for a drinking water facility?
- What level of siltation can occur without adversely reducing fish spawning?
- When dredging a shipping channel, which timeframe would impose the least impact on commercially important species or their prey?

During the Analysis Phase, assessors:

1. Characterize the range and the relative acceptability of values for existing biological, environmental, and stressor conditions.
2. Quantitatively model the relationship between the stressor intensity and effects using data from laboratory studies or field observations.
3. Estimate candidate criterion values that are expected to protect against unacceptable conditions.

2.2. SYNTHESIS PHASE

In the Synthesis Phase, assessors compare the relationships developed from different datasets or study designs that result from the Analysis Phase with the effect thresholds that were identified in the Planning Phase. Decision makers can use the values of the stressor at the effect thresholds to determine acceptable levels for WQC or restoration goals.

3. Hypothetical Example

In this example, we develop WQC to regulate the amount of sediment deposited on moderately steep-gradient streambeds having a gravel or cobble substrate. The dataset used in this example is from the U.S. EPA Environmental Monitoring Assessment Program (EMAP) conducted in the Mid-Atlantic Highlands Assessment (MAHA) during the summers of 1993–1996 [39]. Data from laboratory tests were not included in this example because relevant test results were not found that could be used to estimate risks from deposited and bedded sediments.

3.1. PLANNING PHASE

In this example case, we reviewed several publications [1, 11, 43, 44] to study the effects of SABS on aquatic organisms. We used information from the reviews to develop a conceptual model that shows how SABS can affect invertebrate assemblages (Figure 2).

We considered four modes of action that lead to impaired invertebrate assemblages from increased levels of bedded sediment:

- Loss of suitable habitat
- Decreased dissolved oxygen
- Smothering
- Increased drift and predation

We developed deposited and bedded sediment criterion values for two levels of protection: aquatic life uses (ALU) and minimally acceptable aquatic life uses (MALU). We chose percent fines on the substrate as the bedded sediment metric because it is commonly used by many states. Also, good quality data were available, and acquisition protocols had been consistently applied across the entire dataset [35].

The metric of *Ephemeroptera, Plecoptera*, and *Trichoptera* (EPT) taxa richness was selected as the response measure because their diversity is a valued attribute and benthic aquatic invertebrates are prey for valued fish stocks [6, 12, 19, 23]. EPT taxa richness is strongly affected by sediment levels.

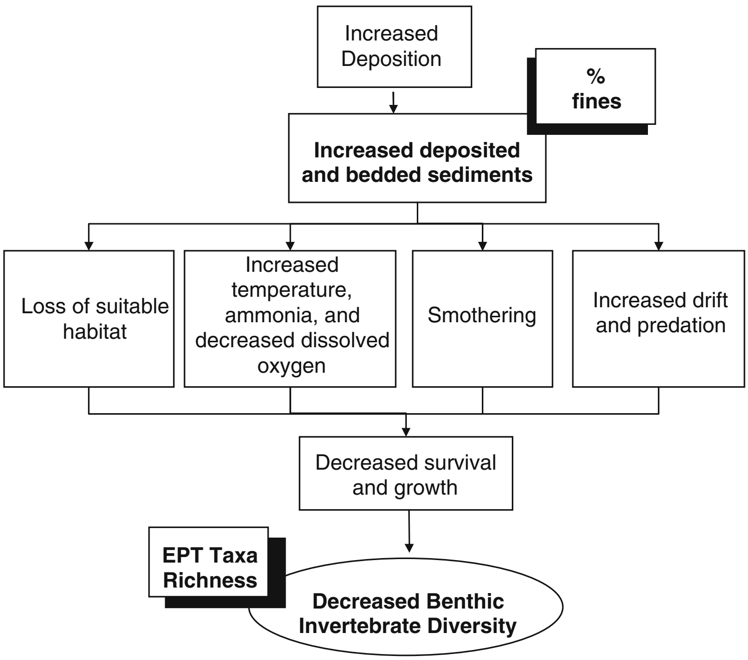

Figure 2. Conceptual Model of the Causal Relationship between Deposited and Bedded Sediments and Decreased Benthic Invertebrate Diversity.

