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AN EVALUATION OF PHYSICAL STREAM HABITAT ATTRIBUTES USED TO MONITOR STREAMS¹

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ABSTRACT: The last few decades have seen an increased reliance on the use of stream attributes to monitor stream conditions. The use of stream attributes has been criticized because of variation in how observers evaluate them, inconsistent protocol application, lack of consistent training, and the difficulty in using them to detect change caused by management activity. In this paper, we evaluate the effect of environmental heterogeneity and observer variation on the use of physical stream attributes as monitoring tools. For most stream habitat attributes evaluated, difference among streams accounted for greater than 80 percent of the total survey variation. To minimize the effect that variation among streams has on evaluating stream conditions, it may be necessary to design survey protocols and analysis that include stratification, permanent sites, and/or analysis of covariance. Although total variation was primarily due to differences among streams, observers also differed in their evaluation of stream attributes. This study suggests that if trained observers conducting a study that is designed to account for environmental heterogeneity can objectively evaluate defined stream attributes, results should prove valuable in monitoring differences in reach scale stream conditions. The failure to address any of these factors will likely lead to the failure of stream attributes as effective monitoring tools.

(KEY TERMS: aquatic ecosystems; quality control; random effects model; sample size; stream monitoring; environmental heterogeneity.)

INTRODUCTION

The past several decades have seen a proliferation of survey methods designed to describe and evaluate stream characteristics at the reach scale (Bain *et al.*, 1999; Bauer and Ralph, 1999; Johnson *et al.*, 2001). These protocols are often used to monitor anthropogenic mediated changes in stream's conditions (MacDonald *et al.*, 1991; Kaufmann *et al.*, 1999). Evaluations of physical stream habitat frequently occur in lieu of monitoring biological conditions (i.e.,

fish density and biomass) because of the relative ease associated with collecting these data and the variability of biological systems (Platts *et al.*, 1983). The outcomes of these surveys are used as evidence for compliance or noncompliance with federal laws (NMFS, 1996; Bauer and Ralph, 2001), federal land management plans (Espy and Babbitt, 1994), and state laws (Washington State, 2001).

The use of stream attributes for monitoring has many critics. The most common critique has been that different observers using the same protocol often arrive at different results (Ralph *et al.*, 1992; Roper and Scarnecchia, 1995; Poole *et al.*, 1997). Additional concerns include inconsistent application of protocols (Hey and Thorne, 1983; Kondolf, 1997), lack of consistent training (Wang *et al.*, 1996), and difficulty in using stream attributes to detect change caused by management activity (MacDonald *et al.*, 1991; Kaufmann *et al.*, 1999; Bauer and Ralph, 2001). At the root of each of these concerns is the variability associated with the measurement of an attribute and how it affects conclusions relative to that attribute.

Variability in how observers evaluate a stream metric has been used to justify discarding the metric in monitoring protocols (Roper and Scarnecchia, 1995; Poole *et al.*, 1997; Bauer and Ralph, 2001). Variability, however, is a component of all sampling protocols (Ramsey *et al.*, 1992; Clark *et al.*, 1996) and is often an important aspect of understanding natural systems (Green, 1984; Frissell and Bayles, 1996). Therefore, instead of using variability as a rationale for discarding an attribute, we suggest that an understanding of variability be included in the design of a monitoring program and in the selection of attributes used in that program (Larsen *et al.*, 2001).

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Variability in the measurement of stream attributes can be divided into three sources: environmental heterogeneity, sampling variance, and measurement error (Ramsey *et al.*, 1992; Clark *et al.*, 1996). For example, consider a study comparing median riffle particle size (D_{50}) between a set of streams where cattle are permitted to graze and a set of streams where cattle are excluded. The ability to discern differences will be affected by: (1) environmental heterogeneity due to the substrate differences among the streams surveyed, (2) sampling variance due to differences in how and where different observers pick up particles to be measured, and (3) the measurement error due to variability associated with how an observer measures an individual rock. Previous papers evaluating stream attributes have primarily addressed the observer (sampling variance + measurement error) portion of variability (Marcus *et al.*, 1995; Wang *et al.*, 1996; Poole *et al.*, 1997; and for an exception see Kaufmann *et al.*, 1999).

