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[Article]

Evaluating the Status and Trends of Physical Stream Habitat in Headwater Streams within the Interior Columbia River and Upper Missouri River Basins Using an Index Approach

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Abstract.—Identifying the overall status of freshwater streams is an important step in evaluating effects of land management and prioritizing restoration activities. To address these needs, we developed an index of physical habitat condition for headwater streams based on physical stream habitat data (2003–2007) and evaluated the condition status of 217 reference streams and 934 managed streams in the interior Columbia River and upper Missouri River basins. We used data collected from reference reaches to generate this index, which consisted of eight commonly collected metrics used in stream habitat monitoring. We incorporated landscape and climatic covariates into multiple linear regression analyses to control for inherent differences in physical habitat attributes among reaches, and we scored the overall condition of reaches with index values ranging from 0 to 100. Our results indicated that the condition index score of physical habitat was significantly higher in reference reaches (mean ± SE = 47.1 ± 1.4) than in managed reaches (30.4 ± 0.7); relative to reference reaches, a greater frequency of managed reaches had low habitat condition and a lower frequency of managed reaches had high habitat condition. Analyses evaluating the relationship between management activities and the condition of physical habitat in streams indicated a significant negative relationship with lower index scores in stream reaches within catchments containing higher densities of roads. When roads and livestock grazing occurred within catchments, we found the presence of grazing had an additional, significant negative effect on the relationship between road density and the condition of physical habitat of streams. Our results suggested that once natural variability and geoclimatic differences among reaches are accounted for, a multimetric index approach can provide managers with an easily interpretable tool to monitor the status of the overall condition of physical habitat.

Stream habitat alteration, degradation, and loss related to forest and rangeland management have been identified as causes for the decline in the distribution and abundance of native salmonid fishes in the Pacific Northwest (Meehan 1991). The listing of resident and anadromous salmonids under the U.S. Endangered Species Act in the 1990s led to substantial changes in federal, state, and tribal land management activities (FEMAT 1993) as well as increased stream restoration efforts (Bernhardt et al. 2005; Katz et al. 2007). Determining whether changes in management or restoration efforts improve physical stream habitat, however, relies on the ability to consistently assess the status and trends of physical habitat.

Efforts to understand how land management activities have affected the quality of stream habitat have largely focused on the status of individual stream attributes (Dose and Roper 1994; McIntosh et al. 2000) or separate assessments of multiple attributes (Woodsmith and Buffington 1996; Kershner et al. 2004b). Concomitantly, many land management and restoration efforts historically focused on specific stream attributes without consideration of the overall condition of the stream or the watershed (Roper et al. 1997). While these assessments have improved our understanding of how management affects individual stream attributes, overall assessments that incorporate multiple attributes into a single measure of the status of stream habitat are still needed (Rheinhardt et al. 2007).

Multimetric indices of biotic integrity (IBIs) have been widely used in fisheries research and management to assess and define the health of stream systems (Karr and Chu 1999; Pont et al. 2006; Tejerina-Garro et al. 2006; Whittier et al. 2007; Roset et al. 2007; Stoddard et al. 2008). Ease of interpretation and effectiveness for monitoring the biotic condition of streams have led to the extensive use of IBIs at the local scale (Angermeier and Schlosser 1987; Lyons et al. 1996) and larger spatial scales (Oberdorff et al. 2002; Whittier et al. 2007). In the Pacific Northwest, IBIs developed for both coldwater fish (Mebane et al. 2003; Hughes et al. 2004) and macroinvertebrate (Herlihy et al. 2005) assemblages have been used to evaluate the effects of land management practices.

Despite the importance of physical conditions for instream biological assemblages (Binns and Eiserman 1979; Minshall et al. 1983), there have been no

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attempts to define a simple multimetric index of stream condition based on physical habitat conditions in the Pacific Northwest. Understanding the condition of physical habitat is particularly relevant as our assessments of the biota in a system may be directly or indirectly affected by factors other than those that appear obvious in terms of local physical structure (e.g., barriers to migration; Budy et al. 2002), the effects of known or unknown pathogens (e.g., whirling disease; Nehring and Walker 1996), water quality issues (Dunham et al. 2003), the direct and indirect effects of nonnative species (Rieman et al. 2006), and reduced habitat quality. As such, robust assessment of the condition of physical habitat can provide insight into which factors limit the biota of interest in a given system.

Biologists have historically evaluated the condition of stream habitat indirectly through measures of fish abundance. These measures are often accompanied by the identification of limiting factors (Reeves et al. 1989) and the scoring of habitat attributes based on use by fish species (Milner et al. 1998). As with other approaches, these methods focused primarily on individual stream attributes that are important for fish rather than focusing on the overall habitat integrity. Furthermore, these approaches can differ in their definition of condition depending on the species of interest, and they are difficult to apply for species that naturally occur at low densities (e.g., Al-Chokhachy and Budy 2007).

Earlier methodologies to evaluate overall stream condition have largely occurred through qualitative habitat assessments (Plafkin et al. 1989; Rankin 1989; Raven et al. 2002). More recently, there have been efforts to develop predictive models to generate expectations of instream habitat based on landscape data and compare these with observed values to assess stream condition (Davies et al. 2000; Parsons et al. 2004). A major tenet of these habitat assessments has been the use of reference conditions (Reynoldson et al. 1997) as the basis to evaluate stream conditions in watersheds experiencing land use disturbances (Kershner et al. 2004b; Parsons et al. 2004). In these cases, condition of a specific stream attribute is not defined by a single state but by the distributions of habitat values in stream reaches that are found in minimally managed watersheds (i.e., reference watersheds; Reynoldson et al. 1997).

The reliance on reference conditions to evaluate stream condition can be challenging given that few reaches are totally unaffected by human influence (Stoddard et al. 2006) and landscape characteristics of managed watersheds often differ from those of reference watersheds (Kershner et al. 2004b). Comparing the distributions of stream habitat conditions in managed reaches with the distributions in reference reaches can therefore be problematic unless there is an effort to account for inherent differences in landscape characteristics, such as watershed area, slope, and riparian vegetation (Hughes et al. 1986). While historic land use in both reference and managed watersheds can overwhelm the effects of current landscape characteristics on stream networks (Harding et al. 1998), quantifying the condition of physical habitat is an important component in the management and recovery of stream biota (Honea et al. 2009).

Despite these challenges, identifying the overall condition of stream habitat is necessary to evaluate the current status of stream habitat and increase the accountability of recent land management decisions and restoration efforts and is an important step in meeting the intended requirements of the U.S. Clean Water Act (Bauer and Ralph 2001). With this in mind, our objective was to develop an index of the condition of physical habitat for headwater streams that occur on federal lands managed by the U.S. Forest Service (USFS) and the U.S. Bureau of Land Management (BLM) within the interior Columbia River and upper Missouri River basins. Our intent was to incorporate information regarding the distribution of reach-level habitat conditions in streams within reference watersheds and use this information as a benchmark to compare similar habitat conditions in streams exposed to various management actions in this area. We evaluated a number of commonly measured habitat attributes in streams within reference watersheds and subsequently used these data to (1) evaluate the frequency distribution of physical habitat conditions in stream attributes in reference and managed reaches, (2) develop an index of stream condition by combining multiple attributes, (3) evaluate the distribution in physical habitat conditions among reference and managed reaches to quantify the status of physical habitat in our study area, and (4) assess the condition of streams in watersheds that experience different levels of management activity. Overall, our goal was to develop an analytical approach whereby a number of physical stream attributes that meet specific criteria and address explicit research and management objectives can be combined into an easily interpretable index of the condition of physical habitat.