It is accepted by regulatory agencies in most parts of the U.S., Canada, South Africa, New Zealand, Australia, Europe, and other places where it has become a commonly used metric within bioassessment indices that assess the condition of aquatic life [4, 15, 16]. Furthermore, data for EPT taxa richness were readily available for analysis and were judged to be of high quality, and the measures of EPT taxa richness could be compared with equally good quality measures of bedded sediment. Since the example is not based on actual state programs, there were no predefined biological criteria that quantitatively identified when aquatic life uses are not met. However, we did consider analyses from two independent datasets, one from the West Virginia Department of Natural Resources (DNR) that has a maximum of 15 EPT taxa at any site and another from the EMAP MAHA dataset with a maximum of 29 EPT [9]. The West Virginia DNR identifies 13 EPT taxa as meeting 100% use within its biocriteria index [9, 39]. Analyses of the EMAP MAHA dataset by Stoddard suggested characterization of condition based on ≤9 EPT taxa as poor, between 9 and 17 marginal, and ≥17 as good [7]. Because the EMAP MAHA data were used in this study and because that dataset had a greater observed maxima of EPT at sites, the values of ≤17 were applied to analyses of ALU and ≤9 EPT to MALU.

Biological effects thresholds for aquatic life uses were based on regulatory precedent, relative loss, and quantitative changepoints in stressor-response relationships (Table 1). Table 1 lists the type of evidence, the analytical method, and the risk estimation method.

TABLE 1. Example Candidate Thresholds of Biological Effect as Used in Hypothetical Example for SABS.

Basis	Evidence	Analytical method	Risk estimation method
Precedent [30]	SABS level for a proportion of streams with a given level of EPT taxa	Percentile	75% of streams ≥17 EPT taxa
Precedent [30]		Percentile	75% of streams ≥9 EPT taxa
Precedent [26]	Proportion of species affected	Species sensitivity distribution	5% of species reduced by 20%
Relative loss	Maximum expected for a SABS level	Quantile regression 90% level	5% reduction from y Intercept
Relative loss	Commonly achieved (mean) for a SABS level	Linear regression	20% reduction from y intercept
Changepoints	Statistical difference in slope (deviance reduction)	Conditional probability analysis	Change in slope from zero to >0

Three of the effect thresholds were based on current regulatory precedent; that is, threshold estimation methods that have been accepted and used by the U.S. EPA for criteria development. The percentile method is simply the SABS level measured at a stream that represents the 75th percentile of streams with an acceptable biological condition and was originally developed to derive WQC for nutrients [30]. In the hypothetical example, two effect thresholds were calculated using the percentile method: one for better quality (ALU) and one for fair quality (MALU) of biological conditions. Another method supported by precedent, species sensitivity distribution (SSD), has been used extensively for WQC for chemicals [26]. We developed a cumulative SSD for aquatic species based on field studies and calculated the level of SABS at which the 5th percentile of species are estimated to show a 20% reduction of abundance as observed in the data set for MAHA streams. This derivation used field associations and departs from the method of Stephan et al. [26], which uses laboratory toxicity tests to derive SSDs.

Biological effect thresholds that compared relative losses of species richness were calculated using linear and quantile regression methods. A 5% change was selected as a loss likely to be within a range of natural variation from forested areas (mean loss and maximum expected loss) and was applied to the ALU evaluation of the linear and quantile regression models. The effect threshold for MALU was set at 20% loss from currently attained conditions (mean and expected maximum).

Changepoints derived from conditional probability analysis (CCPA) plots were used to estimate when the probability of observing ≤17 for ALU and ≤9 for MALU began to increase. The changepoint was determined either from a change in slope of zero to a strong, positive slope (visually derived) or from a change that could be statistically detected.

3.2. ANALYSIS PHASE

3.2.1. Characterizing Biological and Exposure Conditions

Using the methods and thresholds chosen in the Planning Phase, we calculated the thresholds and analyzed the MAHA data to evaluate whether more than one criterion was necessary for different sizes of streams and stream types (Comment 3). Most values ranged from 0–36% fines for sites with >9 EPT taxa (Figure 3) and 0–10% fines for sites with >17 EPT taxa.

