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ARTICLE

Quantifying the Extent of and Factors Associated with the Temporal Variability of Physical Stream Habitat in Headwater Streams in the Interior Columbia River Basin

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Abstract

The quality and quantity of stream habitat can have profound impacts on the distribution and abundance of aquatic species. Stream networks, however, are dynamic in their response to natural- and human-induced disturbance regimes, which results in spatially explicit patterns of temporal variability. Quantifying spatial patterns in habitat (temporal) variability across different sites and identifying those factors associated with different levels of variability are important steps for stream habitat assessments. We evaluated the temporal variability in stream habitat over a 9-year period for 47headwater streams of the interior Columbia River basin. We used repeat-measures analyses to calculate temporal variability as root mean square error for six habitat attributes at each site. Multiple linear regression analyses with root mean square error as the response were then used to quantify which landscape, climate, and disturbance attributes were associated with different levels of temporal variability among habitat attributes. Our results indicated a considerable range of temporal variability in physical stream attributes across sites and an almost fourfold difference in the overall variability at sites. Landscape factors affecting stream power, land management activities, and recent fire regimes were all factors associated with the different levels of temporal variability across sites; surprisingly, we found little association with the different climatic attributes considered herein. The observed differences in temporal variability across sites suggest that a "one-size-fits-all" approach to monitoring stream habitat in response to restoration and management activities may be misleading, particularly in terms of sampling intensity, required resources, and statistical power; thus, in situ measures of temporal variability may be required for accurate assessments of statistical power.

The important influence of physical habitat on the distribution and abundance of stream biota is well documented (Southwood 1977; Minshall et al. 1983; Riley and Fausch 1995). Consequently, quantifying the status and trends of stream habitat is critical for understanding the factors that potentially limit populations (Nickelson and Lawson 1998), for quantifying restoration effectiveness (Bernhardt et al. 2005), for determining how land management activities impact stream ecosystems (Kershner et al. 2004b), and for directing future management plans and restoration efforts (Burnett et al. 2007). However, robust evaluations of habitat status and trends can be difficult as multiple sources of variability can impede our ability to accurately evaluate the structure of stream habitat (Larsen et al. 2004). As such, identifying and minimizing sources of variability are important steps in the design of effective monitoring efforts and in generating expectations of the statistical power to detect changes (Urquhart et al. 1998; Larsen et al. 2001).

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Much of the focus in habitat monitoring has addressed variability associated with sampling error, site-to-site differences, year effects, and site × year interactions. Numerous efforts have identified sampling error as a significant source of variability in habitat assessments (Kaufmann et al. 1999; Whitacre et al. 2007). Sampling error can vary considerably across stream habitat attributes, and selection of those habitat attributes (and protocols) that exhibit low levels of sampling error can reduce variability and improve the power to detect changes in physical habitat (Roper et al. 2010). Recent efforts have illustrated the need for repeated sampling of the same monitoring sites through time, as high site-to-site variability can limit inferences of change in habitat structure over relevant time frames (Urquhart et al. 1998; Larsen et al. 2004; Anlauf et al. 2011). Year effects (or other relevant time frames) are changes in habitat structure that result from yearly climatic patterns (e.g., pool scour during floods; Lisle and Hilton 1999) but do not necessarily result from the particular action of interest (e.g., channel restoration; Larsen et al. 2001). In general, year effects tend to be relatively insignificant across large spatial scales as climate patterns can vary within and across different ecoregions; however, year effects may be considerably higher for smaller regional analyses (Urquhart et al. 1998). Finally, site × year interactions can also result in substantial sources of variability as sites can respond differently to yearly climate patterns due to inherent site-specific characteristics (Larsen et al. 2001). Larsen et al. (2004) observed considerable site × year interactions, suggesting that sites may change differentially through time due to inherent differences in landscape characteristics (e.g., slope of watershed) and climate patterns.

Despite rigorous assessments of stream monitoring designs and approaches (e.g., Kaufmann et al. 1999; Larsen et al. 2004), few efforts have evaluated how the temporal variability of stream habitat differs across sites. It is surprising that there have been relatively few assessments of the temporal variability of stream habitat, particularly given our understanding of the dynamic nature of stream networks (Giberson and Caissie 1998; Woodsmith et al. 2005). Yearly changes in habitat structure (i.e., temporal variability) can substantially affect our ability to accurately and precisely detect relevant changes in habitat status over time (Larsen et al. 2001). Thus, interactions between climate and geomorphic context suggest that even spatially proximate sites experiencing similar climate patterns may differ in their amounts of temporal variability due to inherent geomorphic characteristics (e.g., gradient).

The ability to differentiate temporal variability from other aspects of residual error, such as sampling error (e.g., Larsen et al. 2004), will have important implications for sampling design at any given site (Gibbs et al. 1998). Therefore, understanding which landscape attributes, climatic patterns, and disturbances influence the temporal variability of habitat attributes across sites is an important step in identifying our expectations of monitoring (e.g., the power to detect change). The overarching goals of the present study were to quantify the levels of temporal variability across sites and to provide insight into the processes that cause streams to change over the short time scales associated with most monitoring efforts (i.e., 1–10 years; Marsh and Trenham 2008). To achieve these goals, we used a 9-year, spatially explicit data set to accomplish the following objectives. First, we estimated the total variance at sites and decomposed this variance into site, year, and residual components to better understand the relative importance of each component. Next, we quantified the temporal variability of different stream habitat attributes observed across sites to provide insight into how much this component varies. Finally, we evaluated which site-specific landscape, climate, and disturbance attributes were associated with higher levels of temporal variability in stream habitat across sites as a means to better understand how these factors may influence year-to-year changes in stream habitat.

METHODS

Study Site and Sampling Design

As part of a larger overall stream habitat monitoring project (see Kershner et al. 2004a for specific details), we collected yearly habitat data (2001–2009) at 47sites within the interior Columbia River basin to evaluate temporal patterns in physical stream habitat (Figure 1). The sites were located in watersheds that were randomly selected from a set of watersheds spatially balanced across the landscape; this larger set of watersheds had also been randomly sampled from our study area (see Al-Chokhachy et al. 2010 for specific study design). Within each watershed, we selected the lowermost low-gradient site (gradient <3% based on visual observation) occurring on federally managed land, where the catchments upstream of each site were primarily (>50%) under federal management (Bureau of Land Management [BLM] or U.S. Forest Service [USFS]). As a result of these criteria, the size of the watershed upstream of each site (i.e., catchment) varied considerably across sites (range = 10.1–110.0 km^2). We focused our efforts on low-gradient sites as these areas are thought to be more sensitive to change under variable sediment and climate regimes (Montgomery and MacDonald 2002).

Field Sampling

Each year, we sampled sites during base flow conditions between June and September. To minimize temporal variability associated with sampling date, we generally (80% of visits) sampled sites within the same 2-week window each year. At each site, we collected reach-level (Frissell et al. 1986) habitat data by using methods implemented by the PacFish–InFish Biological Opinion Effectiveness Monitoring Program (Kershner et al. 2004a), and site lengths were determined as 20 times the bank-full width.

In addition to our yearly sampling events, we also conducted within-year sampling at a subset of our sites to obtain measures of sampling error. We randomly selected 17 of our sites to be resampled within the same year in 2003, and we randomly

