A Physical Habitat Index Focusing on Salmonids in the Pacific Northwest

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Biographical Sketches of Authors

Ed Chadd is the co-manager, along with Hannah Merrill, of Streamkeepers of Clallam County, a volunteer watershed monitoring and public involvement program of local government. Among other accomplishments, Streamkeepers has helped to create a unique set of volunteer-friendly physical-habitat monitoring protocols which complement biological and chemical monitoring (see http://www.clallam.net/streamkeepers). He helped to create the program in 1999, after serving as a volunteer in a predecessor program for the prior two years. He has a long history with volunteer and civic organizations (Peace Corps, Student Conservation Association, League of Women Voters), as well as with education (public schools, adult education, Outward Bound, community college). His current mission is as an applied scientist/educator, dedicated to bringing science to the people in a way that can enhance their lives and the planet.

Erin Clancy is a Valle Scholar working on her M.S.E. degree in Civil and Environmental Engineering at the University of Washington. In 2003, she received a B.S. with Distinction in Mathematical Biology from Harvey Mudd College, one of the Claremont Colleges of California. Her current research focus is on water resources and systems engineering, with a particular emphasis on mathematical modeling. She is currently working on the Tualatin Basin (OR) Climate Change Impacts Project, projecting impacts on hydrology, water supply, and stream temperature. Erin did the initial research and drafting of this paper, first as a volunteer and then as a contractor. She is a native of Port Angeles, Streamkeepers' home base.

Jon Mowe is the Environmental/Natural Resources Vocational Teacher at the North Olympic Peninsula Skills Center, a regional training consortium of local school districts, governments, businesses, and Peninsula College. He has created field-based Environmental Science programs both in his current job and a prior job with the Sequim (WA) School District, in both cases collaborating with Streamkeepers to provide a more meaningful experience for his students. He holds an M.Ed. from Eastern Washington University and a B.S. in Fisheries Biology/Management from the University of Washington. He has a long history as a field and laboratory biological technician going back to the 1970s. Jon worked on this paper as a contractor, helping to do further research, analysis, and drafting.

Abstract

Streamkeepers of Clallam County, a volunteer stream monitoring program of local government on the Olympic Peninsula in Washington State, is developing a multimetric Physical Habitat for Salmonids Index (PHI) based on physical-habitat parameters we have been monitoring at several dozen sites since 1999. Such an index would complement indices we already use to characterize biological integrity (B-IBI) and water quality (WQI). In our mostly-rural county undergoing rapid development, with salmon runs ranging from relatively healthy to extinct, we have an opportunity to track environmental impacts while ecosystems are relatively intact, and potentially to forestall further impacts through land-management decisions. This presentation will cover the rationale, development, and further applications of our PHI. We faced the challenge of developing our index with an already-existing data set, in a field where reference-standards are hard to come by. We will discuss some of the challenges we faced and approaches we took to overcome them. We consider our PHI a work-in-progress and have come to the National Water Quality Monitoring Council conference seeking feedback from the scientific community.

Purpose and Design of the Index

A variety of features can tell us important things about the health of a stream, if we know how to interpret them. Streamkeepers of Clallam County, a volunteer program collecting data on physical, chemical, and biological parameters of stream ecosystems on the northern half of the Olympic Peninsula of Washington State, already has access to two multimetric indices of stream health incorporating subsets of the data it collects:

- A Benthic Index of Biological Integrity (B-IBI) (Karr et al., 1986), which assesses integrity of stream biota, and
- A Water Quality Index (WQI) (Clallam County, 2004), which assesses the quality of the water itself.

This paper summarizes efforts we have made toward a third multimetric index: a Physical Habitat Index (PHI), which assesses physical aspects of the stream channel and riparian zone in regard to their ability to support the full life-cycle of Pacific Northwest salmonids, by comparing Streamkeepers' physical-habitat data to norms for a healthy stream. The purposes of the PHI will be to better understand the data; to track trends, correlations, and hot-spots; and to help managers and the general public protect and restore the streams.

If we were conducting scientific research, information-gathering would be driven by analytical specifications. However, because Streamkeepers has already gathered a large body of data from our local streams, the algorithms used for analysis must be chosen from those that can be implemented using the data that are available. Parameters (and the methods used to calculate them) that best describe the integrity of a stream with respect to its presumed ability to support healthy salmonid populations thus had to be selected from this predetermined list. However, the Streamkeepers physical-habitat monitoring program was designed—under considerable constraints with respect to expertise, time, and finances—in anticipation of its relevance to future evaluation of the health of the streams, based on work done at the University of Washington (Scholz and Booth, 1999 and 2000). Therefore, development of this index postliminarily did not significantly limit its power. In fact, it was not necessary to use the entire body of information contained in the Streamkeepers database. Where redundant elements existed, the index incorporates the attribute that is most likely to be accurately measured and for which the clearest decision criteria are available. MacDonald et al. (1991) was useful in identifying superfluous measurements. For example:

- Streamkeepers collects data on both bed material size and embeddedness, both of which relate to the physical process of bedload transport and sorting. Both characteristics also have equally strong effects on bedload, turbidity, and suspended solids. Therefore, we used only particle-size distribution in our metric.
- Pacific Northwest forests in their natural state are dominated by conifers, and Streamkeepers estimates the percentage of conifer vs. hardwood cover. However, we have chosen to use instead our measure of winter canopy closure for the PHI, since the winter canopy is heavily dependent on mature conifers, and canopy closure is a more quantitative and precise measure.
- Salmonids depend on cool temperatures, but Streamkeepers' temperature data is non-continuous and thus difficult to fit easily into a physical habitat metric. (Also, temperature is already incorporated into our WQI.) However, full canopy cover and an abundance of habitat units are assessed by the PHI and virtually assure the availability of an appropriate thermal environment for indigenous fish.

If data from outside sources becomes available to Streamkeepers in a format compatible with the PHI's structure, that data can be incorporated into calculation of future versions of the index. In its current form, however, the PHI can be evaluated using just the records contained in the Streamkeepers database.

It should be noted that all 3 indices mentioned above—B-IBI, WQI, and PHI—are associated with reach-specific data, and although their parameters would all be impacted by upstream and up-slope watershed phenomena, they indicate stream health only at the particular sampling reach. Temporally, the index integrates data from the preceding several years.