The objective of this paper is to evaluate how variability in 13 common physical stream attributes affects their use in monitoring stream conditions. To meet this objective we evaluate the relative importance of variability among streams (environmental heterogeneity) and among observers (both sampling and measurement error), use estimates of total variation to determine sample sizes necessary to detect specified differences between group means of streams treated differently, and suggest variables likely to prove useful in monitoring changes in stream habitat.

METHODS

Study Site

Data for this study were collected during the summer of 2000 on lands managed by the USDA Forest

Service within central Idaho. A single reach in each of six streams was surveyed. Stream reaches were chosen to represent a range of low gradient (less than 2 percent slope) wadable streams within managed watersheds in central Idaho. Within these constraints, streams varied in size, elevation, and geology (Table 1).

Study Design

Stream reaches were independently surveyed with either six or seven two-person crews. All crew members in this study received ten days of training prior to completing surveys. Training consisted of collecting data while under the supervision of individuals who had designed the survey protocol. The intent of training was to minimize variability in how crews assessed stream attributes. Each two-person crew was responsible for estimates of the stream attributes evaluated for this study. Estimates of observer variability were based on the variability among these crews.

Test surveys were conducted during two one-week sessions: one in June and the other in August. Each session involved all crews evaluating a reach in each of three streams. The order in which crews sampled stream reaches was randomly assigned. Crews began surveying each creek in identical locations and sampled by moving upstream a minimum distance of 20 times the bankfull width. Crews were asked to stop surveying at the bottom of the first pool they encountered after surveying the minimum distance. Although a prespecified ending location could have been used, minor variation in where observers end surveys is an unavoidable aspect of most surveys, so evaluation of these differences is included in the analysis (see Table 1). No precipitation fell during either session, so stream behavior remained relatively consistent throughout each test period.

TABLE 1. General Characteristics of Streams Evaluated for This Paper.

Stream	Geology	Elevation (m)	Gradient	Bankfull Width (m)	Reach Length	
					Minimum (m)	Maximum (m)
Big	Granitic	1939	0.30	4.9	127.9	133.5
Jack	Granitic	1610	0.71	2.6	81.2	88.6
Meadow	Granitic	976	0.37	7.8	120.2	128.0
Boulder	Volcanic	1448	0.37	7.3	162.0	180.0
Little Goose	Volcanic	1524	1.58	3.9	84.0	91.9
Lost	Volcanic	1480	0.65	5.4	119.6	128.0

Physical Characteristics

Observers from these six or seven crews made independent observations relative to 13 physical attributes within each reach. These attributes are commonly evaluated physical characteristics in stream monitoring protocols (Platts *et al.*, 1987; MacDonald *et al.*, 1991). The sampling methodologies originated from existing protocols. Some methodologies were refined following two years of use and evaluation (Archer *et al.*, 2003). Additional changes were made to fit within the project study design of determining a stream reach average for each attribute. The original citation for each methodology and a brief description of the protocol are given below.

Gradient. Reach gradient was calculated by dividing the average elevation change (measured to the nearest centimeter) by reach length (measured to the nearest 0.1 m) (Harrelson *et al.*, 1994). Elevation change was estimated as the average of two independent measurements of elevation change using a 20X level transit. Reach length was measured along the thalweg using a tape measure.

Sinuosity. Sinuosity was estimated by dividing the reach length by the straight line distance of the reach. The straight line distance was measured with a tape measure between the start and end of the reach. Both straight line and reach length were measured to the nearest 0.1 m.

Streambank measurements were used to estimate a reach average for bank angle, undercut depth, percent undercut, and bank stability. Bank angle measurement techniques were taken from Platts *et al.* (1987) and bank stability from Bauer and Burton (1993). Measurements were taken from sample plots on both banks at a minimum of 20 equally spaced (every bankfull width) transects throughout the reach.

Bank Angle. The average bank angle was calculated as the average of the bank angles measured at each sample plot. Bank angles were measured by laying a depth rod along the bank at right angle to the flow. A clinometer was placed along the depth rod and the angle recorded to the nearest degree.