Methods

Study Site and Sampling Design

We collected physical stream habitat data at the reach scale (160–500-m stream length; e.g., Frissell et al. 1986) within the interior Columbia River and upper Missouri River basins (Figure 1) using the approach

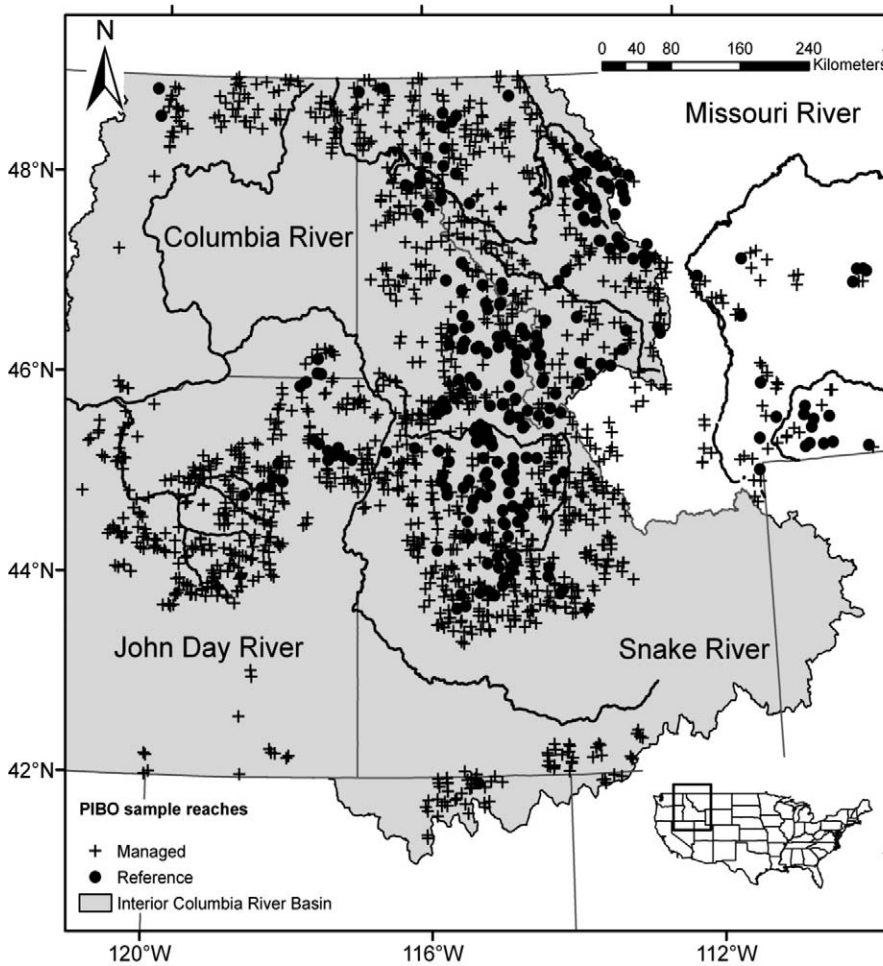

FIGURE 1.—Pacfish/Infish Biological Opinion Effectiveness Monitoring Program study area map, displaying the distribution of reference (black circles) and managed ("plus" symbols) sample reaches.

implemented by the Pacfish/Infish Biological Opinion Effectiveness Monitoring Program (Kershner et al. 2004a; Heitke et al. 2007). Stream reaches were located on federally managed land in catchments that were primarily (>50%) under federal management (BLM or USFS).

Sampled watersheds were selected using a spatially balanced random sample design (Stevens and Olsen 1999). In this study, we used reaches in minimally managed watersheds as reference reaches; reference watersheds included both wilderness areas and watersheds where there was (1) no permitted livestock grazing during the last 30 years, (2) minimal timber harvest (<10%), (3) minimal road density (0.5 km/km^2) at the watershed scale, (4) no roads within the proximate (1 km) riparian buffer, and (5) no evidence of historic mining within riparian areas (Kershner et al. 2004a). Watersheds where land management activities exceeded our reference criteria were considered managed and were subject to a variety of activities, including road building and maintenance, timber harvest, livestock grazing, mining, and motorized recreation.

Within each reference or managed watershed, we selected the lowermost low-gradient reach (<3%, based on visual observation) that occurred on federal land and that met our federal ownership rule of at least 50% of the catchment above the reach (average federal ownership of land was 99.6% for reference reaches and 95.4% for managed reaches). Private land ownership in the lower portion of many of our selected watersheds resulted in considerable variability in the size of the watersheds upstream from each sampled reach (here-

TABLE 1.—Average (SD) of geoclimatic attributes for reference and managed reaches in the study area, including percentage of land under federal ownership, 30-year average annual precipitation, drainage density, road density, slope, percent forested, stream gradient, reach elevation, number of sites grazed, and dominant geology (sedimentary, igneous, metamorphic, and unconsolidated).

Attribute	Reference	Managed
Percent federal ownership[a]	99.6 (1.5)	95.5 (8.74)
Precipitation (m)[a]	1.2 (0.3)	0.9 (0.3)
Drainage density (km/km^2)[a]	1.2 (0.3)	1.35 (0.46)
Road density (km/km^2)[b]	0 (0)	2.7 (2.9)
Slope (%)[b]	30.8 (17.8)	28.2 (14.8)
Percent forested (%)[b]	64.6 (20.3)	54.4 (23.4)
Bank-full width (m)[c]	7.9 (3.3)	5.6 (3.2)
Gradient (%)[c]	1.6 (0.9)	1.7 (0.9)
Elevation (m)[c]	1,548 (395)	1,413 (287)
Number grazed[a]	0	614 (14)
Sedimentary[a,d]	22	142
Igneous[a,d]	134	492
Metamorphic[a,d]	57	269
Unconsolidated[a,d]	4	28

[a] Quantified at the catchment scale.
[b] Quantified at the segment scale.
[c] Quantified at the reach scale.
[d] Count of reaches with the dominant geology type.

after referred to as catchment; size range = 400–15,135 ha).

We focused on sampling low-gradient reaches as these areas are generally thought to be the most sensitive to change under variable sediment and flow regimes (Montgomery and MacDonald 2002). If reaches with gradients less than 3% were not available, we did sample higher-gradient reaches, but for the analyses within this paper we included only those reaches where the measured gradient was less than 4% (see Appendix for additional sample design details).

Stream-Reach- and Catchment-Scale Data Collection

Between 2003 and 2007, we collected physical stream habitat data from 1,151 unique stream reaches; 217 of the catchments above the sampled reaches were reference, and 934 were managed. During the 2003 and 2004 field seasons, we sampled only streams within the interior Columbia River basin. The sampling area was expanded to include reaches in the upper Missouri River basin during the 2005–2007 field seasons.

Stream reach evaluation.—Streams were evaluated between June and September. At each reach, we measured 17 stream attributes that were shown to be often affected by land management activities (e.g., Woodsmith and Buffington 1996; Kershner et al. 2004b) or that were considered important for native fishes in headwater streams. The attributes considered in this index of physical habitat condition were sampled according to Heitke et al. (2007) and included width-to-depth ratio, percentage of banks with undercuts, average undercut depth, average bank angle, percentage of banks with an angle less than 90°, bank stability, streambed particle size (d_{16}, d_{50}, and d_{84}; see Appendix for definition of these symbols), percent of fine sediment in pool tails, frequency of large woody debris (LWD), volume of LWD, percent of pool habitat, frequency of pools, and average residual pool depth (see Appendix for specific sampling methods).