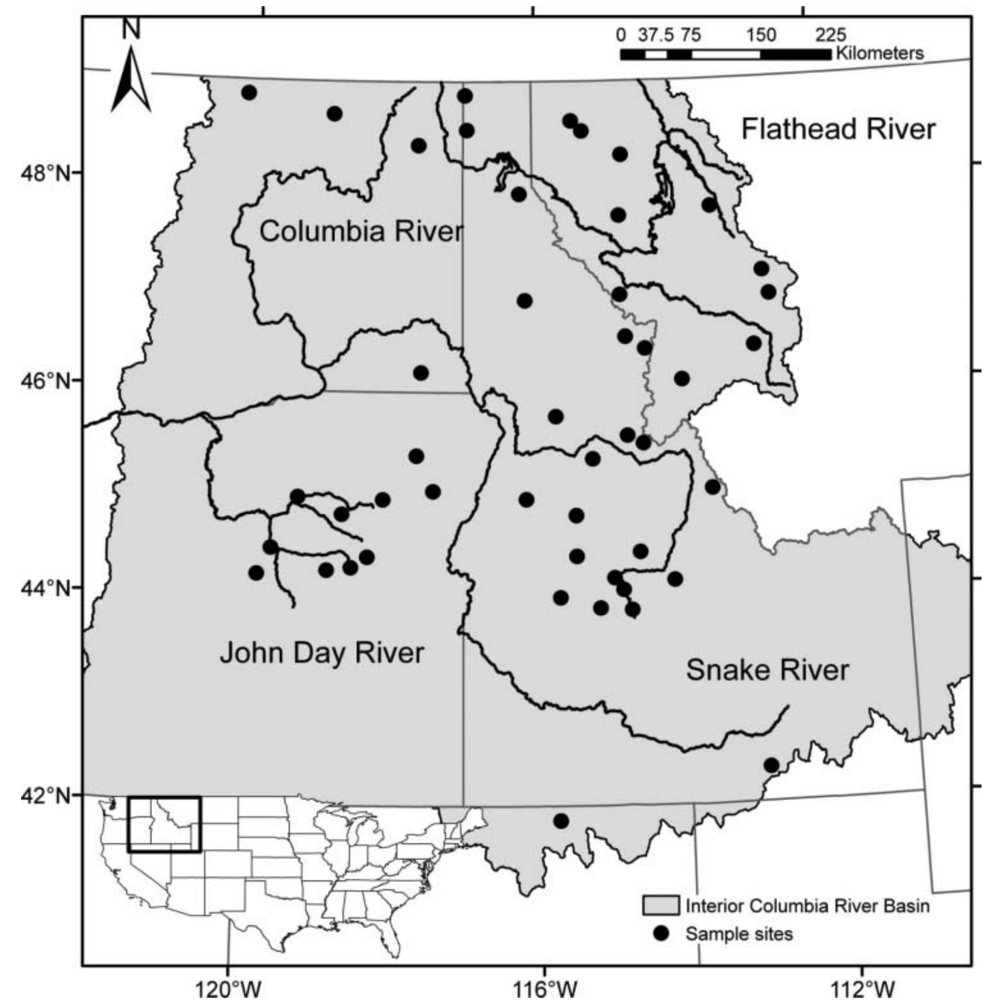

FIGURE 1. Map of the 47 sample sites in the interior Columbia River basin (see Table A.1 for site names).

selected 17 sites again (with replacement) in 2004. We also randomly selected the time intervals between initial visits and repeat-sample visits (median difference between sampling visits = 6d; range = 1–54d).

During each sampling occasion, we evaluated stream physical habitat attributes that are important for coldwater fishes of the Pacific Northwest. At each site, we estimated (1) median particle size (i.e., d_{50}) of all substrate particles measured at systematic transects throughout the site; (2) percent fine sediment in pool tails, which we measured by using a grid sampling approach within each pool tail and then averaged across all pools; (3) average streambank angle, determined from all bank angle measurements at transects; (4) the total number of pieces of large woody debris (LWD) exceeding 10 cm in diameter and 1m in length; (5) the percentage of pool habitat at each site (percent pool); and (6) residual pool depth, a measure of pool volume estimated as the maximum pool depth minus the pool tail depth and averaged across all pools (see Al-Chokhachy et al. 2010 for more specific sampling methods). In addition to these habitat attributes, we also estimated bank-full width and channel gradient to use as covariates in our analyses, which allowed us to control for inherent differences in stream power across sites (Knighton 1998; Kershner et al. 2004b). We measured bank-full width at each transect and calculated an average over the whole reach; the gradient of each site was measured by using a site level and stadia rod.

Landscape Attributes

Climatic patterns (Jorgensen et al. 2009), landscape attributes (Kershner et al. 2004b), and factors affecting stream power (Benda et al. 2005) can have profound impacts on the morphology and quality of stream attributes at a given site. Since stream attributes may be affected by factors at a variety of landscape scales (Frissell et al. 1986; Burnett et al. 2006; Feist et al. 2010), we considered landscape attributes at three spatial scales: (1) the entire catchment; (2) a 90-m buffer on each side of all streams within the catchment (1:24000-scale National Hydrography Dataset, U.S. Geological Survey [nhd.usgs.gov];

hereafter, "buffer scale"); and (3) a 90-m buffer on each side of all streams 1km upstream from the lower end of each site (hereafter, "segment scale").

In our analyses, we initially delineated each scale by using digital elevation models with ArcGIS version 9.2 (ESRI 2008), and we considered differences in the soil characteristics, topography, land cover, and factors affecting the stream power at each site. We quantified differences in soil characteristics by computing a continuous measure of the uniaxial compressive strength of each lithology type (hereafter, "erosivity"; Cao et al. 2007), where increasing values of erosivity positively correspond to the strength of the underlying lithology. At the catchment scale, we included characteristics related to stream power, including catchment area, SD of watershed slope, and drainage density. At the catchment, buffer, and segment scales, we also quantified the average slope.

Two measures of forested vegetation were quantified at each scale (LANDFIRE 2008). Specifically, we quantified the percentage of land covered in forested habitat (i.e., tree-dominated vegetation) and also the percentage of canopy cover as a measure of the extent of woody material in the canopy.

Precipitation is one of the major factors shaping stream morphology and changes in stream habitat (Jorgensen et al. 2009). To account for differences in precipitation across our sites, the weighted average (based on area) of all precipitation grids (30-year-average precipitation values, 1971–2000; PRISM [Parameter-Elevation Regressions on Independent Slopes Model] Climate Group, Oregon State University [www.prismclimate.org]) that overlaid each catchment was calculated to obtain an estimate of catchment-specific average precipitation. In addition to the 30-year average precipitation, we also calculated the coefficient of variation (CV; 100 × [SD/mean]) of annual precipitation for each catchment as a measure of the amount of temporal variability in precipitation events. Here, we considered annual precipitation as the sum of the monthly PRISM precipitation data for September–June because this period occurred outside the range of summer sample dates across all years and coincided with the time of year at which most precipitation occurs in our study area.

Next, information regarding land use and disturbance within each scale was quantified due to the impacts of these activities on stream channel and riparian processes (Trombulak and Frissell 2000; Luce 2002; Bakker and Moore 2007). We compiled grazing allotment boundaries from the USFS and BLM units within our study area and calculated the percentage of each scale that contained a grazing allotment. We also calculated the density of roads at each scale (km/km^2) from the USFS Geodata Clearinghouse (1:24000 scale; svinetfc4.fs.fed.us/clearinghouse/index.html).

Finally, we included recent wildfires as a natural disturbance in our analyses. Large fires can significantly alter and reduce vegetative cover, result in substantial inputs of sediment to stream networks, change streamflow dynamics, and cause large inputs of LWD (Minshall et al. 1997). Despite these potential impacts of fire, the timing of fire effects on stream networks and whether such effects occur at all are highly variable (Roper et al. 2007); therefore, in our analyses we simply quantified the percentage of each scale that experienced wildfire during each year from 1995to 2008 (LANDFIRE 2008).

Analyses

Variance decomposition and within-site temporal variability. —We initially decomposed our variance in a manner similar to that described by Larsen et al. (2004). However, our model differed in that we did not include the interaction term in this model; as such, the model took the following form:

$$X_{ijl} = \mu + St_i + Y_j + R_{ijk}, \tag{1}$$

where X_{ijk} is the response for the kth visit at site i in year j, μ is the overall mean, St_i is the random effect of site i, Y_i is the random year effect, and R_{ijk} is the random effect from the residual variation of sampling site i on the kth visit during year j. We used this analysis to evaluate how much of the total variance could be attributed to site variation, year effects, and residual error, which includes desynchronous yearly variation or site × year interactions and sampling variability (i.e., both sampling error and within-year variability; see Table 1 of Larsen et al. 2004). We did not formally separate sampling variability from desynchronous yearly variation as we were unable to conduct within-year sampling at each site, particularly given our understanding of potential differences in sampling variability across sites (Roper et al. 2010).

Next, we estimated average measures of sampling variability via the methods outlined by Littell et al. (2005). Here, we used our data from within-year sampling events (2003 and 2004; n = 34 total), and our model for this analysis was as follows (see equation 1for definitions of variables):

$$X_{ijk} = \mu + St_i + \mathrm{R}_{ijk}, \tag{2}$$

As described earlier, the time between initial sampling and repeat sampling within years varied, and the personnel that collected field data varied across these within-year visits. As such, our estimates of sampling variability included potential sampling error and variability due to changes in habitat structure between the sampling visits within years (Larsen et al. 2004). We estimated sampling variability by using the MIXED procedure in the Statistical Analysis System (SAS Institute 2004).