Percent Undercut. The percent of a reach with undercut banks was calculated by dividing the number of sample plots that were undercut by the total number of plots. Undercut banks had an angle less than 90 degrees and an undercut depth of greater than 5 cm.

Undercut Depth. The average undercut depth was calculated by summing the depths measured at each of the undercut bank locations and then dividing by the total number of sample plots. Undercut depth was measured to the nearest centimeter from the deepest part of the undercut to the outer edge of the streambank.

Percent Stable Banks. The percent of a reach with stable banks was calculated by dividing the number of sample plots with stable banks by the total number of plots. Each plot was considered stable if there was no sign of erosion, slumps, or fractures.

Bankfull Width. The bankfull width was calculated by averaging the width estimates from four cross-sections. The cross-sections were located at the widest point in four different fast water habitat units and recorded to the nearest 0.1 m. Bankfull widths were estimated at fast water habitat units instead of in a straight reach between two meander bends as suggested by Harrelson *et al.* (1994). This modification was made in an effort to increase the consistency in where crews took measurements. Bankfull indicators as described by Harrelson *et al.* (1994) were used to determine bankfull elevation.

Width-to-Depth Ratio. The average width-to-depth ratio was determined from the same four cross-sections used to determine bankfull width. The average depth at each cross-section was estimated from at least 10 evenly spaced bankfull depths instead of the 20 recommended by Harrelson *et al.* (1994). Fewer measurements were taken at each cross-section because our surveys were interested in width-to-depth ratios only, not accurately describing channel profiles. Depth was recorded to the nearest centimeter. The width-to-depth ratio at each cross-section was calculated as the bankfull width divided by the average depth. An overall reach average was calculated from these four cross-sections.

D₅₀. The D₅₀ was the intermediate axis diameter of the median particle collected from particle counts (Wolman, 1954) within four fast water habitat units in the reach. At least 25 bed material particles were collected along transects within each unit for a minimum of 100 particles. Particles were measured to the nearest millimeter.

Percent Fines. The percent fines was the percent of the substrate collected for the estimation of D₅₀ that had an intermediate axis diameter less than 6 mm.

Pool Tail Fines. The percent of surface fine sediment was averaged from four pool tails. A 50 intersection grid (Bunte and Abt, 2001) was randomly tossed three times within each pool tail. The percent fines were calculated by dividing the number of intersections underlain with fine sediment (less than 6 mm) by the total number of intersections (150).

Percent Pools. The percent of a reach comprised of pool habitat was estimated by summing the lengths of each pool and dividing by the reach length. Pool lengths were measured along the thalweg.

Residual Pool Depth. The average residual pool depth was calculated by averaging the individual residual depths from all pools encountered in the reach. Residual depth for each pool was derived by subtracting the pool tail depth from the maximum depth (Lisle, 1987). Depth at both pool tail and maximum were recorded to the nearest centimeter.

Analysis

A random effects analysis of variance model (PROC MIXED of SAS®) was used to estimate the means and variance associated with streams (stream variability) and crews (observer variability). Estimates of variance were evaluated by treating both stream and observer as a random effect within the model (Littell *et al.*, 1996). We used a random effects model because we wished to generalize results to a larger population of streams and observers. All error in the model not associated with difference among streams was assumed to be due to differences among observers.

Estimates of variance for streams and observers were additive and could be used to calculate the proportion of the variation due to each component (Montgomery, 1984; Ramsey *et al.*, 1992; Clark *et al.*, 1996). Estimates of the variance and means also were used to calculate the coefficient of variation [CV (variance)^{0.5}/mean*100] for each variable.

We calculated minimum sample sizes necessary to detect differences between two population means (Zar, 1996) to provide guidance on which variables would be best used in monitoring differences in stream conditions among groups of streams treated differently (e.g., grazed vs. ungrazed). Sample size estimates are a good tool to evaluate monitoring attributes because they indicate the effort needed to be confident changes in an attribute will be detected (Eckblad, 1991; Hogle *et al.*, 1993).