It should also be noted that most Streamkeepers monitoring reaches are only 100 feet long, which is in most cases considerably shorter than that recommended by scientists for a representative reach length. For example, the EPA recommends a minimum sample reach length of 150 m. or 40 times the baseflow wetted width, whichever is longer (Kaufmann et al., 1999). In addition, our reaches were picked not randomly, but in consideration of a number of factors, including access, interval length, adjacent land-use, and minimum distance from channel alterations. "Representativeness of the suite of sites for the stream as a whole and of the particular site for its location in the watershed" are factors considered when locating sites, but these are not determined in any statistical way, and further sampling would be needed to assess the representativeness of our sampling reaches.

Calculating the PHI

For each parameter chosen, we derived a formula yielding a value generally between 0 and 1, where 1 is to be interpreted as a reference condition for a high-quality site, and 0 is to be interpreted as a condition degraded to the point that further degradation would not result in a distinguishable decline in the site's ability to support salmonid life—a "bottomed-out" point. The formulas for each parameter are discussed in subsequent sections. Numbers less than 0 or greater than 1 are rounded off, so that parameters only take on values between 0 and 1.

Individual parameter scores for a given site are then grouped into subscores representing four habitat components: instream habitat, riparian habitat, sediment transport, and channel integrity. Each subscore is computed as the arithmetic mean of one or more constituent parameters; for example, the instream-habitat subscore averages values calculated for the LWD and Pools parameters (see Figure 1).

In the rare case where the data needed for a formula is not available for a given parameter at a reach, the score for the missing parameter is estimated using the best professional judgment of local biologists. (In many cases, published guidelines exist for making qualitative judgments regarding these physical habitat components—see, for example, McBride 2001). In such cases, an asterisk indicates a PHI score based on incomplete data.

Finally, the habitat-component subscores are averaged to arrive at the overall PHI score, which will be a number between 0 (highly impaired) and 1 (very healthy). Figure 1 on the following page outlines the PHI process.

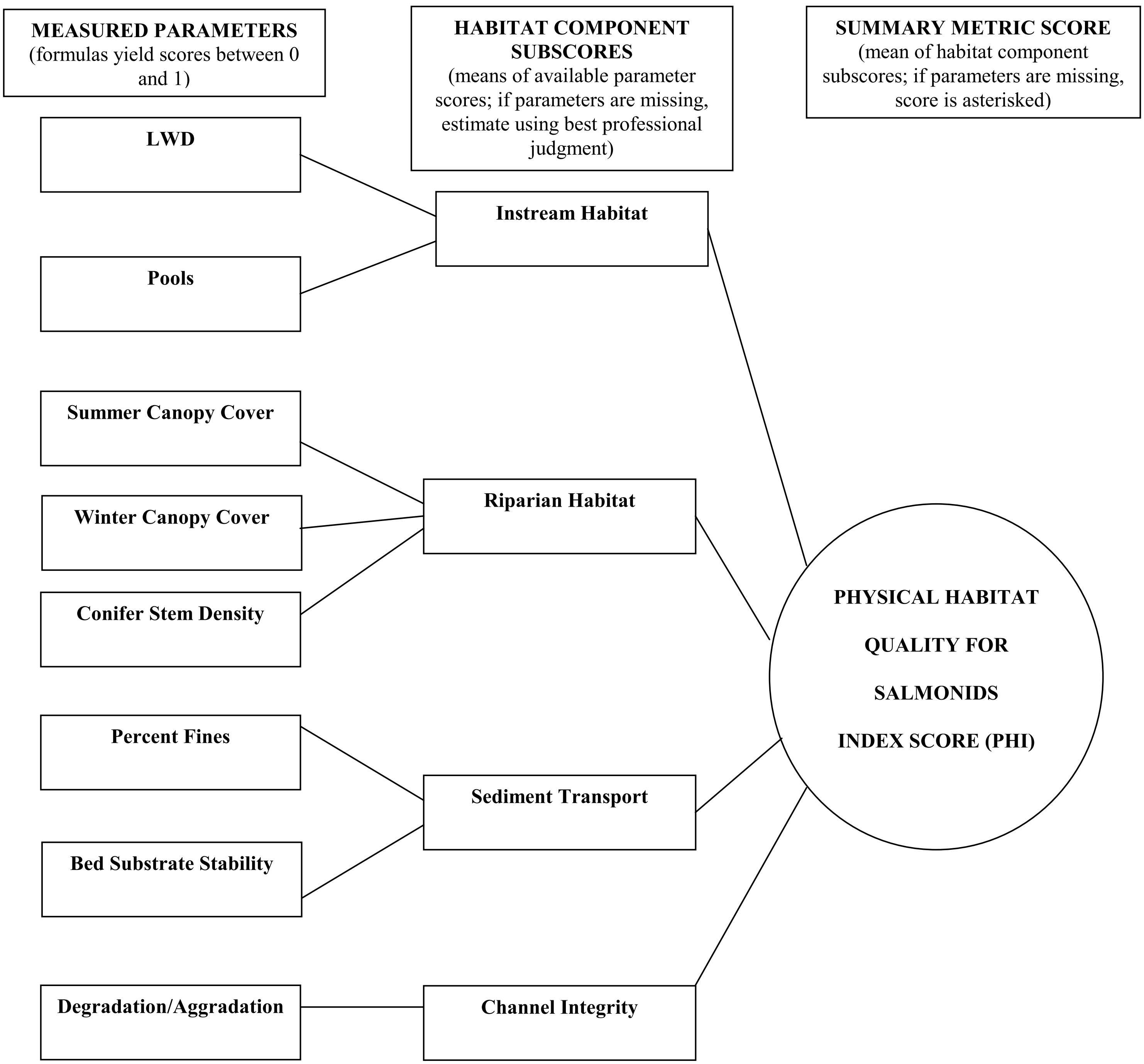

Figure 1. Physical Habitat Index for Salmonids (PHI) Model.

Theoretical Basis and Formulas for Each PHI Component

Instream Habitat Component

Conditions inside the channel are on the short list of characteristics that determine the overall condition of a reach (Reynolds and Peets, 2001). Since the use of streams as salmon habitat is the priority on which the PHI is based, in-channel conditions are evaluated with respect to their presumed ability to support salmonid spawning and development. One important in-channel salmonid habitat requirement is instream cover (Bjornn and Reiser, 1991). Cover has been used in other theoretical assessments of channel character (e.g. Karr et al., 1986) but is difficult to measure in itself; even trained crews estimate fish cover with moderately low precision (Kaufmann et al., 1999). Two important sources of cover that are easily identifiable units are large woody debris and pools. In addition, these features provide streams with nutrients, channel complexity, thermal refuge, sediment storage capacity, habitat stability, and flood refuge. By measuring the frequency of these two features per unit of stream length, we can quantify two crucial components of instream habitat.