Landscape attributes.—Stream morphology can be affected by a variety of landscape characteristics that function over numerous scales (Burnett et al. 2006). To account for some of the inherent differences in catchments within and across management types, we quantified a set of geoclimatic attributes that can affect the physical habitat of stream networks and our expectations of the physical habitat condition within our sampled reaches (Table 1). We used ArcGIS version 9.2 (ESRI 2008) to calculate the area, stream density, average precipitation, and dominant geology within each catchment and the percent of forested vegetation, slope, and road density within a 90-m buffer on both sides of streams from the bottom of each reach and continuing upstream for 1 km (hereafter referred to as segment scale; see Appendix for specific methods).

Data Analysis: Selection of Physical Stream Habitat Attributes

We used physical stream habitat and landscape data from reference reaches as the basis for an index of physical habitat condition. We developed the index using an approach similar to that used for IBIs (Tejerina-Garro et al. 2006; Whittier et al. 2007; Stoddard et al. 2008). Within this framework, we identified candidate attributes from the 17 total attributes considered using the following sequence. First, we selected those physical habitat attributes that exhibited relatively low sampling variation. Next, we tested whether attributes with low sampling variation were responsive to management actions. Finally, we minimized redundancy of those attributes that met the specific criteria in the first two steps to avoid overweighting certain components of the physical instream habitat represented in the overall index.

Reproducibility in the measurement of physical stream habitat attributes.—During 2003 and 2004, we assessed the reproducibility of our field estimates by repeat-sampling a subsample of reaches twice within a 2-week period during the same year. We used these data to estimate signal-to-noise ratios (S:N) for each attribute (Kaufmann et al. 1999). We used linear mixed models (MIXED procedure in the Statistical Analysis System [SAS]; SAS Institute 2004), with

reach treated as a random effect, to decompose variance into estimates of signal (i.e., variability among reaches) and noise (i.e., variability within reaches) for all reaches that were repeat-sampled within a year. The ratio of the variance associated among streams was divided by the variance among field crews to determine S:N; all attributes with S:N less than 2.5 were deemed nonreproducible (Stoddard et al. 2008).

Responsiveness of attributes to management.—Given that our goal was to quantify differences in the overall condition of physical habitat in streams, we wanted to include attributes that differed as a function of land management activities (Mebane et al. 2003; Stoddard et al. 2008). As such, we evaluated the responsiveness of each attribute to management activities by comparing the means of each candidate attribute from reference reaches and managed reaches. We used road density as a surrogate for management effects, and for this comparison we constrained our analysis to managed reaches with high levels of management (i.e., median road density > 2.2 km/km^2 for managed reaches in the segment scale). After evaluating assumptions of normality, we tested for differences between these groups using the MIXED procedure in SAS (SAS Institute 2004). We included bank-full width and gradient as continuous covariates in this analysis to help control for variability in each attribute as a function of stream size and stream power (e.g., Kershner et al. 2004b); we included management (i.e., reference or managed) as a factor and treated all attributes, factors, and covariates as fixed effects. For this and all subsequent analyses, we used an alpha value of 0.10 due to the considerable variability in stream habitat analyses (Bryant et al. 2004) and to minimize the potential effects of stringent alpha values on the interpretation of our results (e.g., Reed and Blaustein 1997).

Redundancy analysis.—The final step in selecting attributes for the index involved evaluating the redundancy of the candidate attributes. We calculated Pearson's product-moment correlation coefficients for all remaining candidate attributes and considered attributes redundant if correlation coefficients exceeded 0.70 (CORR procedure in SAS; SAS Institute 2004). Similar to the concepts discussed by Stoddard et al. (2008), we retained those attributes that represented different geomorphic and structural measures of habitat condition, regardless of their redundancy. For those remaining attributes that were strongly correlated, we selected the attribute with the highest S:N value.

Data Analysis: Index Development and Evaluation

Scoring and evaluating the index.—We randomly selected 10% ($n = 18$) of the reference reaches as validation reaches (i.e., a subset of all reference reaches) and withheld these reaches to validate the broader applicability of the index. After selecting validation reaches, we qualitatively assessed the spatial distribution of these validation reaches to ensure adequate coverage of the distribution of reaches within our study area. For the remaining reference reaches ($n = 199$; hereafter referred to as calibration reaches), we began construction of the index using multiple linear regression (MLR) analyses to account for inherent differences among reaches (e.g., Pont et al. 2006). For each response attribute, we included the following as covariates in the MLRs: reach gradient and elevation, catchment area, drainage density in the catchment, average annual precipitation, dominant geology (categorical), average slope of the segment buffer, and forested extent of the segment area. We conducted a separate MLR analysis for each response attribute (REG procedure in SAS; SAS Institute 2004). Attributes measured in percentage (fine sediment and undercuts) were transformed using an arcsine–square root transformation; we checked for violations of linearity and heteroscedasticity using visual assessment of the residuals, and we tested for multicollinearity by calculating the variance inflation factor (i.e., values $>$ 10 indicate multicollinearity). For each analysis, we used stepwise regression with an alpha level of 0.10 to retain covariates.

We used the residuals from the MLR analyses with the calibration data to score each attribute (hereafter referred to as metric) for each reach. Residuals from these models were used rather than the raw data because the linear regression models accounted for some of the inherent differences among reaches (Kershner et al. 2004b). Similar to previous efforts (Hughes et al. 1998; Whittier et al. 2007), we scored the residuals for each metric continuously from 0 to 10. Within this framework, we used the 5th and 95th percentiles of the residuals as floor and ceiling values, respectively, and scored those values below the 5th percentile as 0 and those above the 95th percentile as 10.

To determine the overall index of physical habitat condition, we summed the individual metric scores included in the index. We then rescaled the index from 0 to 100 by dividing the index (i.e., the sum of individual metric scores) by the range of index values observed in all calibration reaches and subtracting the minimum index score for these calibration reaches.

Next, we applied the regression models developed for the calibration reaches to all validation and managed reaches. We initially calculated the residuals for each dependent variable for the validation reaches to assess the adequacy of our original model. Because

the model was based on a spatially balanced sampling approach (Figure 1) and the reaches used in the calibration were taken from a similar population as the validation reaches (i.e., reference), we hypothesized that there would be no significant difference in the means between the calibration and validation reaches. Next, we calculated the residuals for each attribute in each managed reach to evaluate potential differences in the condition of physical habitat in streams that are exposed to different land management activities. For both the validation and managed reaches, we scored the residuals for each metric from 0 to 10 based on the floor and ceiling values for the calibration reaches and summed the metric scores for reach-specific index scores, which were rescaled to a range from 0 to 100 using similar methods and values used for rescaling the calibration reaches.

Management and the condition of physical habitat.—We used the resulting metric and index scores to evaluate potential differences in the condition of physical habitat between calibration (i.e., reference) and managed reaches within our study area. We initially calculated the frequency distribution of each metric and overall index score for both calibration and managed reaches to qualitatively assess differences in the distributions of metric and index scores.

Next, we evaluated potential differences in average condition of physical habitat across all reference and managed reaches (i.e., a management effect) within our study area. In addition to the comparisons of overall means, we investigated how the condition of physical instream habitat related to different levels of land management activities. We evaluated the relationship between the index scores for each reach and road densities (km/km^2) at the segment scale and the presence or absence of livestock grazing within each watershed. Because consistent, catchment-specific information on grazing across our study area was not available, we used the presence or absence of grazing within each watershed (i.e., sixth field hydrologic unit code watershed) as an indication of grazing within our catchments. We incorporated livestock grazing as a factor and road density as a continuous variable, and we evaluated potential differences in the intercepts (i.e., additive effects) and slopes (i.e., interaction effects) of the linear relationships between road density and index scores in grazed and ungrazed catchments (all variables treated as fixed effects; MIXED procedure in SAS; SAS Institute 2004). We followed the methods outlined by Littell et al. (2006); we initially evaluated whether the slope of the relationship between habitat index score and road density differed from zero. We then assessed whether the slopes differed across grazing categories, and lastly we evaluated differences in the intercepts for each grazing category.