Next, we used our yearly sampling events at each of the 47 sites (excluding the within-year repeat visits) to quantify the total temporal variability observed at each site over the course of this study; the total temporal variability includes year effects, desynchronous variation, and residual error (Table A.1).

We excluded all within-year repeat visits and used the initial visit at each site as our primary sampling occasion where multiple visits occurred within the same year. With this analysis,

TABLE 1. A list of all candidate attributes considered as factors associated with the temporal variability of stream habitat data; the mean and SD for each landscape, disturbance, and geomorphic attribute at each specific scale and for climate attributes at the 47 sites in the interior Columbia River basin study area are shown (CV = coefficient of variation)

Scale	Attribute	Mean	SD
Catchment	Area (km^2)	41.3	23.3
	Slope (%)	34.9	11.0
	Erosivity (unitless)	148.9	78.3
	SD of elevation	189.1	60.8
	Stream density (km of stream/km^2)	1.3	0.4
Buffer	Percent forested	81.1	19.4
	Canopy cover	58.7	19.7
	Slope (%)	30.9	10.0
	Percent burned	14.7	27.8
	Percent grazed	40.1	46.1
	Road density (km/km^2)	1.0	1.1
Segment	Percent forested	79.8	26.5
	Canopy cover	58.9	27.6
	Slope (%)	20.6	13.9
	Percent burned	15.4	35.9
	Percent grazed	41.0	48.4
	Road density (km/km^2)	1.8	2.5
Reach	Channel sinuosity (ratio)	1.39	0.36
	Bank-full width (m)	6.9	3.0
	Stream gradient (%)	1.1	0.7
Climate	Average precipitation (m)	0.93	0.3
	CV of precipitation	0.16	0.04

we estimated how much of the total variability was attributed to site-to-site variability and site was used as a random effect (equation 2; e.g., Larsen et al. 2004); the residuals were used to quantify site-specific estimates of temporal variability via measures of root mean square error (RMSE).

Factors influencing temporal variability among sites.—We integrated our landscape and disturbance data (Table 1) with the site-specific measures of variability to investigate which factors and scales corresponded with more or less variability at a given site. We used multiple linear regression (MLR) analyses with RMSE of temporal variability as the response variable and the scale-specific (i.e., catchment, buffer, and segment) landscape and disturbance attributes as the explanatory variables (Table 1). We conducted a separate MLR analysis for each response attribute by using ordinary least-squares methods (Neter et al. 1983; R Development Core Team 2004).

In addition to individual attributes, we were also interested in quantifying the overall variability in the physical habitat of stream reaches. We were specifically interested in evaluating the consistency with which all six of the habitat attributes at individual sites tended to have high or low temporal variation relative to that at other sites. To evaluate overall variability, each site was assigned a rank from 1to 47 (i.e., the total number of sites included in this study) for each attribute; thus, the site with the lowest RMSE for a particular attribute received a rank of 1, and the site with the highest RMSE for that attribute received a rank of 47. To determine the overall variability at each site, we simply summed the rank transformations for the six attributes; thus, the lowest possible score for a site would be 6, and the highest possible score would be 282.

Our analyses followed a multiple-step process in variable selection. Initially, we evaluated potential correlations among explanatory variables to avoid any issues of multicollinearity, and we removed one variable from any pair that exhibited high correlation ($r \geq 0.60$). Prior to MLR analyses in this step and subsequent steps, we initially performed correlation analyses between response variables and each candidate explanatory attribute to minimize the number of variables included in the MLR analyses and to avoid the use of automated (e.g., stepwise) variable selection procedures. Only those explanatory variables with at least moderate levels of correlation with response variables were included in the MLRs ($r > 0.40$; the maximum number of explanatory variables considered in any model was 9).

Next, we incorporated those explanatory variables selected from the correlation analyses into the MLR analyses to evaluate how these explanatory variables were associated with the temporal variability of stream habitat (i.e., RMSE). To avoid any potential effects on model structure arising from the order in which explanatory variables entered the MLR models (i.e., spatial scales, landscape factors, climate, and disturbance), we used

TABLE 2. Estimates of grand mean, sampling error (as root mean square error [RMSE]), and sampling error expressed as a percentage of the grand mean for percent fine sediment (<6 mm) in pool tails, residual pool depth, median particle size (d_{50}), percent pool habitat, large woody debris (LWD) frequency, and bank angle.

Estimate	Fine sediment (%)	Residual pool depth (cm)	d_{50} (mm)	Percent pool	LWD frequency (pieces/km)	Bank angle (°)
Grand mean	27.8	39.6	35.7	53.4	302.7	100.2
RMSE	4.8	1.6	8.0	5.8	45.0	6.5
Percent of grand mean	17.3	4.0	22.4	10.9	14.9	6.5

an all-subsets modeling approach (R Development Core Team 2004). We evaluated each model by using Akaike's information criterion (AIC) and selected the model with the lowest AIC value as the most plausible model (Burnham and Anderson 1998). The difference (ΔAIC) between the AIC value for each competing model and the most plausible model was calculated as a measure of model support. We also calculated the Akaike weight of each model, which provides a measure of the relative likelihood of a given model on a scale from 0 to 1 (Burnham and Anderson 1998). Although the models were ranked according to the lowest AIC score, we used model averaging for parameter estimates and SEs to maximize the information gained from a multimodel approach. Explanatory variables measured in percentages were arcsine–square root transformed; for all top models, we checked for violations of linearity and heteroscedasticity by using visual assessments of the residuals and we tested for multicollinearity by using the variance inflation factor (>10).

RESULTS

Difficulties in accessing sites due to wildfire and resource limitations during the early years of this project prevented us from obtaining a complete time series at each site; from 2001to 2009, there was a total of 273 sampling occasions at the 47 sites within our study area. The average number of sampling occasions at a given site was 5.8 (range = 4–8; Table 1), and 42 of the 47 sites had at least 5 years of data.

Sampling Error, Variance Decomposition, and Within-Site Temporal Variability

Sampling error.—Estimates of potential sampling error varied considerably among the six measured attributes (Table 2). Residual pool depth (RMSE = 1.6 cm) and bank angle (RMSE = 6.5°) exhibited the lowest sampling error relative to overall mean estimates among all sites, while d_{50} (RMSE = 8 mm) and percent fine sediment (RMSE = 4.8%) had the highest sampling variability.

Variance decomposition.—Among-site variability exceeded 75% (SD = 6.7%) of total variance for all reach-scale variables (Figure 2). The greatest among-site variability was observed for the frequency of LWD (93.6% of total variance), and the lowest among-site variability was observed for percent pool habitat (75.3% of total variance). Overall, we found that year effects constituted a small portion of the total variance (average = 0.5% of total variance; SD = 0.5%). The highest variability attributed to year effects was found for percent pool (1.3% of total variance), while residual pool depth exhibited no detectable year effect; the relatively small amount of variance attributed to year effects suggests that our measures of total temporal variability largely represent desynchronous variation and potential sampling variability. The relative amount of the total variance as residuals (i.e., RMSE as a measure of temporal variability) varied considerably across attributes (average = 16.4% of total variance; SD = 6.4%); the highest amount of temporal variability was observed for percent pool habitat (23.3% of total variance), and the lowest was observed for frequency of LWD (6.0% of total variance).

We observed a considerable range of temporal variability in physical stream attributes across sites (Figure 3). Across attributes, the greatest differences in temporal variability among sites were identified for LWD frequency (CV of RMSE = 83.3%), d_{50} (CV of RMSE = 75.5%), and percent fine sediment in pool tails (CV of RMSE = 73.9%); we observed considerably lower among-site variation for percent pool habitat (CV of RMSE = 38.7%), residual pool depth (CV of RMSE = 31.1%),

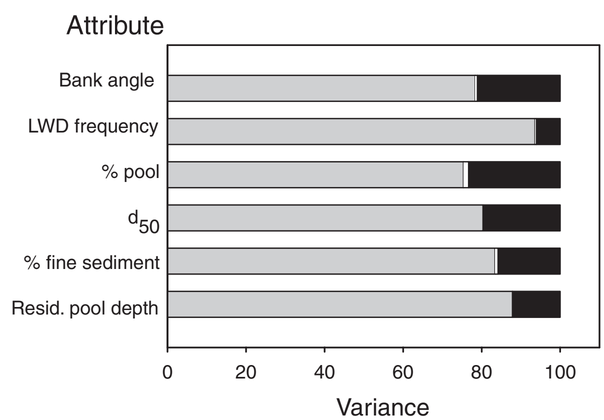

FIGURE 2. Percentage of variance attributed to differences across sites (gray shading), year effects (white), and residual variability (random temporal variability, sampling error, and random error; black shading) for bank angle, frequency of large woody debris (LWD), percent pool habitat, median particle size (d_{50}) of surface substrate, percentage of fine sediment (<6mm) in pool tails, and residual pool depth at 47 PacFish–InFish Biological Opinion monitoring sites in the interior Columbia River basin.