Large Woody Debris

Large woody debris (LWD) contributes to the formation and preservation of fish microhabitat (e.g. Bryant, 1983; MacDonald et al., 1991; Hicks et al., 1991). In these microhabitats, salmonids find food, spawning gravel, and protection from predators and storm flows (Hicks et al., 1991). In addition to simply increasing the number of microhabitats, LWD also increases microhabitat complexity, which insures that cover and suitable habitat can be found over a wide range of flow and climatic conditions (MacDonald et al., 1991). By partitioning the available habitat and increasing the range of hydraulic gradients, LWD allows multiple species of salmon with varying habitat needs to coexist in the stream community (Hicks et al., 1991). For example, Bisson et al. (1987) determined that certain age groups and species of salmon prefer to take cover in pools formed by LWD. When Murphy et al. (1984) examined two Washington streams that were morphologically similar and had similar biomass density, they found that the debris-rich stream had much greater salmon diversity than the debris-poor stream as measured by the Shannon-Weaver index: H = 0.90 in debris-rich stream, H = 0.15 in debris-poor stream (Murphy et al., 1984).

Generally, fish-carrying capacity increases proportionately with available instream cover (Murphy and Meehan, 1991). In fact, salmonid abundance is tightly correlated with LWD abundance—especially in the winter (Hicks et al., 1991)—and no known data indicates an upper end to this relationship (Bisson et al., 1987). For some groups of salmonids, the benefit may be disproportionately large. A study conducted during the winter in southeastern Alaska, for example, found the relationship between LWD mass and juvenile coho salmon density to be exponential (Murphy et al., 1984).

During decades past, laws requiring channels to be cleared of LWD (Hicks et al., 1991) unintentionally created experiments that proved the causal relationship between LWD availability and larger salmon communities. Salmon abundance dropped when their habitat quality declined after LWD, or the trees that create it, was removed from streams (Hicks et al., 1991; Dolloff, 1986; Elliott, 1986). The dwindling frequency of LWD in the streams of western North America (Bisson et al., 1987) is probably a cause of vanishing salmon populations today.

To assess a stream's LWD status, it suffices simply to tally the number of pieces it contains (Scholz and Booth, 2000). Although some parameters measure LWD in terms of pieces / m^2 or m^3 / m^2, the EPA uses a parameter that measures the number of LWD pieces / 100 m (Kaufmann et al., 1999), and Scholz and Booth (2000) report that it is more important to just check that LWD exceeds a minimum size than it is to measure its exact dimensions. Streamkeepers, based on Scholz and Booth's suggestion, has set 10 in. diameter and 10 ft. length as the minimum dimensions for a piece of LWD (except for rootwads, which need to be 10 in. diameter at the base of the trunk with a 3 ft. diameter root system). We have also defined a logjam as an accumulation of 5 or more qualifying logs or rootwads in contact with one another or associated with the same structure; we do not count logjam pieces individually. The value of the LWD count for one year is given by summing over all types of LWD intruding into the wetted or bankfull channels: dead/dying logs, rootwads, and living logs (if fallen or tipping and within the bankfull channel) each count as 1 piece; a logjam counts as 5 pieces (a conservative estimate, as 5 is the minimum qualifying number). LWD counts for the metric are based on the most recent year in which an acceptable tally was performed.

Despite unanimous agreement over its importance, established numerical standards have been hard to come by (MacDonald et al., 1991). The Washington State Forest Practices Board (WFPB) has established an LWD target for a "good" habitat quality rating: 2 pieces (>2 m. length x 10 cm. diameter) per channel width (for channels <20 m. in width) (Washington Forest Practices Board, 1997). However, in a study of 150 stream segments draining unmanaged basins within Washington State, Fox (2001) found that this target does not match reference conditions found on streams <20 m. wide—it is too high for streams <3 m. wide, and too low for streams >12 m. wide—nor does it establish a standard for wider streams (Fox, 2001).

Fox found the "best-fit" metric for LWD statistics to be the number of pieces per length of stream (not per channel-widths of stream-length, as LWD stocking is sometimes reported), when grouped into three categories by bankfull-width size. He suggests that "good" LWD stocking be targeted as the 75th percentile of what he found at his reference sites in Western Washington, described in terms of LWD pieces per 100 m. and categorized into his three stream-width size classes. However, he uses the WFPB definition of a LWD piece, which uses a smaller minimum size than Streamkeepers. Fox estimates that the WFPB definition yields counts that are typically 20-50% higher than the Streamkeepers definition, with the discrepancy greater for wider streams, due to quantities of small wood often found within the large logjams often associated with larger streams (Fox, personal communication). In fact, Streamkeepers' undercounts would tend to be even more extreme than Fox suggests as streams widen, because logjams would tend to be larger, and our metric counts all logjams as just 5 pieces. Taking the above factors into account, we developed the following LWD rating table:

Table 1. Reference targets for LWD pieces per 100 ft. (based on Fox, 2001 and 2003)[1].

Bankfull-width class	Fox's target per 100 ft. (using WFPB's LWD def.)	Adjustment factor for definitional difference	Target per 100 ft. (using Streamkeepers' LWD def.)
0 - 19.7 ft.	11.6	0.8	9
>19.7 - 98.4 ft.	19.3	0.6	12
>98.4 -328.1 ft.	63.5	0.4	25

To derive the lower boundary for the LWD metric, we looked at two sources. Fox recommends using the 25th percentile counts from his reference sites, which using the above adjustment factors, yield a value of approximately 6 pieces per 100 ft. for all three width-classes. However, this standard seems high, as Fox's survey was limited to undisturbed sites. McBride (2001) derived an index based on sites on first- to third-order streams in the Puget Sound lowlands with a gradient of disturbance (but none undisturbed), using the same LWD definition as Streamkeepers. We have set our lower boundary at the upper point of her "poor" category, comprising sites with fewer than 5 pieces per 100 m. (or <1.5 pieces per 100 ft.).

Table 2 shows the derivation of equations for the LWD metric, based on straight lines drawn between the upper and lower boundaries described above, for the three bankfull-width classes described by Fox:

Table 2. Derivation of LWD metric formulas, by bankfull-width class, where *x*-values are LWD pieces per 100 ft. (Streamkeepers definition); *y*-values are Streamkeepers' PHI LWD metric scores.