EPT taxa-richness values were also similar for drainage areas including those greater than 30 km^2 (Figure 4).

Therefore, we judged that sites could be grouped for the three drainage classes: <5, >5 <30, and >30 km^2. The range of values for heavily forested areas was from 0–50% fines compared to 0–100% fines when all sites were included (Figure 5).

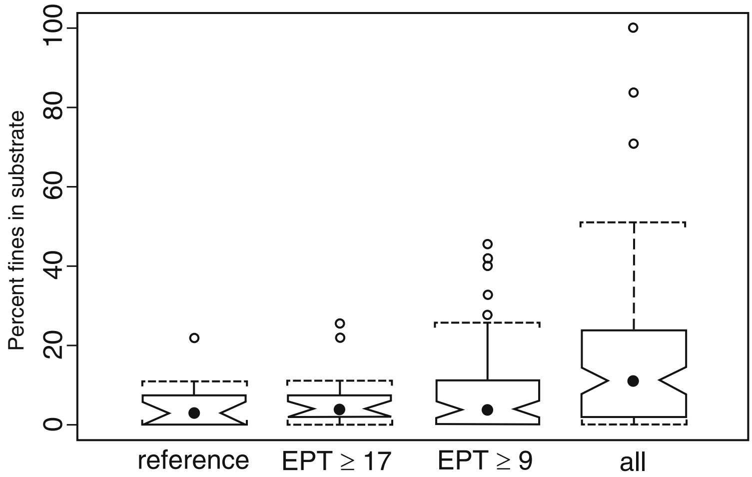

Figure 3. Notched Box and Whisker Plot. Reference Sites are Based on Land Use and Water Chemistry Parameters. The 75th Percentile for EPT >17 is 9.2% Fines and for EPT> 9 is 12.6% Fines.

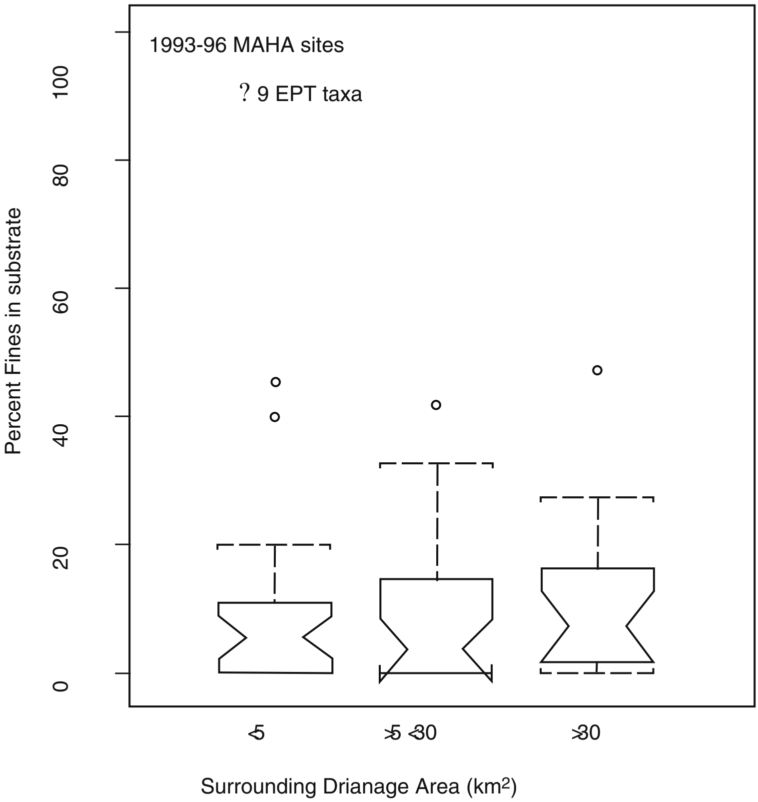

Figure 4. Notched Box and Whisker Plot of Percent Fines for Three Classes of Streams Based on Drainage Area for Lypothetical Example.