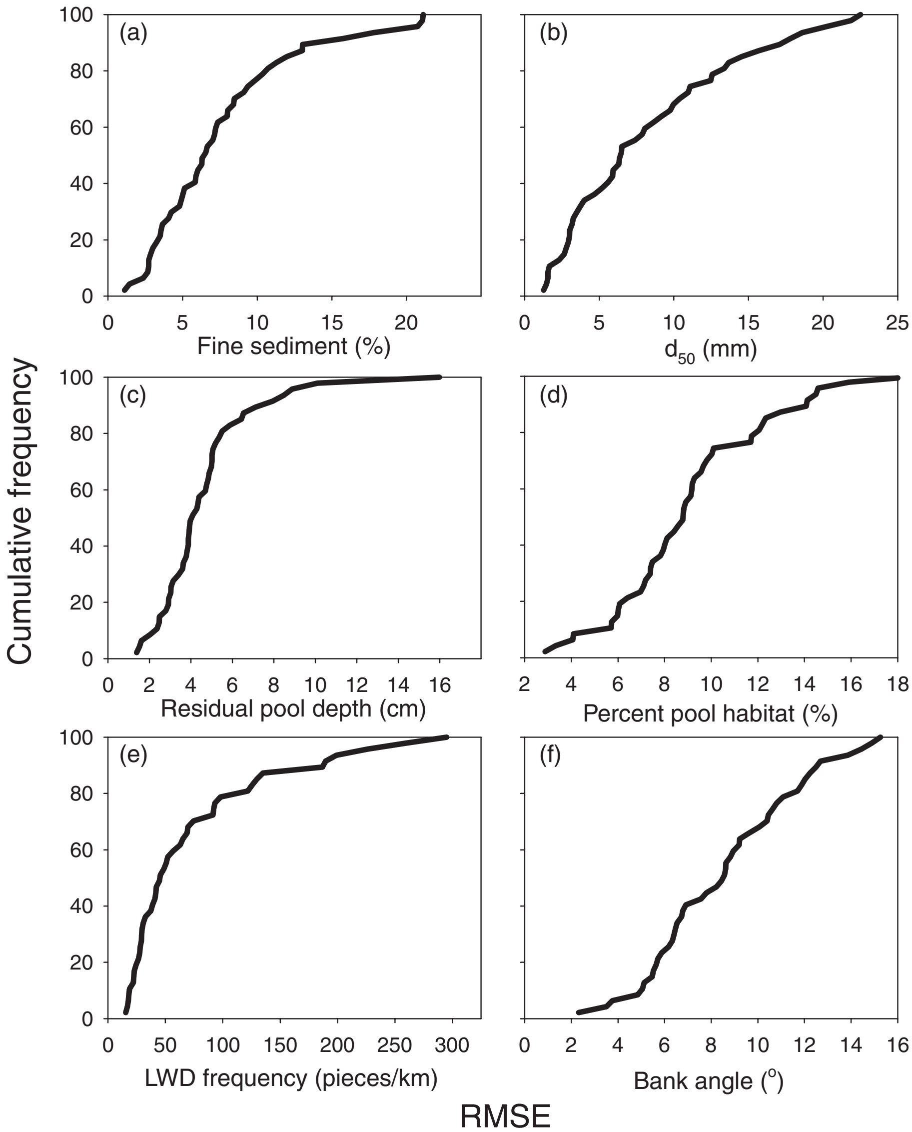

FIGURE 3. Cumulative frequency distributions of temporal variability (root mean square error [RMSE]) for **(a)** percentage of fine sediment (<6mm) in pool tails, **(b)** median particle size (d_{50}) of substrate, **(c)** residual pool depth, **(d)** percent pool habitat, **(e)** frequency of large woody debris (LWD), and **(f)** bank angle across 47 sites in the interior Columbia River basin.

and average bank angle (CV of RMSE = 19.4%). When considering the overall temporal variability (i.e., sum of the rank transformations of all habitat attributes at each site), we found an almost fourfold difference for the sites considered in these analyses (range = 68–235; Figure 4), and the average score for total variability at a site was 150 (SD = 38.1).

Factors Influencing Temporal Variability among Sites

Preliminary diagnostics indicated that landscape attributes and disturbance measures were highly correlated ($r > 0.60$) at the catchment and buffer scales. As such, we restricted our analyses to the stream buffer scale because of the importance of riparian areas for influencing stream conditions (e.g., Naiman et al. 2005) and because of recent management goals for riparian areas (e.g., Young 2000).

Our correlation analyses ($r > 0.40$) between response and explanatory variables substantially reduced the number of candidate explanatory variables considered in our MLRs. The following explanatory variables were included in our model selection approach. For the RMSE of percent fine sediment as the response variable, we evaluated bank-full width, gradient, and slope at the segment scale. For the RMSE of d_{50}, the explanatory variables considered were bank-full width, gradient, catchment area, percent burned at the buffer scale, and percent burned by

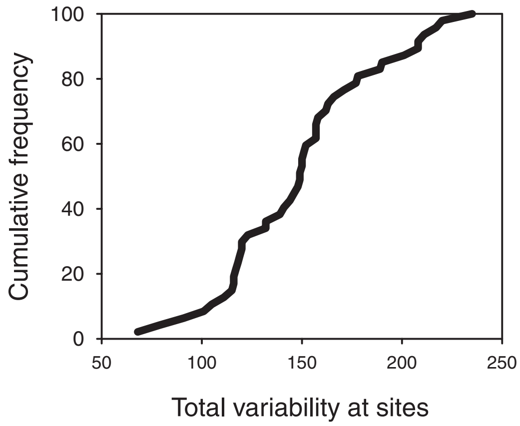

FIGURE 4. Cumulative frequency distribution of total temporal variability across 47 sites in the interior Columbia River basin; total temporal variability was calculated as the summed ranks of the temporal variability for percentage of fine sediment in pool tails, median particle size of substrate, residual pool depth, percent pool habitat, frequency of large woody debris, and bank angle at each site (see Methods).

wildfire at the segment scale. For the RMSE of residual pool depth, we examined bank-full width, gradient, average precipitation, percent forested at the segment scale, percent canopy at the segment scale, percent grazed at the segment scale, and percent grazed at the buffer scale. For the RMSE of percent pool, the catchment slope and erosivity were evaluated as explanatory variables. For the RMSE of LWD frequency, we examined bank-full width, erosivity, percent forested at the segment scale, percent canopy at the segment and buffer scales, percent burned at the buffer scale, and percent grazed at the segment and buffer scales. For the RMSE of bank angle, we considered bank-full width, road density at the buffer scale, and road density at the segment scale.

Percent fine sediment in pool tails.—Using the explanatory attributes considered here, we were able to explain little of the variability in RMSE of percent fine sediment across sites (adjusted $R^2 = 0.06$). Overall, we found that landscape- and reach-level attributes best explained the among-site temporal variability in fine sediment in pool tails. There was little discrimination between the top-two models (ΔAIC $= 0.70$), which had considerably higher Akaike weights than the other models (Table 3). The top model included only stream gradient, and the second-best model included only bank-full width; we also found some support for a model containing slope at the segment scale and a model containing both gradient and bank-full width. Model-averaged parameter estimates (Table 4) indicated that each of the explanatory variables was negatively associated with the RMSE of fine sediment, suggesting lower amounts of temporal variability in higher-gradient streams, streams with greater bank-full widths, and streams with steeper slopes in the segment.

Median particle size.—Our top model for d_{50} included bank-full width, gradient, and a measure of wildfire and had an Akaike weight of 0.84 (Table 3). Parameter estimates for the top model suggested higher levels of temporal variability in d_{50} for streams with larger bank-full widths, steeper gradients, and higher amounts of wildfire within the stream segment. All three attributes explained a significant amount of the variability in the RMSE of d_{50} (adjusted $R^2 = 0.47$; Table 4).