Bankfull-width class:	Small 0 - 19.7 ft.	Medium >19.7 - 98.4 ft.	Large >98.4 -328.1 ft.
x-value for upper limit (y=1)	9	12	25
x-value for lower limit (y=0)	1.5	1.5	1.5
Slope of line	0.13	0.095	0.043
Intercept of line	-0.2	-0.14	-0.064

[1] It should be noted that Streamkeepers measures bankfull width at a single point in each monitoring reach: the cross-section transect, which is placed across a straight and even stretch of a riffle, run, or glide, ideally midway between bends. Bankfull height is determined by either the crest of the bank, or three-year-old woody vegetation growing in humus-content soil, or an extrapolated line taken from another spot where bankfull is clear. This bankfull width measurement is used in PHI metrics for LWD, Pools, Conifer Stocking (in most cases), Degradation/Aggradation, and Bed Substrate Stability.

These slopes and intercepts lead to the following three equations:

LWD score (small streams) = 0.13 * (# LWD pieces * 100 / reach length) - 0.2 **(1a)**
LWD score (medium streams) = 0.095 * (# LWD pieces * 100 / reach length) - 0.14 **(1b)**
LWD score (large streams) = 0.043 * (# LWD pieces * 100 / reach length) - 0.064 **(1c)**

(Note that in all of these metrics, scores are rounded to two significant figures, and numbers less than 0 or greater than 1 are rounded off, so that scores only take on values within the specified range.)

Pools

Pools provide important winter habitat for juveniles (Meehan, 1991), refuge from too-warm summertime water (Reeves et al., 1991), shelter from predators (Hicks et al., 1991), food, and space for rearing (Clark and Gibbons, 1991). They also are an indicator of overall channel complexity, are easier to measure consistently than other channel-habitat units, and show a consistent inverse relationship with human watershed disturbance (Scholz and Booth, 1999).

However, it should be noted that a study conducted in the Puget Sound lowlands by McBride (2001) found pools to show no consistent relationship with channel complexity or urbanization, causing her to reject pools as a metric in her Physical Stream Conditions Index (PSCI). Furthermore, Washington State's *Watershed Analysis Manual* (Washington Forest Practices Board, 1997), from whence we derive our standards, suggests that variations in channel type may make a single standard inappropriate. A study in the Puget Sound lowlands performed by Sossa (2003) found pool frequency to vary with channel type, with forced pool-riffle (FPR) channel-types having more pools than plane-bed (PB) channels; however, Sossa also found that watershed disturbances of various types tend to transform FPR into PB channels, and Sossa found pools to be a useful metric in her PSCI. So we proceed with caution.

Although pool frequency is sometimes monitored in terms of its ratio to riffle frequency (Murphy et al., 1984; Kaufmann et al., 1999), the optimal ratio is highly variable among streams and fish species (MacDonald et al., 1991), and data collected even by trained personnel has a low precision (Kaufmann et al., 1999). Therefore, it is preferable to score pools independently of their relation to riffle frequency.

The scoring criteria, shown in Table 3, are based on guidelines found in Washington State's *Watershed Analysis Manual* (Washington Forest Practices Board, 1997), and corroborated by Sossa's findings (2003). The parameter scored is pools per channel-width of reach length, where pools are defined by minimum residual depths according to channel width (Schuett-Hames et al., 1994) as shown in Table 4, and residual depth is calculated as the difference in measurements between a pool's maximum and outlet depths.

Table 3. Pool grades adapted from *Watershed Analysis Manual* (Washington Forest Practices Board, 1997).

Pool Frequency (Channel Widths per Pool)	Grade	Pool Frequency (Converted to Pools per Channel-Width of Reach Length)	Scoring for PHI Pools Metric
>4	Poor	<0.25	Set 0 at 0.25
2 - 4	Fair	0.25 – 0.50	
<2	Good	>0.50	Set 1 at 0.5

Table 4. Minimum residual pool depth for given bankfull width (adapted from Schuett-Hames et al., 1994)

Bankfull Width	Minimum Residual Pool Depth
0 – 8.2	0.3
8.2 – 16.4	0.7
16.4 – 32.8	0.8
32.8 – 49.2	1.0
49.2 – 65.6	1.2
>65.6	1.3

To compute a PHI component for pools frequency, we fitted a linear curve to the pool targets set in Table 3, yielding the following formula for our pools score:

Pools Score = 4 * Pools per Channel Width – 1 **(2)**

Where Pools per Channel Width = pools per reach * channel width/ reach length

Riparian Habitat Component

A stream's riparian canopy must be intact if it is to contain productive fish habitats within (Hicks et al., 1991). Fish need the food, cover, and shade that a riparian canopy provides (Clark and Gibbons, 1991). The trees that create the canopy also stabilize the banks and provide the channel with a source of woody debris (MacDonald et al., 1991). Management of the riparian zone is thus a necessary component of managing fish habitat (Reeves et al., 1991).

The riparian habitat component of our PHI functions to determine whether the riparian zone is adequately forested, whether conifers dominate the canopy (preferred), and what the future prospects are for mature conifers in the riparian zone. Although most standards for riparian composition are expressed in terms of conifer-to-deciduous ratio, estimates of this ratio are not very reliable, so we look at riparian composition using three different parameters that offer a fair amount of precision:

- Summer canopy cover, which measures the total abundance of mature canopy;
- Winter canopy cover, which serves as a proxy measure of conifer canopy; and
- Conifer stem density in the riparian zone, which measures the future potential for mature conifers.

Thus, a site with full, mature conifer stocking would score high on all three metrics; a site dominated by hardwoods but with an abundance of conifer seedlings would score well on two of them; and a site tangled in weedy underbrush would score poorly on all three.

Canopy Cover (Summer & Winter)

Corridor vegetation is an important component of stream health, and cover percentage is the most effective way to monitor it (Scholz and Booth, 2000). Several studies have demonstrated that stream health depends on an intact canopy (e.g. Steedman, 1988; May, 1996; Horner et al., 1997).

The obvious result of a fragmentary canopy is increased incident solar radiation, and an accompanying increase in water temperature, light levels, and autotrophic production. Although enhanced bioproductivity implies more food for the fish, salmonid production is not necessarily greater when nutrient levels are elevated (Gregory et al., 1987). Salmonid growth rates have evolved toward maximum efficiency, and the accelerated growth that occurs when their food source is enlarged changes the age at which they become smolts and makes them more susceptible to disease (Hicks et al., 1991).

In addition to its direct physiological effects, a patchy canopy compromises the quality of salmonids' habitat because it is inadequate as a source of LWD and as a buffer against increased discharge after precipitation (Hicks et al., 1991). Although more flow is often desirable during the summer, it comes at the cost of elevated flood risk during the rest of the year.