Results

Metric Selection

During 2003 and 2004, we repeat-sampled 37 reaches within 2 weeks of the initial sampling event. Based on these efforts, our assessments of reproducibility of the metrics indicated a wide range of S:N values for candidate metrics (median = 4.6; range = 0.1–28.7). As indicated by Stoddard et al. (2008), the cutoff criteria for S:N can vary for different analyses; here, we excluded four attributes that exhibited S:N values less than 2.5 (Table 2).

Our test for responsiveness included 466 managed reaches and 217 reference reaches. Our analysis indicated significant differences in all 11 candidate metrics that met the S:N criteria. Among these metrics, we found high correlations ($r > 0.70$) between (1) percent fine sediment less than 6 mm and percent fine sediment less than 2 mm ($r = 0.94$), (2) percentage of undercut banks and bank angle ($r = -0.94$), and (3) percentage of undercut banks and average undercut depth ($r = 0.92$). For the index, we excluded percent fine sediment less than 2 mm and average undercut depth due to the higher S:N ratios for percent fine sediment less than 6 mm and for bank angle, respectively. Although highly correlated, we included percentage of undercut banks and bank angle as these metrics provided considerably different information about the quality of instream condition (i.e., channel dimensions and overhead cover for fish). Thus, we included eight metrics in the final index of stream condition (Table 2).

Index Development and Metric Scoring

Based on the distribution of the residuals, we log transformed the d_{50}, LWD frequency, LWD volume, and residual pool depth. To include reaches with no LWD, we added one piece of LWD to the estimates of LWD frequency and 0.1 m^3/km to the estimate of LWD volume for each reach. We did not transform bank angle as the residuals were normally distributed. The average index scores for validation sites did not differ significantly from those for calibration sites ($t = -0.01$, df = 204, $P = 0.98$); therefore, we integrated the validation sites into the calibration data set and recomputed the regression models used to calculate residuals and score the individual metrics.

Model structure and fit of multiple linear regression models varied considerably for individual attributes (adjusted R^2 range = 0.14–0.56), indicating that near-stream, watershed, and climatic variables affected various attributes differently (Table 3). We found that

TABLE 2.—Mean (SE) values for the eight attributes measured in 217 reference reaches and 934 managed reaches included in the index of physical habitat condition; signal-to-noise ratio (S:N) for each attribute from 37 reaches repeatedly sampled by different crews; and F-values from an analysis of covariance used to distinguish between high-management sites and reference sites. High management was defined as road densities within the segment scale (which incorporated the 90-m buffer on each side of stream from the bottom of the reach upstream for 1 km), that were greater than the median for all managed reaches. We evaluated 17 total attributes as potential metrics in our index, and nine attributes were not considered due to (1) low S:N ratios (for width-to-depth ratio, both bank stability metrics, and d_{16}; see Appendix for definitions); (2) no distinguishable difference between reaches in highly managed watersheds and reference watersheds (for pool frequency and d_{84}); and (3) high correlation with other attributes (for percent fine sediment less than 2 mm, undercuts determined by bank angle less than 90°, and average undercut depth).

Attribute	S:N	Managed mean (SE)	Reference mean (SE)	F	P
Bank angle (°)	2.8	108.0 (0.7)	99.3 (1.4)	33.8	<0.001
Percent undercut[a]	2.9	26.4 (0.6)	32.7 (1.3)	15.15	<0.001
d_{50} (m)[b,c]	4.6	0.043 (0.001)	0.058 (0.003)	3.2	0.07
Percent fine sediment (<6 mm)[a]	7.8	26.7 (0.9)	18.0 (1.4)	3.46	0.01
LWD[d] volume (m^3/km)[b]	8.2	69.9 (3.4)	139.9 (11.8)	22.5	<0.001
LWD frequency (pieces/km)[b]	6.1	197 (7.0)	388 (26.3)	27.0	<0.001
Residual pool depth (m)[b]	28.7	0.26 (0.004)	0.31 (0.01)	31.1	<0.001
Percent pool habitat[a]	4.9	40.9 (0.6)	43.3 (1.6)	3.56	<0.001

[a] Arcsine–square root tranformed.
[b] Natural log transformed.
[c] Median substrate particle size.
[d] Large woody debris.

reaches in larger catchments contained less fine sediment and pool habitat, deeper pools, larger d_{50} values, more obtuse banks, and a lower percentage of undercut banks. We found that reaches in catchments with higher mean annual precipitation had less fine sediment, less pool habitat, a lower percentage of undercut banks, larger d_{50} values, deeper pools, and more obtuse bank angles. Stream reaches occurring in catchments with higher drainage densities contained more fine sediment, more pieces of LWD, and larger d_{50} values.

We found that the dominant geology explained a significant amount of variability in substrate and bank attributes. Reaches where the dominant geology of the catchment was igneous contained a greater percentage of undercut banks, steeper bank angles, and higher amounts of fine sediment. In contrast, catchments where the dominant geology was sedimentary had a lower percentage of undercut banks and higher bank angles (i.e., more obtuse).

Local riparian conditions accounted for a significant amount of variability in LWD, substrate characteristics, and channel shape. In particular, we found a positive relationship between the amount of forested habitat within the segment scale and the frequency and volume of LWD within reaches. Where the slope of the segment was steeper, reaches had higher bank angles

TABLE 3.—Parameter estimates, model structure, and model fit (R^2) from reference reach multiple regression models used to score individual metrics for each dependent variable in the index (area = catchment area, km^2; precip = average annual precipitation, m; drainage den = the density of streams within the catchment, km/km^2; ign = a categorical variable denoting whether the dominant geology is igneous; grad = reach gradient, %; elev = elevation of the bottom of the reach, m; % segment forested = percentage of the riparian buffer [90 m on each side of stream] that is forested 1 km upstream from the bottom of reach; sed = a categorical variable denoting whether the dominant geology is sedimentary).

Attribute	Regression model	Adjusted. R^2
Percent undercut[a]	0.98 − 0.06(grad) − 0.15(precip) − 0.002(area) + 0.08(ign) − 0.18(sed)	0.35
Bank angle (°)	58.1 + 6.7(grad) + 14.8(precip) + 0.29(area) + 0.2(segment slope) − 8.9(ign) + 11.7(sed)	0.39
D_{50} (m)[b,c]	−5.5 + 0.63(grad) + 0.65(precip) + 0.02(area) − 0.43(drainage den) − 0.32(ign) + 0.0003(elev)	0.56
Percent fine sediment (<6 mm)[a]	0.76 − 0.004(area) − 0.11(grad) − 0.19(precip) + 0.12(drainage den) + 0.09(ign)	0.36
LWD[d] volume (m^3/km)[b]	5.1 + 0.02(% segment forested) − 0.02(segment slope) − 0.001(elev)	0.17
LWD frequency (pieces/km)[b]	4.1 + 0.02(% segment forested) − 0.02(segment slope) + 0.48(drainage den)	0.14
Residual pool depth (m)[b]	−1.1 − 0.24(grad) + 0.004(area) + 0.25(precip)	0.33
Percent pool[a]	1.6 − 0.2(grad) − 0.003(area) − 0.0001(elev) − 0.20(precip)	0.44

[a] Arcsine–square root transformed.
[b] Natural log transformed.
[c] Median substrate particle size.
[d] Large woody debris.