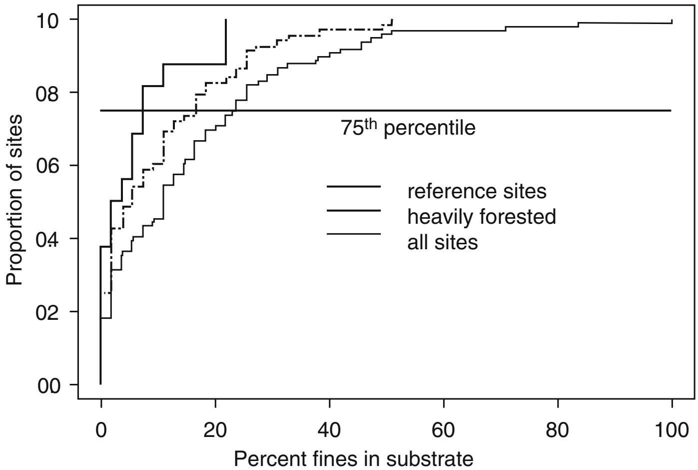

Figure 5. Cumulative Distribution Plot of Percent Fines for All Sites, Reference Sites, and Heavily Forested Sites [20, 21].

In developed but less stressed systems, we do not expect to see many sites with values in the upper range. The difference between observed amounts of percent fines in heavily forested areas and other areas suggests that both ALU and MALU criteria are necessary for a comprehensive management strategy; that is, distinct criteria for intact ecosystems and developed areas, which may need to set achievable restoration targets.

3.2.2. Develop Stressor-Response Models

As mentioned previously, we considered reviews [1, 11, 43, 44] for biological effects to invertebrates, fish, and plants from settled particles and bedded sediments. However, we could not find any suitable papers in these reviews or other published papers that quantitatively modeled for EPT taxa richness and were relevant to the MAHA data set. Therefore, we developed several stressor-response models to determine if bedded sediments were great enough to account for reductions of EPT taxa richness in streams of the mid-Atlantic and to estimate effect thresholds (Tables 1 and 2).

We estimated the proportion of streams that were affected by different levels of percent fines using the percentile method. Values were determined from cumulative distribution plots but could also have been estimated from box plots (Figure 3). The fraction of total streams was plotted against percent fines for sites with EPT taxa scores >17 and >9, and the level of percent fines at the 75th percentile of EPT taxa was determined (Figure 3). The effect threshold for ALU was 9.2% fines and for MALU was 12.6% fines.

TABLE 2. Evidence, Methods, Risk Estimation Methods for Developing Effect Levels Using Different Analytical Methods.

Evidence	Analytical method	Risk estimation method	% fines effect level	Risk estimation method	% fines effect level
		ALU		MALU	
Proportion of streams	Percentile	75th percentile	9.2	75th percentile	12.6%
Proportion of species affected	Species sensitivity distribution	5th percentile	7	Not selected	—
Maximum achievable	Quantile regression, 90% percentile	5 and 10%	5.8 and 11.5	15, 20 and 25%	17.3, 23.0, 28.8
Commonly achieved	Linear regression	5 and 10%	3.9 and 7.9	15, 20 and 25%	11.8,15.7, and 19.7
Changepoint analysis	Conditional probability analysis	Deviance reduction	8.2	Deviance reduction	10.1

We constructed an SSD to estimate the level of percent fines that could occur while still being protective of 95% of invertebrate species observed in MAHA streams [24]. We obtained the estimate from the cumulative distribution function of effect levels of species observed in MAHA streams (Figure 6).

The effect level was the value of percent fines at which each taxon's abundance was reduced by 20%. The maximum abundance was taken from quantile regression plots that modeled the 90th percentile of the relative abundance of several species of invertebrates [24]. The effect threshold for ALU was 7% fines. There was no precedent of a threshold for MALU; therefore, no effect level was estimated.

We determined the number of EPT taxa that were commonly observed at stream sites with different levels of percent fines by plotting the number of EPT taxa observed against percent fines and modeled using least squares linear regression analysis. We modeled the expected maximum number of EPT taxa that were likely to be observed at a site with different levels of percent fines using the 90th percentile from a quantile regression. We estimated the number of EPT taxa commonly encountered for a given SABS level from the linear regression curve. The amount of sediment associated with 5, 10, 15, 20, and 25% reduction from the y-intercept was determined from the 90th quantile and linear regression curves (Figure 7).