Residual pool depth.—Differences in the RMSE of residual pool depth across sites were best described by reach-level attributes, amount of canopy cover at the segment scale, and grazing (Table 3). The top model contained only bank-full width and gradient, and the Akaike weight for this model was 0.64. Parameter estimates for the top model indicated that low-gradient sites with larger bank-full widths had higher levels of temporal variability, and both of these parameters explained a significant amount of the variability in RMSE of residual pool depth across sites (adjusted $R^2 = 0.46$; Table 4). We found moderate support for the second model (ΔAIC $= 2.5$; Akaike weight $= 0.19$), which contained percent canopy at the segment scale in addition to gradient and bank-full width; however, there was considerable variability in the parameter estimate for percent canopy at the segment scale (model-averaged parameter estimate $= 0.13$; SE $= 0.08$). We found limited support for our other competing models (ΔAIC ≥ 3.9; Akaike weights < 0.10), which included grazing at the segment scale and average precipitation, and we found the parameter estimates for these variables to be moderate to highly variable (Table 4).

Percent pool habitat.—Only two attributes illustrated moderate correlations with the RMSE of percent pool habitat. The top model, which had an Akaike weight of 0.89, clearly outperformed the other candidate models (ΔAIC ≥ 4.7) and indicated significantly less temporal variability in percent pool habitat for sites in steeper catchments (Table 4). However, we were able to explain very little of the variability in RMSE of percent pool across our sites (adjusted $R^2 = 0.06$).

Large woody debris frequency.—We found little discrimination between the top models describing the differences in the RMSE of LWD frequency across sites (Table 3). The top model included only percent canopy at the segment scale, and this model accounted for a moderate amount of the variability in RMSE of LWD frequency across sites (adjusted $R^2 = 0.30$). Overall, models that included percent canopy at the segment scale had a combined Akaike weight of 0.89 (for these models, the Akaike weight ranged from 0.04 to 0.28), and our results indicated a positive relationship between percent canopy at this scale and the temporal variability in LWD (model-averaged parameter estimate $= 0.50$; SE $= 0.15$; Table 4). Other competing models included measures of wildfire, grazing, and bank-full width; these competing models indicated that LWD temporal variability had a positive relationship with bank-full width and wildfire and a negative relationship with the percentage of stream buffer grazed (Table 4).

Bank angle.—Our results pointed to a single model as being the best for describing the temporal variability of bank angle (Akaike weight $= 0.72$). This model only included the density

TABLE 3. Model selection results (ΔAIC = difference in Akaike's information criterion; with Akaike weights ≥ 0.02) from separate multiple linear regression (MLR) analyses describing the influence of landscape, climate, and disturbance attributes on the amount of temporal variability (root mean square error [RMSE]) for six stream habitat attributes (defined in Table 2) observed at 47 sites.

Attribute	Model[b]	ΔAIC	Akaike weight
RMSE, fine sediment[a]	Gradient	0.0	0.47
	BF	0.7	0.34
	Slope$_s$	2.9	0.11
	BF + gradient	3.6	0.08
RMSE d_{50}	BF + gradient + burned$_s$	0.0	0.84
	BF + area + gradient + burned$_s$	5.0	0.07
	BF + gradient	5.0	0.07
	BF + gradient + burned$_s$	7.9	0.02
RMSE, residual pool depth[a]	BF + gradient	0.0	0.64
	BF + gradient + canopy$_s$	2.5	0.19
	BF + gradient + grazed$_s$	3.9	0.09
	BF + gradient + precip	4.7	0.06
RMSE, percent pool	Catchment slope	0.0	0.89
	Erosiv	4.7	0.08
	Erosiv + catchment slope	7.1	0.03
RMSE, LWD frequency[a]	Canopy$_s$	0.0	0.28
	Canopy$_s$ + burned$_b$	0.2	0.25
	Canopy$_s$ + grazed$_b$	1.3	0.15
	BF + canopy$_s$	2.1	0.10
	Grazed$_b$ + burned$_b$	3.1	0.06
	Canopy$_s$ + grazed$_b$ + burned$_b$	3.1	0.06
	BF + canopy$_s$ + burned$_b$	4.1	0.04
	Grazed$_b$	4.2	0.03
	BF	5.0	0.02
RMSE, bank angle	Roads$_b$	0.0	0.72
	BF + roads$_b$	3.1	0.15
	BF	4.7	0.07
	Roads$_s$	5.2	0.05

[a]Log transformed to meet the normality assumptions of MLR.

[b]BF = bank-full width; area = catchment area; slope$_s$ = slope at the segment scale; precip = average precipitation; erosiv = erosivity; grazed$_s$ = percentage of segment scale that is grazed; grazed$_b$ = percentage of buffer scale that is grazed; burned$_s$ = percentage of segment scale that has experienced wildfire; burned$_b$ = percentage of buffer scale that has experienced wildfire; canopy$_s$ = forest canopy at the segment scale; grazed$_s$ = grazing at the segment scale; grazed$_b$ = grazing at the buffer scale; roads$_s$ = density of roads at the segment scale; roads$_b$ = density of roads at the buffer scale.

of roads at the buffer scale (Table 3) and accounted for a relatively low amount of variability in the RMSE of bank angle across sites (adjusted $R^2 = 0.11$). Our results indicated a negative relationship between the temporal variability in streambank angle and the density of roads at the buffer scale (model-averaged parameter estimate = −0.99; SE = 0.39). Competing models also included bank-full width; however, we found considerable variability in the parameter estimates for the variables in our models (Table 4).

Overall variability among sites.—Our results suggested that one model clearly outperformed other candidate models (ΔAIC ≥ 4.3; Table 5);this model included bank-full width, erosivity, percent canopy at the segment scale, density of roads at the buffer scale, and extent of grazing at the buffer scale. Competing models indicated simpler model structures. Across models, we found high variability in parameter estimates, and only erosivity illustrated a significant positive association with the overall variability among our sites (Table 5).

DISCUSSION

Our results from stream habitat data collected at 47 sites from 2001 to 2009 indicated extensive variation in the temporal variability across sites. Through our analyses, we were able to explain a substantial amount of the temporal variability in stream habitat for three of the six attributes considered herein. Given the relatively short time period evaluated in this project (e.g., Frissell et al. 1986), our results indicate that stream habitat conditions can vary considerably over short time periods. These large differences in habitat characteristics among years suggest that simple, one-visit assessments of habitat status may

TABLE 4. Model-averaged parameter estimates, SEs, and adjusted R^2 (for the top model) for competing multiple linear regression models explaining temporal variability [RMSE] in large woody debris (LWD) frequency, median particle size (d_{50}), bank angle, percent pool percent fine sediment in pool tails, and residual pool depth in 47 streams (2001–2009). Symbols for parameters are defined in Table 3.

Attribute	Parameter	Estimate	SE	Adjusted R^2
RMSE, fine sediment	BF	−0.03	0.01	0.06
	Gradient	−0.12	0.06	
	$Slope_s$	−0.008	0.003	
RMSE, d_{50}	BF	0.64	0.22	0.47
	Area	0.05	0.03	
	Gradient	3.25	0.95	
	$Burned_s$	3.43	1.20	
RMSE, residual pool depth	BF	0.039	0.01	0.46
	Gradient	−0.14	0.03	
	Precip	−0.009	0.10	
	$Canopy_s$	0.13	0.08	
	$Grazed_s$	−0.58	0.03	
RMSE, percent pool	Catchment slope	−0.09	0.04	0.06
	Erosiv	0.01	0.006	
RMSE, LWD frequency	BF	0.04	0.02	0.30
	$Grazed_b$	−0.18	0.08	
	$Burned_b$	0.22	0.10	
	$Canopy_s$	0.50	0.15	
RMSE, bank angle	$Roads_b$	−0.99	0.39	0.11
	BF	0.18	0.18	

be problematic for describing any given site (Al-Chokhachy and Roper 2010). Ultimately, explicit consideration of the geomorphic, landscape, and disturbance contexts for any particular site will help managers and restoration practitioners to consider appropriate monitoring designs (e.g., allocation of effort) and analytical methods (e.g., stratification) and to define their expectations for detecting trends (or changes) in stream habitat.

Sources of Variability and Implications for Monitoring

The inherent differences in stream habitat characteristics at sites indicate that repeat visits to the same sites constitute the most effective strategy for monitoring the status and trends of habitat attributes (Larsen et al. 2004). When site-to-site differences are removed, robust estimates of changes in the structure of stream habitat can largely be affected by sampling error and

TABLE 5. Model selection results (Δ AIC = difference in Akaike's information criterion), model fit (for the top model), and model-averaged parameter estimates and SE from multiple linear regression analyses describing the influence of landscape, climate, and disturbance attributes on the overall temporal variability observed at 47 sites. Symbols for explanatory variables are defined in Table 3.