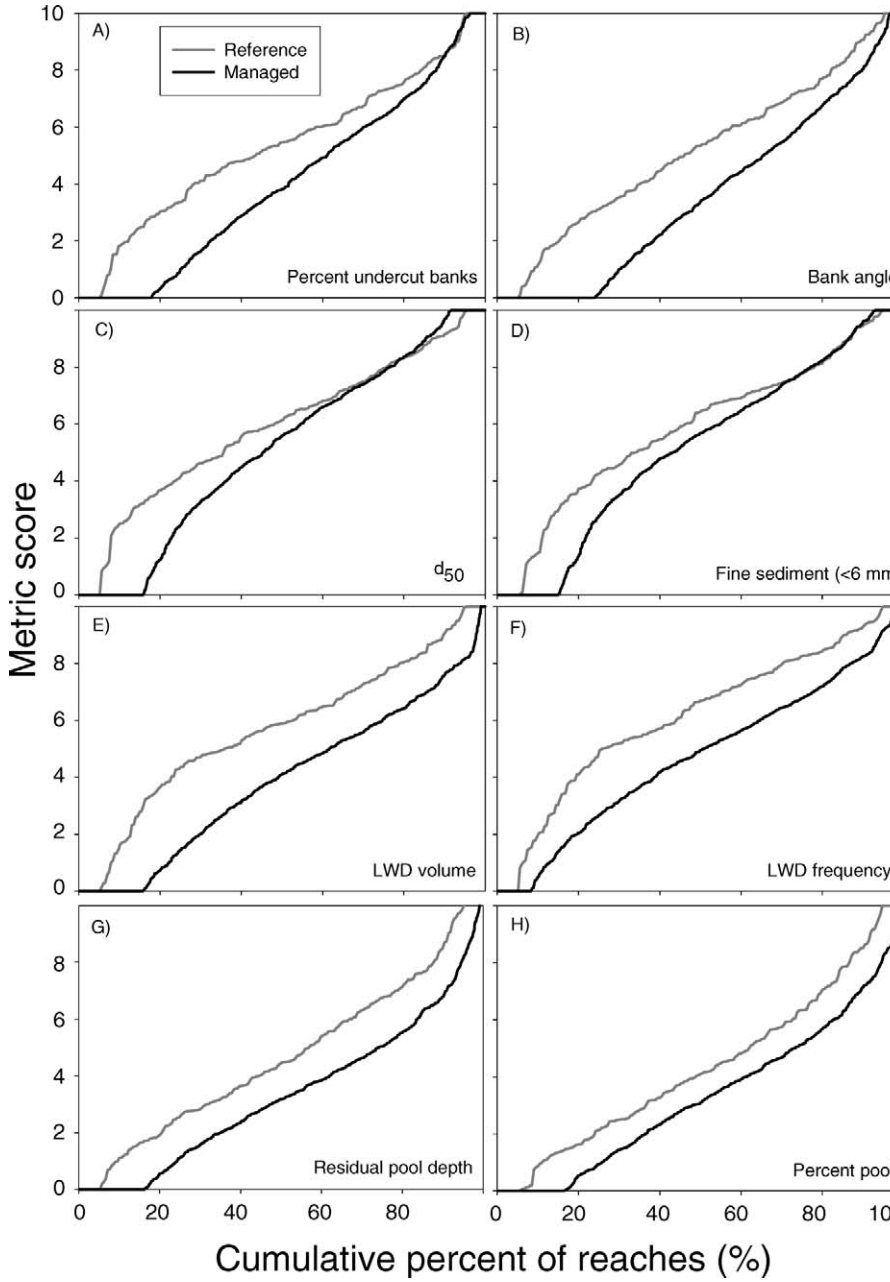

FIGURE 2.—Cumulative frequency distributions of reference (gray lines) and managed (black lines) reaches with metric scores for **(A)** percentage of undercut banks, **(B)** bank angle, **(C)** median substrate particle size (d_{50}), **(D)** percentage of fine sediment less than 6 mm, **(E)** large woody debris (LWD) volume, **(F)** LWD frequency, **(G)** residual pool depth, and **(H)** percent pool.

and lower amounts and volume of LWD. Higher-gradient reaches contained less fine sediment and pool habitat, shallower pools, larger d_{50} values, a greater percentage of banks with undercuts, and more obtuse bank angles. Reaches at higher elevations contained larger d_{50} values, smaller volumes of LWD, and less pool habitat.

Cumulative frequency distributions of individual metric scores.—The cumulative frequency distribution of each of the eight metrics revealed three interesting

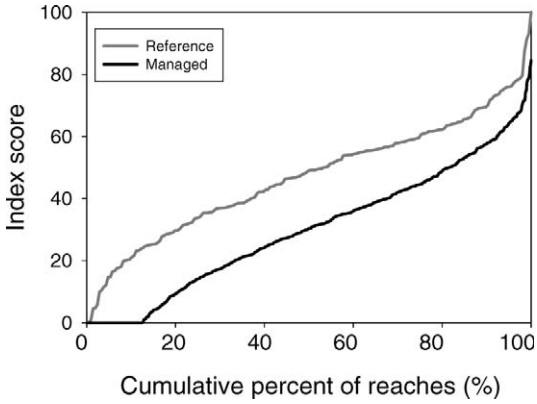

FIGURE 3.—Cumulative frequency distributions of reference (gray line) and managed (black line) reaches with a given index score of physical habitat condition.

patterns (Figure 2). First, there were substantial differences in the frequency of reference and managed reaches with relatively low metric scores (i.e., <4.0). On average, 30.1% (range = 20.4–48.6%) of reference reaches had metric scores below 4.0, which was considerably lower than the 49.2% observed in managed reaches (range = 33.6–61.8%). Next, despite the differences in the frequency of reaches with low metric scores, there appeared to be little difference in the percentage of reaches with high metric scores (>7.0) between reference and managed reaches for d_{50} (36.1% and 35.0%, respectively) and percent fine sediment (38.7% and 33.3%, respectively). Finally, we found a higher percentage of reference reaches than managed reaches with high metric scores for the percentage of undercut banks (28.5% and 20.7% for reference and managed reaches, respectively), bank angle (28.5% and 18.3%), LWD frequency (38.4% and 19.8%), LWD volume (35.6% and 17.7%), percent pool habitat (20% and 10.5%), and residual pool depth (21.5% and 9.1%).

Index Scores

Management analyses.—The mean (±SE) index score for all managed reaches (30.1 ± 0.68, $n = 883$) was significantly lower ($F = 118.2$, df = 1,087, $P < 0.0001$) than that observed for reference reaches (47.0 ± 1.4, $n = 206$). The cumulative frequency distribution of index scores indicated that (1) the percentage of reaches with index scores less than 40 was substantially lower for reference reaches (37%) than for managed reaches (68%) and (2) the percentage of reaches with index scores greater than 70 was higher for reference reaches (10%) than for managed reaches (2%; Figure 3).

Land management activities and condition of physical habitat.—The average (±SE) road density in the segment scale above managed reaches was 2.64 ± 0.09 km/km^2, and 614 catchments upstream from our sample reaches contained livestock grazing allotments. We found a significant negative relationship (mean slope ± SE = −1.11 ± 0.22) between road density and the index scores of streams ($F = 24.7$, df = 880, $P < 0.0001$). When catchments were delineated based on the presence or absence of livestock grazing, we did not find a significant interaction effect ($F = 1.02$, $P = 0.32$) in the relationship between road density and index score for reaches within grazed and ungrazed catchments. However, we did find an additive grazing effect ($F = 82.6$, df = 879, $P < 0.0001$) wherein the intercept for grazed catchments (mean intercept ± SE = 26.3 ± 0.8) was significantly lower than that for ungrazed catchments (39.0 ± 1.1). Together, these results suggest a consistent negative slope in the relationship between the condition of physical habitat and road density in grazed and ungrazed catchments but significantly lower condition of physical habitat in grazed catchments for a given road density.