For ALU, a 5% reduction from the number of EPT taxa commonly observed was estimated to occur at 3.9% fines. A 5% reduction from the maximum

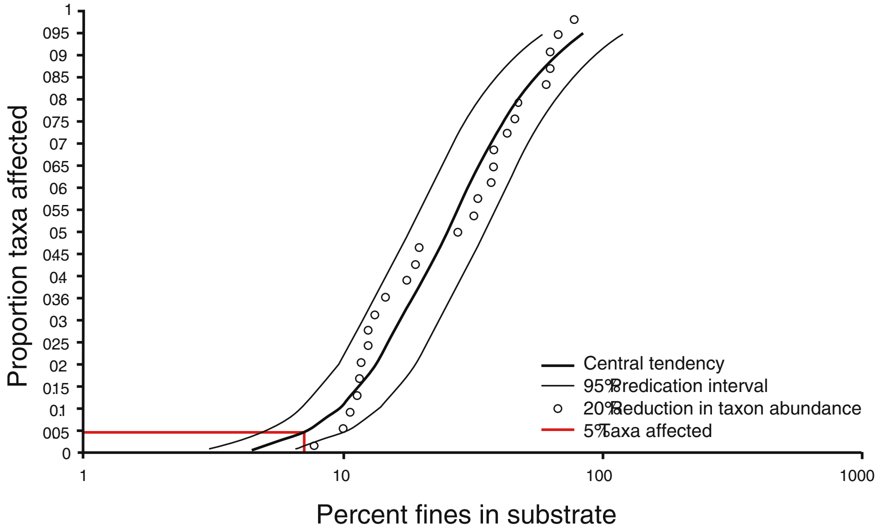

Figure 6. SSD Plots. The Abundance of 5% of the Species are Reduced by 20% at 7% Fines.

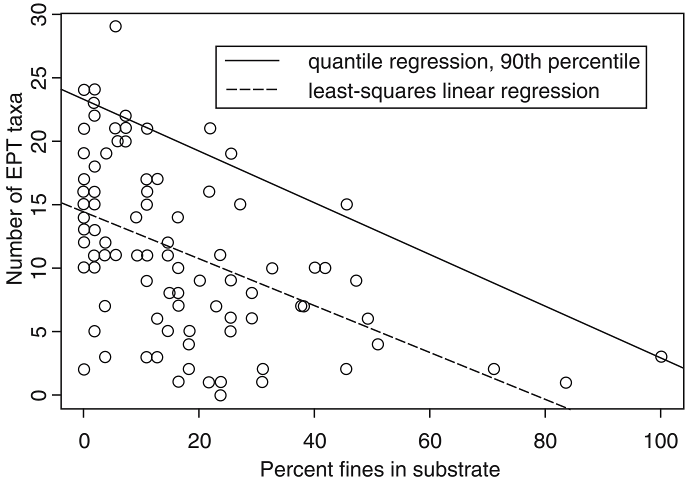

Figure 7. Scatter Wlot with 90th Percentile Quantile Regression (solid line) and Least-Squares Regression (Dashed Line).

number of EPT taxa was estimated to occur at 5.8% fines. For MALU, the 20% reduction from the number of EPT taxa commonly observed was estimated to occur at 15.7% fines. Also for MALU, a 20% reduction from the maximum number of EPT taxa was estimated to occur at 23% fines.

We used CCPA to estimate the probability of observing <17 and <9 EPT taxa richness for observed levels of percent fines. For ALU, the conditional probabilities for observing <17 EPT had a slope of zero from 0–7% fines (Figure 8).

From deviance reduction analysis, the changepoint occurred at 8.2% fines. For MALU, the slope of the probabilities of observing <9 sharply increased from 0% to about 17% fines; a statistically distinct difference was determined at 10.1% fines (Figure 9). Note that the point at the far left of Figures 8 and 9 represents the probability for observing <17 or <9 EPA for the entire range of percent fines (0–50% fines) and not the probability of observing <17 or <9 EPT at zero percent fines.