Streamkeepers measures percent canopy cover with a spherical densiometer mirror, held in four directions at the mid-channel point of the reach's cross-section transect, in which closed-canopy areas are counted in a 96-point grid. Reference conditions for canopy measured in this manner can be found in Appendix 1 of the Water Quality module in the Washington State *Watershed Analysis Manual* (Washington Forest Practices Board, 1997), which reports on a study done in mature coniferous forests using spherical densiometers to compare bankfull width to canopy. The study showed that potential canopy is reduced as a stream widens, due to the open area of the channel, in a relatively linear relationship for channels <130 ft. wide (which would include all of Streamkeepers' regular monitoring sites). That relationship is described by the equation:

Canopy closure % potential = 100 - (bankfull width * 42.5 / 125) **(3)**

This relationship yields the equations:

Summer/Winter Canopy Cover Scores = % canopy cover / canopy closure % potential **(4)**

Conifer Stem Density

Riparian forest composition is salient to the interaction between the channel and the riparian zone. Woody debris from coniferous trees is larger and more resistant to biodegradation than woody debris from deciduous trees (MacDonald et al., 1991). Larger branches and root wads are more stable, contribute to greater habitat diversity, and redirect flow in small streams. While still living, the roots of large conifers stabilize the bank and protect it during high-flow occurrences (MacDonald et al., 1991).

Deciduous trees are less desirable. In our area, conifers have been removed from many riparian zones, with alders typically succeeding them. Although alders fix nitrogen, and are the natural and most common first-stage successional deciduous species in both naturally- and artificially-disturbed environments, they do not compare to conifers in terms of providing winter cover, bank stability or long-lasting, channel-forming LWD. Many alder stands in our vicinity are at the point in their life history where their death can be expected within 20 years, which will produce gaps in the canopy. In virtually all cases, it is desirable that conifers succeed the alder. In a hardwood-dominated riparian zone, the conifer stem density metric measures the potential for this succession.

The *conifer stem density score* measures the stocking-level of conifers by counting stems of all sizes in the reach. Since 60 stems is considered a full stocking of conifers for a 10,000 sq. ft. area (Freudenthal, personal communication), the score at a reach with *n* conifers is:

Conifer Stem Density score = (n / 60) * (10,000 sq. ft. / conifer stem count area) **(5)**

Sediment Transport Component

Researchers agree that the range of sediment sizes in a stream is an important component of its health, but there is much disagreement over how to assess it (Bauer and Ralph, 1999). Assessment is complicated by the facts that sediment size varies with underlying geology (Bauer and Ralph, 1999), can vary greatly along a stream, both temporally and spatially, and often varies more directly with nearby sediment sources than with large-scale factors related to watershed disturbance (Doyle and Shields, 2000). The temporal variation is especially problematic in that Streamkeepers measures sediment sizes during summer low-flows, the time of year when salmonids are least dependent on sediment quality (Kondolf, 2000).

Assessment is further complicated by the lack of a simple means by which to assess sediment in the smaller size ranges. The only simple way to assess sediment size in the field is by a pebble count (Wolman, 1954; Kondolf, 1997), and Streamkeepers uses this method. However, it is not possible to accurately assess the proportion of fines from a pebble count (Bunte and Abt, 2001; Fripp and Diplas, 1993), because the pebble-count method is based on the assumption of random particle selection, which becomes more tenuous as the particles become smaller. When one is selecting a fine grain, even a line drawn on one's fingernail is too broad to indicate one particle obviously and uniquely (Bunte and Abt, 2001). This problem introduces an element of subjectivity into the sample selection process at the fine tail of the sediment-size distribution, rendering the division of small particles into multiple size classes meaningless (Booth, personal communication).

To score sediment-transport processes for the streams, we looked for parameters that would be fairly robust, given the various measurement difficulties. We chose one parameter based on the proportion of fine sediment in the reach, and another that compares sediment size to the stream's ability to move sediment at high flows. Both of these are devised in ways that do not depend on precise measuring technique and that are not greatly influenced by the individual variation that exists among streams.

Percent Fines

Percentage of fine particles is the basis on which sediment quality is traditionally assessed. In order to use percent fines as the basis of a score, one must first define the category "fines." In the 1990s and earlier, fines were often limited to particles <0.85 mm in diameter (e.g. Washington Forest Practices Board, 1997), although cutoff values in the literature range between 0.75 and 9.5 mm (McCullough and Espinosa, 1996; Tappel and Bjornn, 1983). This assortment of definitions complicates comparison among studies and monitoring programs of the relationship between percent fines and salmonid population health.

It is clear, however, that substrate size is crucial to stream biota (Scholz and Booth, 2000). There are five ways that infiltration of fine sediment into spawning gravel can harm fish. One, the deposits compress eggs and alevins and seal in fry, thereby reducing fish productivity (Hicks et al., 1991; McCullough and Espinosa, 1996). Two, the deposits reduce interstitial flow velocity, which limits alevin activity and causes fry to be smaller (McCullough and Espinosa, 1996; Bjornn and Reiser, 1991). Three, they lower the dissolved oxygen content (DO) in the space between gravel particles, which damages developing fry (McCullough and Espinosa, 1996; Chapman, 1988). Four, they reduce salmonids' feeding by reducing their efficiency and decreasing the stream ecosystem's primary productivity (Hicks et al., 1991; Chapman, 1988). Five, they block intergravel spaces and prevent their use as a winter refuge (Hicks et al., 1991). Elevated fines percentages have been experimentally shown to reduce both embryo survival and fry emergence (Bjornn and Reiser, 1991). In fact, influx of fine sediment via hillslope erosion is the mechanism identified by which logging activity has resulted in lowered emergence of salmon fry on the Olympic Peninsula, as well as in other areas (Cederholm and Salo, 1979; Cederholm et al., 1981; Cederholm et al., 1982). However, it should be noted that fine sediment occurs naturally, natural levels vary, and salmonids have shown the ability to adapt to varying levels of it (McHenry et al., 1994).

The above discussion makes clear that fines are most critically a problem in the subsurface layer. However, in this regard we face yet another measurement problem in addition to the ones described in the prior section: to measure this subsurface layer requires a core sample; pebble counts only sample the sediment particles on the channel-bed surface, and thus miss this subsurface layer (Bunte and Abt, 2001).