Discussion

Development of the Index

Within our study area, we observed a wide range of stream conditions in both reference and managed streams. This observed variability in stream condition is probably a reflection of recent and historical land management and natural disturbances (Woodsmith and Buffington 1996; Kershner et al. 2004b). For example, substantial fires (>10 ha; average = 1,557 ha) have occurred in 101 of the 210 reference reach catchments since 1995 (USFS and USGS 2009). Part of this variability in the condition of reference reaches may also be due to the legacy effects of past management activities within our reference catchments that continue to shape the physical habitat of these headwater streams (Moore and Wondzell 2005). Therefore, it is clear that our use of the term "reference" should not be confused with "undisturbed." In spite of this high natural disturbance rate in reference watersheds over the past 25 years and the potential legacy effects of past management practices, our index of physical habitat condition demonstrated the potential to differentiate between managed and reference reaches as well as provide a simple measure to help managers understand stream condition (Bisson et al. 1997).

One of the strengths of this multimetric approach is the incorporation of a multivariate tool (i.e., regression) to account for some of the variation in stream conditions due to difference in stream power and landscape features (Kershner et al. 2004b; Pont et al. 2006). By using regression, we were able to account

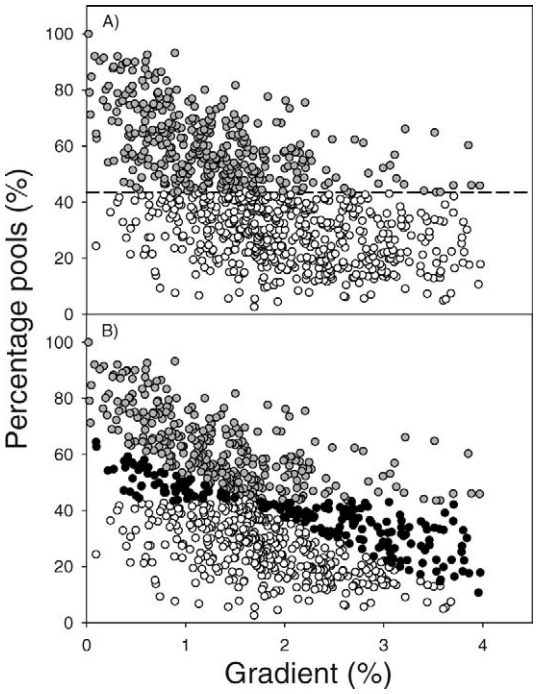

FIGURE 4.—Plot of all managed reaches, illustrating the potential misclassification of individual managed reaches (black circles) when scoring the stream reach condition based on **(A)** the average value (dashed line) of percent pools from reference reaches (43%) without inclusion of covariates versus **(B)** consideration of the geomorphic relationship between gradient and percent pool via simple linear regression with gradient as a covariate. In both panels, individual reaches categorized as above average are represented as gray circles and those categorized as below average are represented as open circles.

for differences across watersheds that can substantially affect our expectations of stream conditions. For example, pool-forming processes vary with respect to differences in gradient; high-gradient reaches generally have a lower percentage of area as pools than do low-gradient reaches (Wohl et al. 1993). If we had not accounted for this relationship when scoring the metric for percent pools, we would have classified all reaches with percent pools above the overall mean value of 43% as having relatively high condition for this metric and all reaches below the mean as having relatively low condition (Figure 4a). In our analyses, we found that gradient accounted for a significant amount of variability in the percentage of pool habitat within reaches (Table 3). By incorporating gradient, there is now a sliding scale of expectations as those reaches with higher gradients are expected to have lower values of percent pools than lower-gradient reaches and vice versa (Figure 4b). Incorporation of covariates not only permits the formation of an index over larger spatial scales and over a greater range of stream sizes, it also increases the accuracy and precision of those estimates.

Our use of regression differs from other predictive habitat assessment approaches that rely on either quantitative (Jeffers 1998; Davies et al. 2000) or qualitative (Raven et al. 1998; Thomson et al. 2001) classification to minimize variation within groups. Such approaches are valuable in the prediction of biotic assemblages (Hawkins et al. 2000), where the classification of landscapes can control for the observed variation among biota. However, landscape classifications may not help to improve prediction of stream conditions as local stream shape is dominated by stream power (Knighton 1998) and juxtaposed stream reaches can have dramatically different characteristics, such as gradient or streamflow (inclusion of a tributary), over small distances. Furthermore, identifying the criteria for designating classifications of landscape and environmental covariates can be extremely difficult. As a result, we advocate the use of continuous covariates to reduce the variance in variables of interest by accounting for inherent differences in the landscape and climate across sample reaches, particularly for spatially extensive analyses.

Reference versus Managed Streams

Individual metrics.—Evaluating the effectiveness of management actions will benefit from monitoring specific stream attributes. Recent management decisions have increased the protection of riparian and stream systems from land management activities in managed watersheds to restore instream conditions (USFS 1995; USFS and BLM 1995). For example, stream buffers for fish-bearing streams were increased in the mid-1990s to a minimum of 90 m so as to minimize the effects of land use activities on stream conditions. In lieu of these past management changes, our results from recent field surveys were similar to those from previous assessments (1999–2001; Kershner et al. 2004b), which found that reaches within managed watersheds contained significantly shallower residual pool depths, a lower percentage of undercut banks, higher bank angles, and smaller d_{50} values than reference reaches. However, Kershner et al. (2004a) did not find significant differences in pool tail fines or the percent of pool habitat between reference and managed reaches; this was probably the result of the small sample size of reference sites used in these analyses. The consistency of our results, which encompass a substantially broader sample of reference and managed streams in our study area compared with Kershner et al. (2004a), supports the findings that the quality of physical habitat in headwater systems within the interior Columbia River and upper Missouri River

basins has been and continues to be degraded by current management actions, past management actions, or both.

In addition to differences in metric scores, evaluations of the distributions of the metric scores help describe how management has altered stream conditions. The general expectations for management effects are a shift in the overall distribution of managed reaches relative to reference reaches (Platts 1991). Six attributes used in this index (percentage of undercut banks, bank angle, LWD frequency, LWD volume, residual pool depth, and percent pool) have distributions that indicated this type of shift (Figure 2). Shifts in the distribution of each of these physical habitat attributes are troubling because they indicate that relative to the reference reaches, a considerably smaller proportion of the managed stream reaches received high metric scores (i.e., >7.0) and a higher proportion of managed reaches were given low metric scores (i.e., <4.0). Ultimately, these results indicate that land use practices aimed at meeting management standards can result in an overall shift in the distribution and an additional reduction in the proportion of streams with the highest condition as streams tend to be managed down to a standard (Bisson et al. 1997). For example, managed reaches exhibited substantially lower percentages of undercut banks and higher bank angles than were observed in reference reaches. These observed differences are probably the result of current and historical grazing practices (Kauffman and Krueger 1984; Platts 1991), road construction (Furniss et al. 1991), and forest harvest (Dose and Roper 1994), all of which can decrease streambank stability and increase bank angle. This pattern was also apparent for the frequency and volume of LWD. Possible reasons for fewer LWD pieces and lower LWD volume in managed streams have been examined elsewhere (Kauffman et al. 1997; Roni et al. 2002) and include historic wood removal from the active channel, historic conversion of older riparian stands to younger stands, and loss of recruitment due to timber harvest, livestock grazing, and fire. Percent pools and residual pool depth also illustrated overall shifts in the distribution of metric scores in managed reaches. These reductions in pool quantity and quality in managed reaches are probably due to (1) fewer stream reaches possessing the high LWD levels that can provide critical structure for the formation and scouring of pools (Bilby 1981; Montgomery et al. 1995); (2) increased width-to-depth ratios from reductions in bank stability (Belsky et al. 1999); (3) increases in fine sediment (McIntosh et al. 2000); or (4) some combination of these factors. Together, the observed reductions in the condition of LWD and pool attributes in managed reaches suggest that one of the most profound effects of management is the loss of the high-quality, complex pool habitat.