Model or parameter	ΔAIC	Akaike weight	Estimate	SE	Adjusted R^2
BF + erosiv + $canopy^s$ + $roads^b$ + $grazing^b$	0.0	0.75			
BF + erosiv + $canopy^s$ + $roads^b$	4.3	0.09			
BF + erosiv + $canopy^s$ + $grazing^b$	4.5	0.08			
BF + $canopy^s$	6.1	0.04			
$Canopy^s$ + $roads^b$	7.1	0.02			
$Canopy^s$ + $grazing^b$	7.8	0.02			
BF			3.6	2.1	0.27
Erosiv			0.12	0.06	
$Canopy_s$			16.3	18.9	
$Roads_b$			−6.0	5.1	
$Grazing_b$			−2.0	9.1	

desynchronous variation. Our measures of sampling error were comparable with those observed in previous efforts (Whitacre et al. 2007; Roper et al. 2010). Specifically, the sampling error estimates (RMSE) from our analyses and those of Whitacre et al. (2007) and Roper et al. (2010) were within 3% for percent fine sediment in pool tails, within 1cm for residual pool depth, within 6 mm for d_{50}, within 7% for percent pool habitat, and identical for LWD frequency (from Roper et al. 2010 only; not comparable with the LWD error estimate from Whitacre et al. 2007). Sampling error for bank angle was not reported from these previous efforts. The temporal variability observed at the majority of sites considered in the present study exceeded these consistent estimates of potential sampling error, suggesting that our ability to accurately detect changes in the status of habitat attributes will vary by site. Overall, these results clearly indicate that the number of years of sampling required to achieve any desired statistical power will also vary by site; understanding those factors associated with different levels of temporal variability, which we describe further below, should be considered prior to the implementation of monitoring efforts.

Landscape Attributes

Landscape attributes affecting stream power (i.e., bank-full width, stream gradient, catchment area, and catchment slope) were found to be important in describing the temporal variability of the majority of stream attributes (6 of 7attributes). For three of the attributes (LWD frequency, d_{50}, and residual pool depth), we found a positive relationship between increasing stream size (bank-full width) and temporal variability. For the first- through third-order streams considered here (bank-full width range = 1.9–13.3 m), our results indicate that the positive association with increasing stream width is probably attributable to increased mobilization and transport of substrate and LWD (Benda et al. 2005; Wohl and Jaeger 2009). Given the significant role of LWD in shaping stream habitat (Montgomery et al. 1995; Hassan et al. 2005), the increased temporal variability of residual pool depth and d_{50} are probably associated with the temporal changes in LWD frequency at a site, which is consistent with our models explaining the overall temporal variability at sites. We found an opposite relationship with the temporal variability of fine sediment as larger streams had less temporal variability than smaller streams. This pattern is somewhat surprising due to our results for LWD, particularly as fine sediment levels can be sensitive to changes in LWD at sites (Gomi et al. 2001). However, the amount of fine sediment within pools can be affected by a myriad of factors, including both high-flow (i.e., scour) and low-flow events (i.e., drought), channel morphology, and LWD structures (Lisle and Hilton 1992). These potential complex interactions may be partially responsible for the relatively low explanatory power in our analysis of fine sediment temporal variability.

Despite the relatively low-gradient streams considered in these analyses, we did find that stream gradient was significantly associated with the temporal variability of stream habitat attributes. Sites with steeper gradients exhibited more temporal variability in d_{50} but lower levels of temporal variability in residual pool depth and fine sediment in pool tails. The impact of gradient on stream power (Knighton 1998) would suggest that the size of transported sediment varied substantially due to temporal differences in flow (even the moderate flows that occurred during this study: see below; Gomi and Sidle 2003). Increased stream power at sites with higher gradients also explains the low temporal variability of fine sediment and residual pool depth since fines can be transported under most flow conditions, thus preventing fines from infilling pools at these steeper sites (Lisle and Hilton 1999). We included two measures of land cover in our analyses, and our results indicated a positive relationship between the temporal variability of LWD and the percentage of canopy cover but no apparent relationship with the percentage of forested area at the segment or buffer scale. Our results are consistent with those of other authors who have found that LWD sources are often from proximate riparian areas (May and Gresswell 2003; Burnett et al. 2006); our results also indicate that the increased presence of mature forests with high amounts of canopy cover can result in more temporally diverse stream networks. We acknowledge that our method for inventorying LWD (i.e., 1 m in length) includes relatively small LWD pieces that may be trivial in more precipitous regions (e.g., the Cascade Mountains and coastal mountains of the Pacific Northwest), and this may have led to the inclusion of forest canopy as opposed to the extent of forested vegetation (i.e., percent forested) in the models. However, in the small headwater streams of the interior Columbia River basin, which is relatively arid, these smaller pieces of LWD can provide key habitat structure for aquatic species (Fetherston et al. 1995).

The parental geology of catchments can have strong influences on sediment sources and morphology of stream channels (Knighton 1998). Our results indicate that streams in catchments with stronger parental geologies (e.g., granitic) can exhibit higher levels of temporal variability (overall variability). Soils dominated by less-erodible parental material tend to have low amounts of silt and clay, resulting in more-erodible banks (David et al. 2009); with little cohesive materials, the erosion of these banks is likely to be a substantial source of LWD recruitment and change in the structure of the stream channels (Burnett et al. 2006).

Disturbance Attributes

Land use and temporal variability.—The impacts of land use on stream habitat status (Schlosser 1991; Richards et al. 1996; Kershner et al. 2004b) and stream biota (Allan et al. 1997; Paller 2002; Kaufmann and Hughes 2006) have been well documented. Landscape alterations and use can affect vegetative cover, soil characteristics, sediment retention (Kondolf et al. 2002), and the frequency and magnitude of severe hydrologic events within watersheds (Jones et al. 2000; Tonina et al. 2008). The impacts of such changes to stream networks can vary substantially. Where upland and riparian inputs of LWD have been removed, patterns of channel simplification—both spatially and temporally—have been observed (Reeves et al. 1993; Burnett et al. 2007). However, land use in headwater mountainous systems can also exhibit substantially higher temporal variability due to the increase of large, channel-altering events (Tonina et al. 2008; Bisson et al. 2009).

In our analyses, we found only limited evidence indicating lower amounts of temporal variability at sites with increased land use. Specifically, we found lower temporal variability of LWD at sites with higher levels of grazing. This negative relationship is probably attributable to the substantial reduction in recruitment of woody vegetation in riparian areas that are under grazing pressure (Green and Kauffman 1995; Belsky et al. 1999). These results are troubling given our understanding of the importance of LWD for fishes in headwater streams of the Pacific Northwest (Bisson et al. 1988; Roni 2003; Muhlfeld and Marotz 2005). Further, the negative association with grazing suggests an overall reduction in the complexity of stream habitat, particularly given the importance of LWD as a geomorphic control (Montgomery et al. 1995).

The lack of any additional associations with land use in our data appeared to be due to the wide range of temporal variability at sites under different land use regimes. For example, when we considered the overall temporal variability at sites, both road density and grazing were included in the top model. The parameter estimates for both road density and grazing at the buffer scale (Table 5) indicated negative associations between increased land use and the overall temporal variability at sites, but we found considerable variability among our sites and these estimates were not statistically significant.A post hoc evaluation of the underlying data suggests that the sites with the lowest temporal variability had some of the highest measures of land use quantified in this study (Figure 5); these results are consistent with underlying theories indicating that land use activities have simplified stream processes (Schlosser 1991; McIntosh et al. 2000). However, we also found considerable variability in these

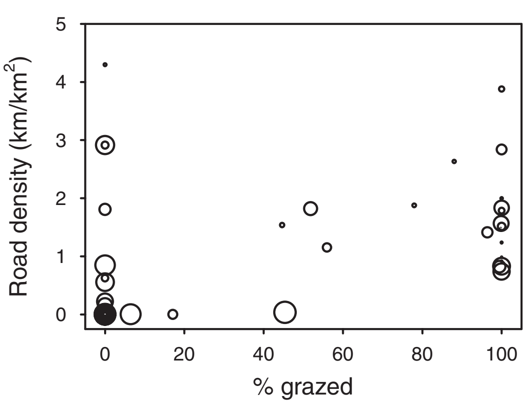

FIGURE 5. Bubble plot illustrating the total variability of all stream habitat attributes (diameter of the bubble is proportional to the total variability; see Figure 4 for a list of attributes) at 47 sites in the interior Columbia River basin with different levels of road density and grazing (buffer scale).

patterns. For example, 13 of our sites had no roads and no cattle grazing; of these 13 sites, one site exhibited the second-lowest measure of temporal variability and another had the second-highest measure of temporal variability among our sites. These results indicate that the temporal variability of stream habitat can vary substantially across sites and that multiple factors and interactions among factors are influencing the temporal variability of stream habitat (e.g., Feist et al. 2010). Although our limited sample size prevented formal evaluations of these complex interactions, future research is needed to generate a better understanding of how land use patterns impact the temporal variability of stream habitat across a variety of landscapes and geomorphic settings.