3.3. SYNTHESIS PHASE

3.3.1. Compare Risk Estimates

The recommendation for criterion values for the hypothetical case includes:

- **Aquatic life use (ALU)—criterion of no more than 7% fines.** This criterion is similar to existing precedents. Based on the proportion of species affected, 75% of sites with >17 EPT had an effect threshold at 9.2% fines. According to the results of the SSD analysis, 95% of EPT taxa would be protected most of the time when levels remained below 7% fines. There was an estimated

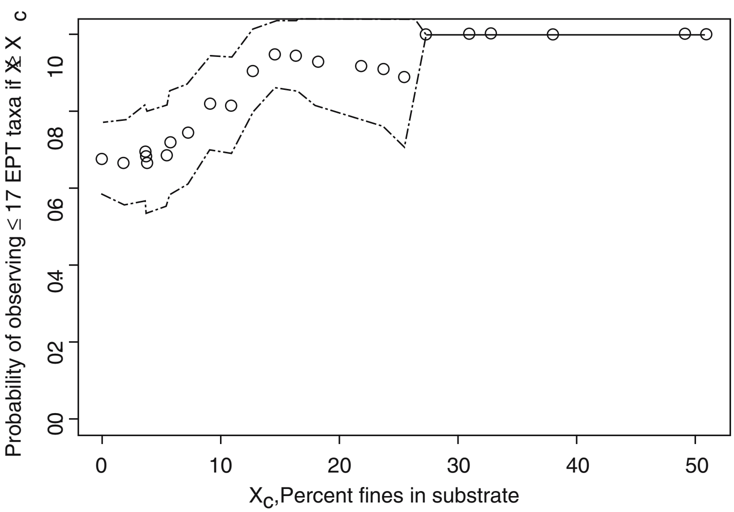

Figure 8. CCPA Plot of Probabilities of Observing <17 EPT Taxa for Different Levels of Percent Fines. Confidence Intervals Indicated by Hashed Lines.

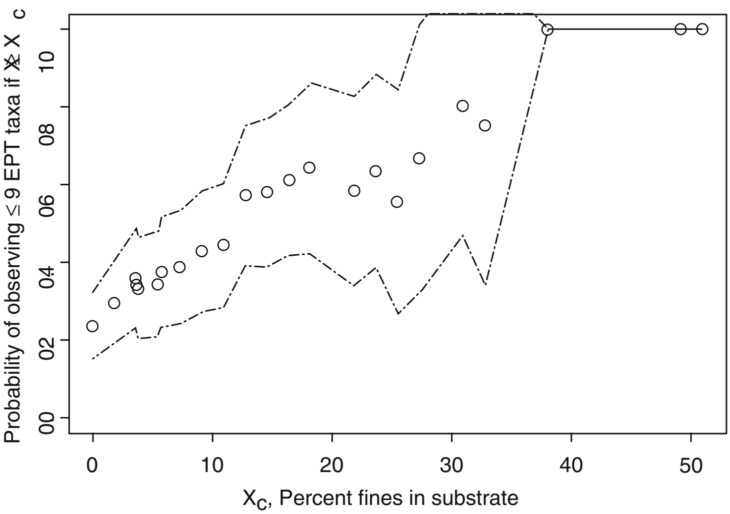

Figure 9. CCPA Plot of Probabilities of Observing <9 EPT Taxa for Different Levels of Percent Fines. Confidence Intervals Indicated by Hashed Lines.

loss of 5% from the maximum attainable number of EPT taxa at 5.8% fines and a 5% reduction for EPT values commonly observed at 3.9% fines. Furthermore, there was an increased probability of observing <17 EPT taxa at sites above 8.2% fines. Values for all methods were from 3.9–9.2% fines. A value of 7% fines was judged to be protective of the resource and conform to the most protective precedent, which was from the SSD method.

- **Minimally acceptable aquatic life use (MALU)—criterion of no more than 15% fines.** The minimal marginal conditions (<9 EPT taxa) were observed in 75% of the streams below 12.6% fines. There was an estimated loss of 20% from the maximum attainable numbers of EPT taxa at 23% fines, and a 20% reduction was estimated to be commonly observed at 15.7% fines. Furthermore, there was an increasing probability of observing <9 EPT taxa at sites from 0–17% fines and a statistically significant change-point at 10.1% fines. Values ranged from 10.1–23.0% fines. The mean effect threshold of all methods was 15% fines.