We observed a different distribution pattern of metric scores for d_{50} and pool tail fines (Figure 2); frequencies of reference and managed reaches with high metric scores were similar, but the frequency of reaches with low metric scores was considerably higher for managed reaches than for reference reaches. Our observations of small d_{50} values and increasing pool tail fines (i.e., a high frequency of managed reaches with low metric scores) fit well with the expectations that management activities can increase the amount of fine sediment inputs from hillsides (Myers and Swanson 1995) and streambank disturbances (Platts 1991). On the other hand, the similarity in the frequency of reference and managed reaches with high metric scores for d_{50} and pool tail fines does not fit our expectations. These results may come from two separate populations of reaches. The first may consist of reaches where the levels of fine sediment are low due to the lack of management activities. Conversely, the low levels of fine sediment may be the indirect result of overall reductions in channel roughness in managed reaches due to greater bank angles, lower amounts of LWD, and smaller bedform fluctuations (i.e., shallower and fewer pools; Buffington and Montgomery 1999; Figure 2). Ultimately, simplified channels in managed areas may be more efficient at transporting fines through the reach, thus acting more like irrigation canals rather than natural stream systems (Buffington and Montgomery 1999).

Overall assessment of the condition of physical habitat.—While evaluating the condition of individual attributes provides insight into specific differences between reference and managed areas (Rheinhardt et al. 2007), the overall condition of streams is probably more relevant for evaluating stream integrity for fishes (Vadas and Orth 2001) and for use in decision-making by managers. The index scores for our study area indicated that stream conditions within managed landscapes were significantly lower than those observed in reference areas. These observed differences in the average condition of streams within reference and managed catchments may indicate that recent management regulations (e.g., riparian buffers) in combination with the legacy of previous management actions (e.g., Harding et al. 1998) may not be sufficient to improve the status of streams within managed watersheds, particularly over relatively short time periods (i.e., 10–20 years).

Our assessment of the cumulative distribution of the index scores indicated a greater frequency of managed reaches with low habitat condition and a lower frequency of managed reaches with high habitat

condition than observed in reference reaches. While the effects of these types of shifts in physical habitat condition on fish are not documented here, the degradation of physical habitat significantly reduces fish production (Scheuerell et al. 2006) and distribution (Thurow et al. 1997) and increases the invasion success of nonnative fishes (Shepard 2004; Benjamin et al. 2007). The loss of high-quality stream reaches may significantly affect the viability of a species, particularly where persistence is related to the disproportionate capacity of high-quality physical habitat to produce fish. Under such circumstances, the loss or decline of robust populations that act as a source population to other, less-viable populations (e.g., Pulliam 1988) may be directly and indirectly detrimental to the overall metapopulation.

Land management activities and condition of physical habitat.—The negative effects of roads on physical instream habitat and aquatic biota have been well documented (see review by Trombulak and Frissell 2000). The presence of roads can lead to substantial alterations of the catchment hydrology, sediment load, and channel structure (Chamberlin et al. 1991; Jones et al. 2000) and can have significant effects on instream biota (Eaglin and Hubert 1993; Baxter et al. 1999). Here, we found a negative relationship between the index scores in managed reaches and road density, indicating that the effects of high road densities and the activities associated with forest roads (e.g., timber harvest) can significantly reduce the overall condition of instream physical habitat.

Our analyses indicated that the presence of cattle grazing in sampled catchments significantly reduced the condition of physical habitat in the headwater reaches included in our analyses. The effects of cattle grazing on instream habitat have been summarized previously (Platts 1991; Belsky et al. 1999) and include increased width-to-depth ratios, higher levels of fine sediment resulting from collapsed banks, fewer undercut banks, increased bank angles, reductions in pool habitat, and degradation of the riparian zone. In general, we found that reaches with the lowest condition occurred in grazed catchments with high road densities, which suggests that the synergistic effect of these disturbances is more detrimental to the physical habitat than each management action alone (Myers and Swanson 1995). While legacy effects of cattle grazing may still occur in some of our reference watersheds (e.g., Harding et al. 1998), our results indicate a divergence in the average condition of habitat in streams where grazing had not occurred within the last 30 years and streams within watersheds that continue to experience grazing practices.

We acknowledge two main limitations in our assessments of the relationships between management activities and the condition of physical habitat. First, our summary of road densities included only roads maintained by the USFS and BLM and did not include decommissioned roads or motorized recreational trails. While finer-resolution data are generally available at the individual forest level, the inconsistencies in data across all forests and BLM districts ($n = 30$) in our study area prevented the inclusion of these data in our analyses. Next, due to the lack of available grazing data or consistently collected grazing data, we were unable to include information describing livestock grazing in terms of its intensity, timing, or proximity to sampled reaches in our analyses; each of these variables can substantially affect the impacts of grazing on instream physical habitat (Platts 1991; Myers and Swanson 1995) and aquatic biota (Saunders and Fausch 2007). Despite these shortcomings, our results suggest that increased management activities are negatively related to the condition of instream physical habitat; our results also highlight the need for more robust data on the extent of management actions and landscape disturbances to better understand the mechanistic linkages between management and the physical habitat of headwater streams. Ultimately, our results demonstrate that land managers need to incorporate the additive effects of multiple land management activities when prescribing future management standards that will prevent further degradation of streams and begin to restore physical habitat.

Conclusions

It has been argued that the merit of an index of condition increases when the index can account for the natural range of conditions observed across relevant spatial scales (Smogor and Angermeier 2001). This has been done for IBIs by building different models for different locations (Lyons et al. 1996; Hughes et al. 2004) and ecoregions (Whittier et al. 2007) or by predicting the presence of specific species through models incorporating landscape characteristics (Hawkins et al. 2000). Currently, most evaluations of physical stream habitat condition are based on specific habitat standards that are used for the entire population of interest or for several strata within that population (e.g., stream size; USFS 1995, USFS and BLM 1995). The use of threshold values, however, may be a naïve approach in evaluating stream condition due to the considerable variability of watershed state as a result of natural disturbance regimes (e.g., fire; Roper et al. 2007), continuous relationships between landscape attributes and instream physical habitat (e.g., gradient and percent pool as illustrated above; Knighton 1998), and our understanding of how stream power affects

most physical stream characteristics (e.g., sediment; Kaufmann et al. 2008). In general, the failure to implicitly incorporate natural variability into measures of the condition of stream habitat may result in misleading conclusions, as we have illustrated.

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Appendix: Development of the Physical Habitat Condition Index

Sample Design

We organized our study area into 227 blocks (interior Columbia River basin = 177 blocks, upper Missouri River basin = 50 blocks) of 20 contiguous sixth field hydrologic unit code (HUC) watersheds. Each year, we randomly selected (without replacement) 20% of the total number of blocks (approximate size = fifth field HUC watersheds) to determine which blocks would be sampled in a given year and to ensure a temporally balanced design. We assigned each of the sixth field HUC watersheds (hereafter referred to simply as watersheds) within each block as either reference or managed based on current and historical management activities.

Once a block was selected, we randomly selected 7 of the 20 watersheds within that block for instream habitat sampling. Due to the relatively low number of watersheds meeting our definition of reference watersheds within each block, we targeted sampling in four reference watersheds and three managed watersheds. If sufficient numbers of watersheds of either type were not available, we increased the number of the other type of watershed sampled.