Minimum sample sizes were calculated using specified differences between two means – Type I and Type II error rates. We evaluated differences between means that ranged from 5 to 50 percent. For this

exercise, the value of 5 percent was chosen for sampling practicality (Eckblad, 1991). Variation in ecosystems is high, so detecting differences of less than 5 percent almost always requires hundreds if not thousands of samples. The upper value, 50 percent, was chosen because when differences among groups are of this magnitude, they can affect land management activities thought to influence them (Espy and Babbitt 1994; see Table 1 in NMFS, 1996).

In calculating sample sizes, we were concerned with two types of statistical errors: (1) a Type I error (α), which is rejecting the null hypothesis (no mean difference between groups) when the null hypothesis is actually true; and (2) a Type II error (β), which is failing to reject the null hypothesis when it is actually false (Peterman, 1990). We limited our evaluation to a single Type I error rate, $\alpha = 0.1$, and two Type II error rates, $\beta = 0.1$ and $\beta = 0.25$. Type I errors are commonly reported in the literature. Knowing the Type II error rate helps ensure that findings of no significant difference between groups are not used as evidence that the null hypothesis is true unless the test also has high statistical power (Robinson and Wainer, 2002).

Estimates of sample size were calculated following the iterative procedure outlined by Zar (1996, pg. 133, Equation 8.22):

$$n \geq \frac{2 s^2}{d^2} (t_{\alpha(2),v} + t_{\beta(1),v})^2 \quad (1)$$

where s^2 is our estimate of total variance; $t_{\alpha(2),v}$ is the t-value for an acceptable two-tailed Type I error rate; $t_{\beta(1),v}$ is the t-value for an acceptable one tailed Type II error rate; v is $2*(n-1)$ the degrees of freedom for s^2 ; and d is the minimum difference one wishes to detect. This equation calculates the number of samples needed from each population assuming equal sample sizes and equal variances. If the number of samples from one population is constrained (e.g. few unmanaged stream reaches), it is necessary to adjust the sample size of the unconstrained population (Zar, 1996). If variance of the two groups is unequal, an estimate of pooled variance should be used. When n exceeded 30, values for infinite sample size were substituted (z-values for t-values) because differences in results were minimal.

All analyses were conducted using raw data. Three variables – gradient, width-to-depth, and D_{50} – were skewed, but because of the small number of streams sampled, statistical models being robust to departures from normality (Zar, 1996), no underlying theory suggesting data should be transformed (for discussion see Cohen and Cohen, 1983), and raw data resulting in conservative (larger) estimates of variance, these

attributes were not transformed. For example, gradient of the six streams we surveyed could be described as a log normal distribution (see Table 1). There is, however, no reason to think the distribution of all stream reaches between 0 and 2 percent gradient would have this same distribution. If the underlying distributions of these attributes are truly skewed, transforming data could reduce sample sizes needed to detect mean differences between groups.

RESULTS

The results of this study suggest that many commonly evaluated physical characteristics of streams can have considerable variation when measured, and variation can be attributed both to differences among streams and differences in how observers evaluate streams. This variation has important implications on the design and interpretation of monitoring programs based on physical stream habitat variables.

Percent of Total Variation

In 10 of the 13 physical stream attributes evaluated, variation among streams accounted for 80 percent or more of the total variation (Table 2). In one variable, percent fines, stream variation accounted for about two-thirds of the total variation. Variation in two variables – percent stable banks and percent

pools – was nearly equally divided between streams and observers.

Coefficient of Variation

Coefficients of variation (CV) for a given variable were considerably higher among streams than among observers. Estimates of variation resulted in the CVs ranging from 5 to 72 among streams and 5 to 38 among observers (Table 2). Coefficients of variation for all but three variables – bank stability, percent fines, and percent pool – were considerably greater among streams than among observers.

High CVs among streams indicated that values of the measured attributes varied greatly among the sampled streams. Only percent stable banks showed little variation among the streams sampled (CV less than 10). Four variables – sinuosity, bank angle, percent pool, and residual pool depth – had CVs between 10 and 25. The remaining attributes all had stream CVs indicating that the average deviation was at least one-third of the mean. The most variable attribute among streams was gradient. This was true even though we only evaluated stream with gradients less than 2 percent. Within this narrow category of streams, the standard deviation among streams still was nearly three-fourths of the mean.