Nevertheless, sediment size is an important component of stream health, and pebble counts are the best simple method available to assess that component (Scholz and Booth, 2000). We knew from digging in the subsurface layer for our macroinvertebrate samplings that our streams tend to be armored at the surface layer, with smaller sediment underneath; in such a case, elevated fines in a surface-layer pebble count are a sign of even worse problems beneath. Looking at our data corroborated this hunch, because it did help to distinguish high-quality from low-quality sites, if only at a crude level. Therefore, we decided to include a fines-based metric in our index. However, since a fines-based metric relates to salmonid spawning areas, we limited application of this metric to sampling sites with gradients of 0.5 – 4%. Sites outside this range did not receive a fines score.

We based our index on the only norms available: those established for levels of fines in the subsurface layer. Kondolf (2000) gives a good overview of salmonid-emergence studies, two performed in the field and eleven in the lab. He presents data from these studies relating percent fines of various sizes to 50% incubation/emergence success, a success rate he considers to be a good rough cutoff point between healthy and impaired conditions. For the categories of percent fines of both <3.35 and <6.35 mm, values of about 30% fines were related with the 50% emergence threshold. However, 50% emergence might be setting the threshold low, as emergence values can be found >80% (Tappel and Bjornn, 1983); other investigators have set the fines threshold as low as 10% for the beginning of impairment for fines <6.3 mm (Mobrand Biometrics, 2003); and our pebble-count method is likely to undercount fines due to factors listed above. Therefore, we decided upon 15% fines <4 mm (the smallest increment Streamkeepers measures) as our threshold for the beginning of impairment. For the upper limit of impairment, we used 70%, the percentage of fines <3.3 mm associated in a study with survival to emergence of less than 10% (Phillips et al., 1975).

Because we have found percent fines to vary temporally both in the literature and in our own data from year to year, we wanted our metric to consider the entire data set going back to 1999, and when we had more than one count available, we wanted the metric to consider the worst of the counts, as well as the central tendency. So we decided to use (MAX + MEAN)/2, where MAX is the highest of the <4 mm percentages, and MEAN is the arithmetic mean of those percentages, over all the years in which there were counts since 1999. When there is just one value, both MAX and MEAN are that value.

Using the targets and definitions described above, we derive our fine-sediment metric:

$$\text{Fines Score} = (140 - \text{MAX} - \text{MEAN}) / 110 \qquad \textbf{(6)}$$

Bed Substrate Stability (Relative Bed Stability)

Another way to evaluate sediment transport in a stream relative to reference conditions is to determine how much the substrate size deviates from what would be expected without human influence. If the substrate is fine relative to the stream's transport capacity, the channel is excessively unstable, and the fines may cause problems for salmonids. If the substrate is coarse relative to transport capacity, the channel is excessively stable, and therefore will tend to lose habitat complexity (Kaufmann et al., 1999). One measure of bed substrate stability, Relative Bed Stability (RBS), measures the relationship between the average particle size in a streambed and the largest-sized particle that the stream should be able to move at bankfull flows, based on the topography of the channel. All equations, estimations, and procedures are taken from Kaufmann et al. (1999), who use RBS to rate channel condition for the EPA. RBS reflects a traditional engineering approach, involving informed estimations and empirical knowledge, to place the problem of substrate fining in the context of the physical processes in which it occurs.

The first step in calculating RBS is to determine D_{50} , the median particle diameter, based on a 100-particle pebble count. Streamkeepers' pebble counts are in "half-phi" size classes whose boundaries are defined as $(\sqrt{2})^x$ mm, starting with a class whose upper limit is $x = 4$ (i.e., $(\sqrt{2})^4$ or 4 mm), and continuing with classes whose lower limits begin with $x = 4$ and end with $x = 18$ (i.e., $(\sqrt{2})^{18}$ or 512 mm). For a count of n pebbles, we estimate the median by first finding the class into which the n/2-biggest particle would lie, defined as its "half-phi" lower-limit exponent x_L (in the case of the smallest class, x_L is assumed to be 3), then interpolate geometrically within that class by interpolating exponents:

$$D_{50} = (\sqrt{2})^{(x_L + (n/2 - CF)/f)} \qquad \textbf{(7)}$$

where: CF = the cumulative number of particles in classes smaller than the x_L class
f = the number of particles in the x_L class

(If the n/2-biggest particle fell into the smallest class, x_L would be assumed to be 3; and if the n/2-biggest particle fell into the largest class, that class would be assumed to have an upper limit of $x = 19$. These assumptions truncate both ends of the spectrum, but in practice, we do not have stream reaches in which the D_{50} falls into either of these classes.)

D_{50} will be compared to the maximum diameter of particles that can be transported at bankfull flow (D_{max}), so the next step is to calculate D_{max}. Since the size of these particles would be very difficult to actually measure, we must rely on an estimate of the bankfull channel bed shear stress that is derived from rules of fluid mechanics, and then choose the largest particle size that can be moved by shear stress of that magnitude. D_{max} can be estimated as:

$$D_{max} = 13.7 * \text{Gradient} * R^*_{bf} \quad \text{(Kaufmann et al., 1999)} \qquad \textbf{(8)}$$

where: D_{max} = Maximum diameter of particles that can be transported during bankfull flow
Gradient = Channel water surface slope (Dimensionless: rise/run)
R^*_{bf} = Effective bankfull hydraulic radius (mm) (see discussion below)

Kaufmann et al. average gradient, cross-section, and pebble counts over a reach scale (150 m. minimum), whereas Streamkeepers measures gradient over a single 50-100' line between similar habitat units (e.g., the heads of riffles), in the same area where both a cross-section and pebble-count are taken. For purposes of this metric, we assume that the formulas will still work reasonably well within the smaller spatial scale of our measurements.

To calculate R^*_{bf}, Kaufman et al. begin by estimating R_{bf}, the "raw" bankfull hydraulic radius, as the mean bankfull channel depth, which we compute as:

$$R_{bf} = \text{bankfull cross-sectional area / bankfull width} \qquad \textbf{(9)}$$

Actually, R_{bf} should be calculated as (bankfull area / bankfull wetted perimeter), but Kaufmann et al. substitute bankfull width for wetted perimeter on the assumption that the two are approximately equal for relatively wide channels. This assumption is less true for relatively deeper channels, but Kaufman et al. have found the error to be acceptable (Faustini, personal communication), and we follow their practice.

The effective shear stress on particles of the bed is typically reduced from this potential value by sources of roughness in the streams, such as large woody debris and channel-shape irregularities, which shorten the effective hydraulic radius, R^*_{bf}. To factor in these irregularities, Kaufmann et al. subtract factors for both wood and channel irregularities.