Justification of and Sampling Methods for Stream Reach Attributes

To establish the overall reach length, we initially measured bank-full width at four random locations and used the average of these four measurements to categorize reaches into 2-m width categories with a minimum bank-full width category of 8 m and a maximum category of 25 m. We determined the reach length as 20 times the bank-full width category to increase the likelihood that multiple riffle–pool sequences were sampled in each reach (Knighton 1998). After establishing the overall reach length, we measured 17 stream attributes to characterize the status of each reach.

Stream depth is an important habitat attribute for headwater fishes (Northcote and Ennis 1994; Bateman and Li 2001) and is generally evaluated using a proxy of width-to-depth ratio. To quantify the average width-to-depth ratio, we measured bank-full widths and depths at each of the first four riffles in a reach. We measured depths at 10 equally spaced locations perpendicular to the thalweg between the top of bank-full height on each side of the stream and calculated the average width-to-depth ratio for the reach.

Undercut and vertical banks provide cover for fish in headwater systems (Watson and Hillman 1997), and streambank failure can be a significant source of fine sediment in streams (Platts 1991). To quantify streambank characteristics, we established a minimum of 20 equally spaced transects placed perpendicular to the water's flow at bank-full stage height. At each of these transects, we measured bank angle, bank stability, and undercut depth on both streambanks. Our measurement of bank

attributes focused on the area between the coarser streambed substrate up to the bank-full elevation. All attributes were measured on both banks ($n \approx 40$), and the presented values are reachwide averages. We measured bank angle using a clinometer placed on a rod that followed the bank's contour (Platts et al. 1989). We quantified two measures of bank stability using a dichotomous key (see Heitke et al. 2007), which classified areas as either vegetated or bare and either stable or unstable. One measure of bank stability (stability 1) required vegetative cover for banks to be categorized as stable. The other measure (stability 2) included both vegetated and unvegetated stable banks in the classification of bank stability. We quantified undercut banks using two criteria: (1) streambanks with angles less than 90° and (2) banks that were undercut by at least 5 cm in depth and at least 10 cm in height and in width (Kershner et al. 2004a).

Substrate can be an important source of refugia from predation and high-flow events for headwater fishes (Roni 2002; Anderson 2008). To assess the substrate composition of each reach, we collected five surface bed particles at each of the 20 transects used to estimate streambank characteristics. Pebbles were collected at equal intervals along transects, and the *b*-axis (i.e., intermediate axis) of selected particles was measured (Kondolf 1997). Only particles collected within the active channel (no bank material) were included for analysis. From this sample, we estimated the particle size at which 16% of the particles were smaller (d_{16}), the median particle size (d_{50}), and the particle size at which 84% of the particles were smaller (d_{84}).

Because most trout and salmon spawn in shallow habitat that is proximate to or within pools, the amount of fine sediment in these areas can have a direct effect on egg-to-fry survival (Bjornn and Reiser 1991). We evaluated how much of the lowermost 10% of each habitat unit designated as a pool (see below) was covered by surface fines by randomly placing a 0.35- × 0.35-m grid with 50 intersections (49 equally spaced and upper-right corner of grid) at three equidistant locations across the wetted width. We then counted the number of intersections that overlaid particles less than 2 mm or less than 6 mm. Total grid counts for each of these size categories were divided by the total number of intersections in the grid. We calculated mean estimates of percent fines for each pool and then averaged the pool estimates to obtain an overall reach estimate for each category of percent fine sediment.

Large woody debris (LWD) is an important source of cover and breaks in flow velocity for many fishes in headwater streams (Roni 2003) and is a critical component in the formation of complex stream habitat (Montgomery et al. 1995). Within each reach, we enumerated all LWD pieces greater than 10 cm in diameter (measured at one-third of the way up from the base) and greater than 1 m in length for estimates of LWD frequency (number of pieces/km). We quantified the volume of LWD in each reach by summing the estimated volume of each piece of LWD.

The frequency and dimensions of pool habitat are important stream features affecting the distribution and abundance of headwater fishes (Roper et al. 1994; Roni 2002). We defined pools as habitat areas that are concave in profile, both laterally and longitudinally, and that are bounded by the steepening of the streambed slope on the upstream portion of the units (i.e., pool head) and a downstream break in streambed slope (i.e., pool tail crest). Minimum criteria defining a pool included two components: (1) the maximum depth had to be at least 1.5 times greater than the water depth at the pool tail crest and (2) the length of the unit had to be equal to or greater than the wetted channel width (Heitke et al. 2007). We measured the length of each reach along the thalweg, and the portion of the total reach length deemed to be pool habitat was used as an estimate of percent pool. We also divided the number of pools by the reach length and multiplied this estimate by 1,000 to calculate the pool frequency per kilometer. In each pool, we measured the residual pool depth as the maximum pool depth minus the depth at the pool tail crest; the reach-level average residual pool depth (m) was then calculated.

Finally, we estimated bank-full width and channel gradient within each reach to control for inherent differences in stream power (Knighton 1998) across reaches (to be used as covariates; see Kershner et al. 2004b). We measured bank-full width at each of the 20 transects used to evaluate streambank characteristics and calculated the reach average bank-full width. Because bank-full dimensions can be altered by management activities, however, we only included this attribute as a covariate in the metric selection process and not in the scoring of the metrics (see next section). We estimated channel gradient in the field by measuring the change in elevation of the water surface between the top and bottom of the reach using a site level and tripod and dividing this elevation change by the reach length.

Justification and Estimation of Landscape Attributes

We delineated each sample reach's catchment with digital elevation models (DEMs) and Arc Macro Language using ArcGIS version 9.2 (ESRI 2008). Once delineated, we calculated the catchment area and stream drainage density, both of which can affect the timing and magnitude of streamflow (Benda et al. 2004). We used these catchments in conjunction with stream layers from the 1:24,000-scale U.S. Geological Survey (USGS) National Hydrography Dataset (USGS 2000) to calculate stream drainage density in each catchment.

Given the high variability of precipitation across the catchments within our study (range = 327–942 mm) and the possibility that these differences could substantially alter stream channel processes and therefore alter the expectations for individual channel attributes within the evaluated stream reaches (Knighton 1998), we estimated the average annual precipitation for each catchment. We calculated the average precipitation of each catchment as the weighted average (by area) of all precipitation grids (16 km^2) that were intercepted by each individual catchment. For each grid, we used 30-year average precipitation values (1971–2000; PRISM 2004).

The spatial extent of the Pacfish/Infish Biological Opinion Effectiveness Monitoring Program study area dictated the need to control for differences in the geology, which can significantly influence the size distribution of substrate within streams (Knighton 1998), bank morphology (Nelson et al. 1992), and delivery of LWD (Burnett et al. 2006). Here, we quantified the percent of each catchment with sedimentary, igneous, metamorphic, and unconsolidated geologies (USGS 2005) and then determined the dominant geology for each reach's catchment.

The structure of the riparian vegetation can have substantial effects on the morphology of streambanks (Wynn and Mostaghimi 2006) and LWD recruitment within small stream channels (May and Gresswell 2003). We quantified the extent of forested habitat (i.e., all tree-dominated vegetation) at the segment scale using land cover data from the Landscape, Fire, and Resource Management Planning Tools Project (LANDFIRE 2008). In addition to vegetation, we quantified the slope at the segment scale using 30-m DEMs obtained from USGS 1:24,000-scale topographic data (USGS 1999) and the density of roads at the segment scale from the USFS Geodata Clearinghouse (1:24,000 scale; USFS 1995).

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