Table 3 summarizes the estimated effect level for percent fines based on the several methods evaluated for aquatic life use and marginal aquatic life use.

No criteria were developed for regulatory use in this case because this is a hypothetical example. Although real data were used in the examples, the resulting “criteria” should not be construed as a rigorous recommendation. Moreover, the criteria values were derived for bedded sediments and using only benthic invertebrates (EPT taxa richness), which is not likely to

TABLE 3. Summary of Effect Levels of Percent Fines Based on Five Analytical Methods.

Method	ALU	MALU
Percentile	7	12
SSD	7	
Quantile regression	5.8–11.5	17.3–28.8
Linear regression	3.9–7.9	11.8–19.7
Change point-conditional probability	8.2	10.1
Hypothetical candidate criteria values	7% fines	15% fines

be protective of overall designated use, even for Mid-Atlantic high-gradient streams. Additional assessment endpoints (e.g., coldwater fish production) would need to be considered along with EPT taxa richness and potentially other response measures to be confident that the criteria would be protective of all desired designated uses. Also downstream effects from transported sediment were not evaluated in a full risk assessment before selecting criteria.

4. Discussion

The SABS Framework provides a scientifically defensible approach for identifying effect thresholds that is useful for nontraditional modes of action and risks. Because the approach compares results from several analytical methods, there may be greater confidence in the decision, and the expectations of potential outcomes from actions may be more realistic. Also, for nontraditional stressors, statutory and legal precedents have not been time-tested and knowledge from several corroborating methods strengthens an assessor's credibility and the resulting decision.

The percentile method has precedent for nutrients, a nontraditional stressor [30]. The precedents for SSDs are strongly supported by legal and statutory precedent; however, the precedent is based on controlled laboratory toxicity tests while the analyses described here were based on a novel application using field observations [24]. As such, the precedent of the 5th percentile is reasonable and informative but not a precedent that has been fully reviewed by either the scientific community or the courts. Likewise, estimates based on relative loss were not grounded in legal precedent but do provide reasonably objective technical information for evaluating the impact of selecting different levels of percent fines as criteria. The values based on deviations from maxima and median values were comparable with other estimates. The statistically based changepoint analysis is objective and repeatable, but there is no known legal or statutory precedent for its use.

Both rapid and deliberative decision making can be informed by predeveloped risk estimates of known or commonly occurring stressors. When there is an emergency threatening life, property, or irreplaceable natural resources, assessors can expeditiously use available criteria, and the stressor-response models on which they are based, to estimate immediate effects and continued risks as management actions attempt to control deleterious effects. When time is not crucial, a slower, more deliberative gathering of information to support decision making is possible and preferred. This process can accommodate time to find and assure the quality of datasets, seek published stressor-response models, and even implement new data collection and analysis. This is the approach illustrated in this chapter. However, we recognize that the selected criteria could also include thresholds for total loss of the resource, which could be valuable for emergency situations. Also, the stressor-response models could be quickly reanalyzed for other purposes that might not be recognized until the situation arises. Therefore, it is good scientific practice to make stressor-response models and datasets open to others rather than to simply publish final values.

Most existing risk estimates assess exposures to single chemicals [26]. However, wildlife can be harmed by nontraditional stressors for which most toxicity test methods are not suitable. The SABS Framework was developed for determining effect thresholds for an agent with a mode of action that causes physical abrasion, reduction in water transparency, burial, and alteration of substrates that make them unsuitable habitats for aquatic life. Laboratory toxicity tests are not capable of evaluating these modes of action. Therefore, the SABS Framework combines techniques using toxicity tests developed by the U.S. EPA's WQC program along with an expanded repertoire of analytical tools and approaches. By using different datasets, different endpoints, and different analytical methods, systematic biases, which might have been overlooked, can be qualitatively evaluated in the synthesis phase. Overall, this approach ensures that credible scientific input will inform decision making that is more likely to protect the environment and the functions it provides to protect all life.

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This paper has not been subjected to Agency review; therefore, it does not necessarily reflect the views of the Agency. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

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