For wood, they calculate R_w, a factor for large woody debris "mean depth," as (LWD volume / channel area) (in mm. units). By making some assumptions, we can estimate R_w using Streamkeepers' data. Our minimum size for a log is 10' long by 10" diameter at the widest point. If we assume the geometric mean radius of such a log along its entire length to be 4", we can convert to mm. and calculate a minimum LWD volume per piece of $\pi r^2 l = \pi (101.6 \text{ mm.})^2 (3048 \text{ mm.}) = 9.88 * 10^7 \text{ mm}^3$. We can then multiply this volume by the number of pieces calculated in our LWD metric and divide by the channel area:

$$R_w = (9.88 * 10^7 \text{ mm}^3 * \text{\# of LWD pieces}) / [\text{bankfull width (mm.)} * \text{channel length (mm.)}] \quad \mathbf{(10)}$$

There are numerous sources of error in this equation: many LWD pieces will be significantly larger than this minimum size; some rootwads may be smaller; logjams are estimated at their minimum definition of 5 pieces; and many pieces may have a significant portion lying outside the bankfull zone. However, these errors may tend to cancel each other out, and we decided that since there were significant wood-loads on some of our streams, we would be more accurate including rather than excluding R_w, in spite of its crude calculation.

To factor in channel irregularities, Kaufmann et al. estimate R_p, the amount of residual-pool area in the thalweg profile. We cannot make such a calculation with Streamkeepers' data. However, their data for 102 randomly-sampled stream sites in the Oregon/Washington Coast Range shows that R_p / R_{bf} ranges up to 0.33 at the 75th percentile, for channels with ample pool complexity. To estimate R_p from Streamkeepers' data, we index our Pools metric to this 75th-percentile score, whereby for a Streamkeepers Pools score of 1, $R_p / R_{bf} = 0.33$, and for a Streamkeepers Pools score of 0, $R_p / R_{bf} = 0$. Again, this is a crude estimation, but in our judgment it generates less error than making no adjustment at all for R_p. This yields a formula for R_p:

$$R_p = 0.33 * R_{bf} * \text{Streamkeepers' PHI Pools score} \quad \mathbf{(11)}$$

Using these adjustment factors for wood and channel irregularity, we estimate effective hydraulic radius, R^*_{bf}, as follows:

$$R^*_{bf} = R_{bf} - R_w - R_p \quad \mathbf{(12)}$$

Kaufmann et al. set a limit to R_w at $0.9 * R_{bf}$, on the assumption that water flowing downhill must exert some shear-stress, even with extremely large volumes of wood (Peck, personal communication). We have generalized this practice to set the maximum value for $(R_w + R_p)$ at $0.9 * R_{bf}$.

Now RBS can be calculated as:

$$RBS = D_{50} / D_{max} \quad \mathbf{(13)}$$

For convenience and to normalize their variances, RBS values are usually expressed logarithmically:

$$LRBS = \log_{10} RBS \quad \mathbf{(14)}$$

The EPA uses the LRBS as a tool for evaluating the condition of or trends in stream physical habitat quality at the regional scale, and does not use it as a tool for making site-specific assessments, however, in theory it could be used for the latter purpose if the data is adequately detailed and reach-specific (Kaufmann, personal communication). We have decided to include an LRBS score in our PHI index because:

- Our sediment, cross-section, wood, and pools data are all gathered in the same reach, and therefore are fairly detailed and reach-specific.
- The metric is based on the logarithm of the calculated expression, thus reducing the impact of measurement and sampling error. For example, an error in the RBS score by a factor of 2 would only change the LRBS by +/- 0.3.
- Our purpose in developing our PHI index is not for legal purposes but rather to better understand the data and manage streams; we believe this index may help our understanding, in conjunction with the other metrics that constitute this index.

EPA has field-tested the LRBS index and compared it to watershed ecological integrity across the United States. We have based our reference standards on their regional findings for the Oregon/Washington Coast Range (Faustini, personal communication):

Table 5. Stream condition LRBS thresholds.

Condition	LRBS range	Function values for Bed Stability metric (x = LRBS; y = PHI Bed Stability score)
Good	> -1.0 to 0.5	(-1.0, 1) and (0.5, 1)
Highly Impaired	< -2.0 or > 1.5	(-2.0, 0) and (1.5, 0)

The upper "Highly Impaired" figure is extrapolated because EPA's data-set didn't include many sites that were impaired in that direction. Ultimately, Streamkeepers will have to develop and calibrate our own reference values, because these figures are intended for regional interpretation and based on less site-specific data; however, we choose these values as a good place to start.

Using linear curves between the function values in the above table, we calculate the bed stability score as follows:

For $LRBS < 0$, $bed\ stability\ score = 2 + LRBS$ **(15a)**

For $LRBS \geq 0$, $bed\ stability\ score = 1.5 - LRBS$ **(15b)**

Although it is undesirable for a channel to have an LRBS that is either particularly low or particularly high, there are different causes for each case. If LRBS is < -2.0, then RBS is < 0.01, and D_{50} is 100 times smaller than D_{max}, which means the sediment has undergone textural fining, because sediment supply exceeds transport capacity (Faustini, personal communication). Such a situation, often the result of intense land use, can result in large debris flows during high floods, and the fine particles can be deposited on salmon spawning grounds. High LRBS values, where $D_{50} >> D_{max}$, can be caused by channelization, bank armoring, or increased stormwater runoff. We consider deviation in either direction undesirable.

Channel Integrity Component

To learn about the stability of a stream's channel and bank, we can look at how its cross-sectional profile changes over time (Beschta and Platts, 1986). A channel's cross-section is defined as "a topographic profile of the stream banks and streambed along a transect perpendicular to the direction of flow" (MacDonald et al., 1991). Streamkeepers measures cross-section between permanent monuments, in the middle of straight, channel-spanning riffles, at the same locations each summer.

The rubric we developed to assess channel integrity is based on the idea of an optimal channel as one that is stable and has a discharge that balances its sediment load. Instability arises in the form of three sediment-related processes: erosion, degradation, and aggradation. Evidence of erosion would provide a way to monitor instability directly, but we decided against including erosion directly as a subcategory of channel integrity because the reaches are probably too short to provide an accurate measure of the frequency of eroded areas along the bank, and because severity of erosion is not easily quantified. However, when other data are lacking, evidence of erosion or revetment can be helpful in estimating the channel integrity score for a particular reach. In place of monitoring erosion directly, the PHI looks at changes in cross-sectional area to assess channel integrity.