High CVs among observers indicated disagreement in how observers evaluated an attribute. When observer CV is zero, all observers evaluate the attribute in the same manner. Four variables – gradient,

TABLE 2. Means, Standard Deviations (stdev), Percent of Variation (% Var), and Coefficient of Variation (CV) Associated With the 13 Physical Stream Attributes Measured in This Study.

Stream Attribute	Total Mean (stdev)	Stream		Observer	
		% Var	(CV)	% Var	(CV)
Gradient (percent)	0.66 (0.4811)	99	(72)	1	(8)
Sinuosity	1.95 (0.3303)	86	(16)	14	(6)
Bank Angle (degrees)	96.44 (21.0778)	85	(20)	15	(8)
Undercut Depth (m)	0.10 (0.0446)	83	(39)	17	(18)
Bank Undercut (percent)	40.92 (19.5487)	86	(44)	14	(18)
Bank Stability (percent)	91.80 (6.5506)	53	(5)	47	(5)
Bankfull Width (m)	5.30 (2.0768)	88	(37)	12	(13)
Width:Depth Ratio	14.57 (6.2364)	85	(40)	15	(16)
D50 (mm)	25.03 (16.7820)	89	(63)	11	(22)
Percent Fines	31.61 (21.4809)	69	(57)	31	(38)
Pool Tail Fines (percent)	26.16 (20.0107)	80	(69)	20	(34)
Percent Pools	69.98 (13.1694)	44	(13)	56	(14)
Residual Pool Depth (m)	0.37 (0.0905)	80	(22)	20	(11)

sinuosity, bank angle, and bank stability – had CVs that were not appreciably greater than zero (CV less than 10). Approximately half of the attributes had CVs between 10 and 25. All three substrate attributes – D₅₀, percent fines, and pool-tail fines – all had CVs in excess of 20.

Sample Size

Of the 13 variables evaluated using protocols specified in this study (Table 3), only percent stable banks could discern small changes (5 percent) with sample sizes of fewer than 50 per group. If the desired minimum detectable difference between means was 10 percent and the Type II error rate increased to 0.25, sample sizes of fewer than 65 from each group would be acceptable for five – sinuosity, bank angle, bank stability, percent pools, and residual pool depth – of the 13 variables. Four attributes – gradient, D₅₀, percent fines, and pool-tail fines – consistently required twice as many samples as the remaining attributes to detect the same magnitude of difference between groups.

DISCUSSION

Most studies evaluating stream attributes as a monitoring tool have focused on variability associated with observers (Hogle *et al.*, 1993; Roper and Scarnecchia, 1995; Wang *et al.*, 1996). These studies found high observer variability and suggested that monitoring by using stream attributes either be discontinued or include more operational definition of stream attributes and additional training of observers (Platts *et al.*, 1983). Our study suggests that while minimizing variability among observers is a necessary step in designing monitoring protocols (Peterson and Wollrab, 1999), it may be difficult to totally eliminate this source of variation. Even with the 10 days of training used in this study, the standard deviation for observers in 9 of the 13 variables exceeds 10 percent of the mean and two of the variables have standard deviations exceeding 30 percent of the mean value.

With extensive training and protocol development, we did find that for 10 of the 13 attributes, observer variability accounted for 20 percent or less of the total variability. Studies evaluating water chemistry (Clark *et al.*, 1996), geochemistry (Ramsey *et al.*, 1992) and stream attributes (Kaufmann *et al.*, 1999) have suggested that when sampling variance (variation among

TABLE 3. Minimum Sample Sizes Needed to Detect Difference Among Physical Variables When Both Observer Variability and Stream Heterogeneity are Accounted for. Sample size estimates assume equal sample size and variance (Zar, 1996). Values listed below indicate half the total sample needed. The first value listed in each column is the sample size needed to detect the stated change with a Type I error of 0.1 and a Type II error of 0.1. The value in parentheses has the same Type I error rate but a Type II error rate of 0.25.