We also acknowledge that historic land use may have exerted a continuing effect on the patterns of temporal variability we observed at all of our sites (Harding et al. 1998; Foster et al. 2003; Allan 2004). Land management activities in the mid- to late-19th century and early 20th century had significant impacts on the landscapes and stream networks of the western United States (Platts and Nelson 1985). These historic alterations in the landscape may have resulted in substantial stream channel adjustments such that our assessments of the temporal variability associated with land management today are unable to detect the underlying relationships. In the last 20 years, extractive land management activities have been markedly reduced on federal land in our study area as a result of the listing of resident and anadromous salmonids under the Endangered Species Act in the 1990s (FEMAT 1993). While these actions appear to have impeded further degradation of federal lands in our study area (E.K.A., unpublished data), understanding the time frame over which watershed processes can be restored is challenging, particularly as the effects of current and historic land use may operate at multiple scales (Allan 2004).

Additionally, the lack of clear patterns between land use and the variability of stream habitat in our analyses may also be due to the available information regarding land use and the locations of our study sites. Specifically, our analyses did not include all possible land uses in forested landscapes (e.g., off-highway vehicles), which can substantially alter watersheds and the quality of stream habitat (Ouren et al. 2007); however, obtaining consistent information regarding nonroad disturbances, such as off-highway vehicle use, can be difficult. Furthermore, we recognize that the majority of lands upstream of our study sites were publicly owned. Given the presence of degraded landscapes in many private, urban, and agricultural riparian areas and their effects on stream habitat (Burnett et al. 2007), analyses that include these types of sites may uncover different relationships with the temporal variability of stream habitat.

Natural disturbances.—Within our study area, wildfire and floods represent two of the major natural disturbances influencing stream ecosystems (Rieman and Clayton 1997; Bisson et al. 2003; Benda et al. 2005). In our analyses, we found that the extent of wildfire was positively associated with the temporal variability of d_{50} and LWD frequency. These changes are expected as fires can remove vegetative cover and result in overland transport of sediment or debris flows (Benda et al. 2003; Istanbulluoglu et al. 2004). Additionally, wildfires can result in large pulses of LWD inputs into and through stream channels, and continued inputs of LWD to stream networks adjacent to fires can continue for decades (Gresswell 1999; Benda et al. 2003). When considered in combination with our results, the anticipated impacts of global climate change on fire frequency and magnitude in the Pacific Northwest (Westerling et al. 2006) suggest that stream habitats may become highly variable through time. We urge further research of the interactions between wildfires and landscape and climatic patterns to increase our understanding of how disturbance translates to observed stream conditions through time.

Despite the relative influence of hydrologic conditions on changes in physical habitat (e.g., Tonina et al. 2008), we found little evidence of a relationship between the variability in climate and the temporal variability of stream habitat in our analyses. The lack of clear patterns between temporal variability in precipitation and stream habitat may relate to the range of climate patterns observed over this period and the resolution of our measure of hydrologic conditions. In particular, the period of this study did not include any major climatic events: at only 6 of our 47 sites did annual precipitation in any given year exceed the 30-year averages. Furthermore, of the 320 annual precipitation records used to quantify CV at our sites, 73% did not exceed the 30-year averages. Additionally, our use of precipitation as a surrogate for hydrologic conditions at our sites may have reduced the clarity of patterns between variability in climate and stream habitat. However, our choice to examine precipitation instead of stream gage data was largely due to the substantial distances to fixed flow gages and the general inadequacies of hydrologic models for headwater tributary systems, which are parameterized by these distant flow gages (Liang et al. 1994). More accurate measures of annual flow regimes may soon be available (see Wenger et al. 2010), and the use of these measures may provide stronger relationships between the variability of seasonal and annual flow regimes and the temporal variability of physical stream habitat.

Limitations and Value of Long-Term Monitoring at Sites

We acknowledge that the temporal variability of stream habitat at sites was quantified through data collected over relatively short time scales (maximum = 8 years), particularly in the context of geomorphic change (e.g., Frissell et al. 1986). Longer time series of habitat data collected at sites may have a higher probability of observing change in response to stochastic climate and disturbance events, which may affect the range of temporal variability observed and the strength of documented associations and may uncover additional relationships. However, our results provide insight into the extent to which stream habitat can change over the relatively short time periods that are commonly used in habitat monitoring programs. Furthermore, our results also illustrate the importance of collecting long-term habitat data at fixed sites, and we urge further research and monitoring for more robust evaluations of the temporal patterns of stream habitat.

Conclusions

Few studies have evaluated the temporal variability of stream habitat across a variety of geomorphic settings and disturbance gradients in headwater systems. Our results from data collected over a 9-year period illustrated a wide array of temporal patterns at sites; our ability to identify specific relationships between landscape attributes, climate, and disturbance regimes varied by attribute. Across attributes, a considerable amount of the temporal variability went unexplained, indicating the complexity of factors affecting the temporal variability of stream habitat (e.g., Wang et al. 2006). Given the importance of the physical habitat in headwater systems for supporting sensitive aquatic biota (e.g., bull trout *Salvelinus confluentus*; Rieman et al. 1997), better insight into the factors affecting the temporal variability of physical habitat may improve our understanding of temporal dynamics of the populations of interest (Schlosser 1991). The variation in temporal variability across sites also suggests that "one-size-fits-all" approaches to monitoring physical habitat in response to restoration activities and land management may be misleading at any given site, particularly in terms of statistical power and the resources that are required to understand the effects of these actions. Although the sample sizes associated with large monitoring programs may be able to offset this temporal component of variance, in situ measures of temporal variability may be required for accurate assessments of statistical power for smaller-scale or individual-site monitoring programs.

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REFERENCES

Al-Chokhachy, R., and B. B. Roper. 2010. Quantifying the effects of sampling error in stream habitat data on the conservation and management of salmonids. Fisheries 35:476–488.

Al-Chokhachy, R., B. B. Roper, and E. Archer. 2010. Using a multimetric approach to evaluate the abiotic condition of streams in the upper Columbia and Missouri river basins. Transactions of the American Fisheries Society 139:1041–1059.

Allan, J. D. 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology and Systematics 35:257–284.

Allan, J. D., D. L. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. Freshwater Biology 37:149–161.

Anlauf, K. J., W. Gaeuman, and K. K. Jones. 2011. Detection of regional trends in salmonial habitat in coastal streams, Oregon. Transactions of the American Fisheries Society 140:52–66.

Bakker, J. D., and M. M. Moore. 2007. Controls on vegetation structure in southwestern ponderosa pine forests, 1941and 2004. Ecology 88:2305–2319.

Belsky, A. J., A. Matzke, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. Journal of Soil and Water Conservation 54:419–431.

Benda, L., M. A. Hassan, M. Church, and C. L. May. 2005. Geomorphology of steepland headwaters: the transition from hillslopes to channels. Journal of the American Water Resources Association 41:835–851.

Benda, L., D. Miller, P. Bigelow, and K. Andras. 2003. Effects of post-wildfire erosion on channel environments, Boise River, Idaho. Forest Ecology and Management 178:105–119.

Bernhardt, E. S., M. A. Palmer, J. D. Allan, G. Alexander, K. Barnas, S. Brooks, J. Carr, S. Clayton, C. Dahm, J. Follstad-Shah, D. Galat, S. Gloss, P. Goodwin, D. Hart, B. Hassett, R. Jenkinson, S. Katz, G. M. Kondolf, P. S. Lake, R. Lave, J. L. Meyer, T. K. O'Donnell, L. Pagano, B. Powell, and E. Sudduth. 2005. Synthesizing U.S. river restoration efforts. Science 308:636–637.