Aggradation/Degradation

Unstable channel profiles can be the cause of ecological degradation; both aggradation and degradation can be causes for concern. Impacts such as urbanization, deforestation, and channelization can dramatically change the water- and sediment-transport patterns in a watershed, causing destructive changes in channel morphology. Channels may widen, and thus be subject to greater sunlight penetration and more extreme temperatures (MacDonald et al., 1991); or they may deepen and become disconnected from their floodplains (Sossa, 2003); or they may fill in with sediment from farther upstream. In all such cases, degradation of biological systems follows (Booth and Henshaw, 2001).

Since there are no established standards for evaluating instability of channel cross-sections (MacDonald et al., 1991), the channels monitored by Streamkeepers are scored relative to each other. The variable scored is the fractional change in bankfull cross-sectional area, and the scores are based on the 25th and 75th percentiles of these values. Some investigators have assessed channel instability by measuring the change in mean bankfull channel depth, but we prefer to measure the change in bankfull cross-sectional area, because a stream that widens but does not deepen is nevertheless experiencing channel instability.

Before proceeding, we should acknowledge the limitations of such an analytical approach. Booth and Henshaw (2001) found that a number of factors confound the correlation between watershed disturbance, channel instability, and measured cross-sectional change:

- Exact location of the measurement site within the channel network can be a determining factor in the degree of observed channel change. Some reaches respond readily to watershed disturbance, and others simply pass heightened levels of water and sediment downstream without changing themselves.
- The reach may not be well-placed to trace the impact of the particular watershed disturbance. For instance, a reach just downstream of a culvert or drainage ditch may show exaggerated impacts.
- The reach may have already gone through its degradation process by the time the monitoring starts, particularly if most of the development in the watershed occurred some years before. Henshaw (1999) has found that most Puget Sound lowlands streams are likely to restabilize naturally within 10 to 20 years of infilled development.

Having stated these qualifications, we will cautiously proceed. The limitations on data interpretation will tend to produce Type II rather than Type I errors; i.e., it may not find all streams that have disturbed water- and sediment-inputs, but the disturbances it finds will be real.

Streamkeepers data is available for 1999-2003, but cross-sectional profile was not measured at every reach every year. Comparisons between the change at one reach in 2000 and the change at another reach in 2001 are not valid, because high-flow may have been much greater in one year, which might make it seem that the reach measured in that year is less stable than the reach that was measured in a year when flows were moderate. Therefore, comparisons must only be made between changes that occurred over the same span of time. (Here we assume that a bad storm-year on one Clallam County stream or reach was also a bad storm-year on any other Clallam County stream or reach.) Since we want to use as much data as we have available, we determine the number of pairs of cross-section measurements we have at our various reaches in ten year-pair classes, each defined by the initial and subsequent year of the comparison:

1. 1999-2003	2. 2000-2003	3. 2001-2003	4. 2002--2003	5. 1999-2002
6. 2000-2002	7. 2001-2002	8. 1999-2001	9. 2000-2001	10. 1999-2000

A given reach might have all ten pairs of data; to qualify for this metric, a reach has to have at least one pair. Then we filter out year-pair classes by discarding those classes with fewer than 7 pairs, which will not meet our statistical needs.

We also factored in the inherent measurement error of our method, which we estimate at 0.1 ft., the increment of our channel-depth measurements. We therefore label as "no change" any change in bankfull-width area that computes (when divided by bankfull width) to a change in average bankfull-channel depth of < 0.1 ft.

This measurement-error cutoff corresponds well with those established by other investigators using similar methods (Booth and Henshaw, 2001; King County Department of Natural Resources, n.d.).

We calculate the fractional change in bankfull cross-sectional area in a given reach across a given year-pair as follows: If A_i equals the area of the cross-section in the initial year and A_s equals the area of the cross-section in the subsequent year, $|\mathbf{\Delta}_{i\text{-}s}|$ represents the absolute value of the difference in cross-sectional areas between the two years, and $|\mathbf{\Delta}_{i\text{-}s}| / A_i$ represents the fractional change from year i to year s. We calculate this fraction for every reach in that year-pair i-s, then find the 25th and 75th percentile of the fraction for that year-pair, using a standard quartile function. Letting $Q1_{i\text{-}s}$ equal the 25th percentile of fractional change in cross-sectional area in the year-pair i-s and $Q3_{i\text{-}s}$ equal the 75th percentile, the score for a reach that changes $|\mathbf{\Delta}_{i\text{-}s}| / A_i$ between year i and year s is:

$$AggDegScore_{i\text{-}s} = [Q3_{i\text{-}s} - (|\mathbf{\Delta}_{i\text{-}s}| / A_i)] / (Q3_{i\text{-}s} - Q1_{i\text{-}s}) \qquad \textbf{(16)}$$

As with other PHI metrics, scores < 0 or > 1 are rounded off. To derive the AggDeg score for a given reach, we simply average the individual AggDeg scores for that reach for the various year-pair classes into which it falls. This gives us the best measure of the stability of the channel cross-section over the entire period of study (since 1999).

AggDegScore for a reach = sum of its year-pair AggDegScores / # of its year-pairs **(17)**

Interpreting the PHI

Once we have obtained the overall PHI scores at our reaches, we will meet with our technical advisory committee to begin to interpret and analyze the scores. Roughly speaking, we will follow this strategy:

- Start with the upper and lower half of the scores. Find where it makes sense to make a rough division between good and bad based on the technical review committee's knowledge of the reaches. Do the scores make sense at this level?
- Then look at the upper category. Are there natural divisions within it, for instance between "excellent" and "good"? Are there scores for reaches that we wouldn't expect? If so, examine what factors gave that reach that score. Are there weighting factors to be considered, or other logical operators such as automatic downgrades for a certain type of dataset? Do the evaluators' intuitions neglect certain considerations that the PHI scores reflect? Repeat this analysis with the lower category.
- As this process continues, determine a working division of the scores into categories that seem to fit reality. Further sampling and correlations with other data will help to refine the index over time.

Further Applications and Extensions

As we have previously stated, our PHI may be a crude discriminator, but we hope it will help us understand both our data and the ecological interactions at work in our local streams, the better to manage them. Some further applications we foresee include the following:

- Adding additional parameters to the index, such as invasive exotic plants and channel size per drainage area.
- Devising a study to test and calibrate the index, perhaps by comparing it with other indexes.
- Correlating the index with other types of data, such as biological integrity, flow regime, and land disturbance.
- Using the index as an educational tool to help deepen people's understanding of watershed processes and interactions.

Whatever its shortcomings, our PHI will have been a success if it gets people to look more carefully at the data and apply it to what they see when they walk out their doors.

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