Stream Attribute	Sample Sizes Needed to Detect Changes of				
	5 Percent	10 Percent	20 Percent	30 Percent	50 Percent
Gradient (percent)	3,611 (2,268)	903 (567)	226 (142)	101 (63)	37 (24)
Sinuosity	198 (125)	50 (32)	14 (9)	7 (5)	3 (2)
Bank Angle (degrees)	328 (206)	82 (52)	22 (14)	10 (7)	5 (3)
Undercut Depth (m)	1,282 (806)	321 (202)	81 (51)	36 (24)	15 (9)
Bank Undercut (percent)	1,564 (938)	391 (246)	98 (62)	44 (29)	17 (11)
Bank Stability (percent)	35 (23)	10 (7)	4 (3)	3 (2)	2 (2)
Bankfull Width (m)	1,054 (662)	264 (166)	66 (42)	30 (20)	12 (8)
Width: Depth Ratio	1,266 (789)	314 (198)	79 (50)	35 (23)	14 (9)
D50 (mm)	3,081 (1,935)	771 (483)	193 (121)	86 (54)	31 (21)
Percent Fines	3,165 (1,988)	792 (497)	198 (125)	88 (56)	32 (21)
Pool Tail Fines	4,011 (2,519)	1,003 (630)	251 (158)	112 (70)	41 (26)
Percent Pools	243 (153)	61 (39)	17 (11)	8 (6)	4 (3)
Residual Pool Depth (m)	405 (255)	102 (64)	27 (17)	13 (8)	6 (4)

observers, noise) accounts for 20 percent or less of the total variability, it is likely to be a reliable monitoring attribute. Using this criterion, all but percent stable banks, percent fines, and percent pools should prove reliable attributes to measure when monitoring.

The problem with relying on the percent total variability attributable to differences among observers (or signal-to-noise ratio as proposed by Kaufmann *et al.*, 1999) as the sole guide for selecting monitoring attributes is that this measure fails to account for the total variance associated with the measurement of an attribute. For example, observer variability accounted for 20 percent of the total variability associated with evaluating pool tail fines. But the CVs associated with pool tail fines indicated that the average deviation among observers was 34 percent of the mean value, while the average deviation among streams was 69 percent of the mean value. Of all the attributes evaluated, this variable required the largest sample size to detect differences among sets of stream (Table 3). If one of the objectives of a monitoring program is to detect differences among treatments (when they exist) at the lowest cost, then both the total variance (sample size) and the proportion of the variability due to observers needs to be of concern.

Our data indicate that for most physical attributes, variability among streams has a far greater consequence on monitoring efforts than do differences among observers. The sample size equation is calculated using total variance (s_{total}^2), which is the sum of the variability due to streams (s_{stream}^2) and observers ($s_{observer}^2$). This means the proportion of the total sample due to stream and observer is directly related to its proportion of the total variability. This suggests that for all but three variables – bank stability, percent fines, and percent pools – the overwhelming majority (greater than or equal to 80 percent) of the samples needed in a monitoring program are driven by differences among streams not differences among observers.

Stream differences are a product of large-scale geomorphic features (Isaak, 2001; Jensen *et al.*, 2001), the position of the stream within the watershed (Montgomery and Buffington 1997), effects of management (Ralph *et al.*, 1993, Wood-Smith and Buffington, 1996), and disturbance history (Reeves *et al.*, 1995). Monitoring efforts that rely on describing physical stream attributes must therefore focus additional effort on minimizing the effect that variability among streams has on addressing the monitoring objective. Variation among streams can be reduced through stratification, measuring attributes at permanent sites, and/or incorporating analysis of covariance into comparisons.

Stratification can appreciably reduce variation and result in more precise estimates of an attribute

(Montgomery, 1984). In a stream monitoring program, stratification should focus on minimizing the effect that stream formation processes have on differences in stream attributes. Landscape characteristics that may prove useful in stratifying include: geology (Hicks, 1989), watershed size (Lee *et al.*, 1997), channel type (Rosgen, 1994; Montgomery and Buffington, 1997), and anthropogenic disturbance level (Dose and Roper, 1994; McIntosh *et al.*, 2000). Characteristics used to stratify stream systems will depend upon the objective for which the data are collected. The primary disadvantage of stratification is that it can result in limiting the number of sampling units for some strata.

A second strategy for reducing variation among streams is sampling at permanent sites. The use of permanent sites has the potential to reduce the variance and therefore the number of samples needed when there is a high degree of correlation between sampling unit values through time (Elzinga *et al.*, 1998). The disadvantage of permanent sites is that more effort is focused in fewer locations, so results may have limited applicability toward answering larger scale questions.

A third strategy for reducing variation among streams is using analysis of covariance (ANCOVA). The use of ANCOVA removes variability in an experimental unit that the survey design could not control (Littell *et al.*, 1996). This is a common occurrence in studies on the effects of land management. For example, managed watersheds can differ from unmanaged watersheds in their gradient, basin area, and elevation. Because these differences are difficult to account for through stratification (i.e., few large, low elevation unmanaged watersheds), ANCOVA may be the only tool to reduce the amount of variability among streams. ANCOVA is best applied to variables unaffected by the process under study such as gradient, geology, or elevation in the case of evaluating the effects of land management. Inferences based on ANCOVA can be difficult to interpret if there is considerable error in measuring the attribute used as the covariate or when there are sizable differences between means of the evaluated groups (Dunn and Clark, 1987).

Although total variation, how it is partitioned between streams and observers, and CVs, all provide important information necessary in selecting an attribute to be evaluated, these cannot be the only justification for deciding which stream attributes to monitor. Selection of variables also must include an understanding of how relevant the attribute is to the biota of concern, how rapidly the attribute can change, and how land management might alter the attribute (Robison, 1997; Bauer and Ralph, 2001; Montgomery and MacDonald, 2002).

Of the attributes we evaluated, only those associated with describing the stream substrate (D_{50} , percent fines, pool tail fines) may prove impractical in evaluating stream conditions subject to different treatments. All three of these attributes had high total variation, resulting in much higher total sample sizes than most other variables (Table 3). Our study suggests that alternative protocols need to be developed to more precisely describe stream substrate at the reach level (Kondolf, 1997; Bunte and Abt, 2001).

CONCLUSION

The primary objectives of this study were to evaluate variation associated with the measurement of physical stream characteristics and to suggest variables that would be useful in monitoring physical stream habitat characteristics. Our evaluation of these 13 attributes suggests that given extensive training and protocol development, all but three (D_{50} , percent fines, and pool tail fines) would likely prove useful in discerning differences in streams subject to different treatments.

Due to high total variance, most attributes are best suited for well designed, well funded monitoring programs. In contrast, these attributes are ill suited for the quick evaluation of single streams by poorly trained observers. The success of monitoring studies between these two extremes will be based on formulating specific monitoring objectives (Kershner *et al.*, 1991), careful selection of stream attributes based on the estimates of variation found in this study, pilot studies conducted in the region of interest, an understanding of the magnitude of change the attribute is likely to exhibit (Montgomery and MacDonald, 2002), and sufficient resources to conduct the monitoring study as planned.

Even in well designed studies, monitoring objectives will have to be formulated in a way that provides context for both statistical and biological significance. The variance associated with the measurement of these attributes could lead to a failure to find statistically significant differences (α greater than 0.05) between groups of streams (i.e., managed versus unmanaged), although the means of the two groups differ by 10 to 30 percent. Interpretation of such results cannot be based on statistical significance alone but must include an understanding of monitoring objectives, biological significance, and the cost of Type I and Type II errors (Peterman, 1990; Johnson, 1999; Anderson *et al.*, 2000).

Our results suggest that careful evaluation of variability associated with the measurement of stream

attributes can improve sampling designs so that many physical stream attributes can be used to monitor stream conditions. The high total variability associated with the measurement of these variables suggests that failure to incorporate a well designed sampling scheme, extensive training, and operational protocols will result in stream monitoring programs that fail to detect altered or changing stream conditions.

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