Bisson, P. A., J. B. Dunham, and G. H. Reeves. 2009. Freshwater ecosystems and resilience of Pacific salmon: habitat management based on natural variability. Ecology and Society 14:45.

Bisson, P. A., B. E. Rieman, C. Luce, P. F. Hessburg, D. C. Lee, J. L. Kershner, G. H. Reeves, and R. E. Gresswell. 2003. Fire and aquatic ecosystems of the western USA: current knowledge and key questions. Forest Ecology and Management 178:213–229.

Bisson, P. A., K. Sullivan, and J. L. Nielsen. 1988. Channel hydraulics, habitat use, and body form of juvenile coho salmon, steelhead, and cutthroat trout in streams. Transactions of the American Fisheries Society 117:262–273.

Burnett, K. M., G. H. Reeves, S. E. Clarke, and K. R. Christiansen. 2006. Comparing riparian and catchment influences on stream habitat in a forested, montane, landscape. Pages 175–197 in R. M. Hughes, L. Wang, and P. W. Seelbach, editors. Landscape influences on stream habitats and biological assemblages. American Fisheries Society, Symposium 48, Bethesda, Maryland.

Burnett, K. M., G. H. Reeves, D. J. Miller, S. Clarke, K. Vance-Borland, and K. Christiansen. 2007. Distribution of salmon-habitat potential relative to landscape characteristics and implications for conservation. Ecological Applications 17:66–80.

Burnham, K. P., and D. R. Anderson. 1998. Model selection and multimodel inference: a practical information-theoretic approach. Springer-Verlag, New York.

Cao, Y., C. P. Hawkins, J. Olson, and M. A. Kosterman. 2007. Modeling natural environmental gradients improves the accuracy and precision of diatom-based indicators. Journal of the North American Benthological Society 26:566–585.

David, G. C.L., B. P. Bledsoe, D. M. Merritt, and E. Wohl. 2009. The impacts of ski slope development on stream channel morphology in the White River National Forest, Colorado, USA. Geomorphology 103:375–388.

ESRI (Environmental Systems Research Institute). 2008. Getting to know ArcView GIS, 3rd edition. ESRI, Redlands, California.

Feist, B. E., E. A. Steel, D. W. Jensen, D. N. D. Sather. 2010. Does the scale of our observational window affect our conclusions about correlations between endangered salmon populations and their habitat? Landscape Ecology 25:727–743.

FEMAT (Forest Ecosystem Management Assessment Team). 1993. Forest ecosystem management: an ecological, economic, and social assessment. U.S. Forest Service, Bureau of Land Management, National Marine Fisheries Service, U.S. Fish and Wildlife Service, U.S. Park Service, and U.S. Environmental Protection Agency, Portland, Oregon.

Fetherston, K. L., R. J. Naiman, and R. E. Bilby. 1995. Large woody debris, physical process, and riparian forest development in montane river networks of the Pacific Northwest. Geomorphology 13:133–144.

Foster, D., F. Swanson, J. Aber, I. Burke, N. Brokaw, D. Tilman, and A. Knapp. 2003. The importance of land-use legacies to ecology and conservation. BioScience 53:77–88.

Frissell, C. A., W. J. Liss, C. E. Warren, and M. D. Hurley. 1986.A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environmental Management 10:199–214.

Gibbs, J. P., S. Droege, and P. Eagle. 1998. Monitoring populations of plants and animals. BioScience 48:935–940.

Giberson, D. J., and D. Caissie. 1998. Stream habitat hydraulics: interannual variability in three reaches of Catamaran Brook, New Brunswick. Canadian Journal of Fisheries and Aquatic Sciences 55:485–494.

Gomi, T., and R. C. Sidle. 2003. Bed load transport in managed steep-gradient headwater streams of southeastern Alaska. Water Resources Research 39:1336. DOI: 10.1029/2003WR002440.

Gomi, T., R. C. Sidle, M. D. Bryant, and R. D. Woodsmith. 2001. The characteristics of woody debris and sediment distribution in headwater streams, Southeastern Alaska. Canadian Journal of Forest Research 31:1386–1399.

Green, D. M., and J. B. Kauffman. 1995. Succession and livestock grazing in a northeast Oregon riparian ecosystem. Journal of Range Management 48:307–313.

Gresswell, R. E. 1999. Fire and aquatic ecosystems in forested biomes of North America. Transactions of the American Fisheries Society 128:193–221.

Harding, J. S., E. F. Benfield, P. V. Bolstad, G. S. Helfman, and E. B. D. Jones III. 1998. Stream biodiversity: the ghost of land use past of the, United States of America 95:14843–14847.

Hassan, M. A., D. L. Hogan, S. A. Bird, C. L. May, T. Gomi, and D. Campbell. 2005. Spatial and temporal dynamics of wood in headwater streams of the Pacific Northwest. Journal of the American Water Resources Association 41:899–919.

Istanbulluoglu, E., D. G. Tarboten, R. T. Pack, and C. H. Luce. 2004. Modeling of the interactions between forest vegetation, disturbances, and sediment yields. Journal of Geophysical Research 109:F01009. DOI: 10.1029/2003JF000041.

Jones, J. A., E. J. Swanson, B. C. Wemple, and K. U. Snyder. 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conservation Biology 14:76–85.

Jorgensen, J. C., J. M. Honea, M. M. McClure, T. D. Cooney, K. Engie, and D. M. Holzer. 2009. Linking landscape-level change to habitat quality: an evaluation of restoration actions on the freshwater habitat of spring-run Chinook salmon. Freshwater Biology 54:1560–1575.

Kaufmann, P. R., and R. M. Hughes. 2006. Geomorphic and anthropogenic influences on fish and amphibians in Pacific Northwest coastal streams. Pages 429–455 in R. M. Hughes, L. Wang, and P. W. Seelbach, editors. Landscape influences on stream habitats and biological assemblages. American Fisheries Society, Symposium 48, Bethesda, Maryland.

Kaufmann, P. R., P. Levine, E. G. Robison, C. Seeliger, and D. V. Peck. 1999. Quantifying physical habitat in wadeable streams. U.S. Environmental Protection Agency, EPA/620/R-99/003, Washington, D.C.

Kershner, J. L., E. K. Archer, M. Coles-Ritchie, E. R. Cowley, R. C. Henderson, K. Kratz, C. M. Quimby, D. L. Turner, L. C. Ulmer, and M. R. Vinson. 2004a. Guide to effective monitoring of aquatic and riparian resources. U.S. Forest Service General Technical Report RMRS-GTR-121.

Kershner, J. L., B. B. Roper, N. Bouwes, R. Henderson, and E. Archer. 2004b. An analysis of stream habitat conditions in reference and managed watersheds on some federal lands within the Columbia River basin. North American Journal of Fisheries Management 24:1363–1375.

Knighton, D. 1998. Fluvial forms and processes. Oxford University Press, New York.

Kondolf, G. M., H. Piegay, and N. Landon. 2002. Channel response to increased and decreased bedload supply from land use change: contrasts between two catchments. Geomorphology 45:35–51.

LANDFIRE (Landscape Fire and Resource Management Planning Tools). 2008. Landscape, fire, and resource management planning tools project. U.S. Geological Survey. Available: www.landfire.gov/products_national.php. (July 2010).

Larsen, D. P., P. R. Kaufmann, T. M. Kincaid, and N. S. Urquhart. 2004. Detecting persistent change in the habitat of salmon-bearing streams in the Pacific Northwest. Canadian Journal of Fisheries and Aquatic Sciences 61:283–291.

Larsen, D. P., T. M. Kincaid, S. E. Jacobs, and N. S. Urquhart. 2001. Designs for evaluating local and regional scale trends. BioScience 51:1069–1078.

Liang, X., D. P. Lettenmaier, E. F. Wood, and S. J. Burges. 1994.A simple hydrologically based model of land-surface water and energy fluxes for general-circulation models. Journal of Geophysical Research 99:14415–14428.

Lisle, T. E., and S. Hilton. 1999. Fine bed material in pools of natural gravel bed channels. Water Resources Research 35:1291–1304.

Lisle, T. E., and S. Hilton. 1992. The volume of fine sediment in pools: an index of sediment supply in gravel-bed streams. Water Resources Bulletin 28:371–383.

Littell, R. C., G. A. Milliken, W. W. Stroup, R. D. Wolfinger, and O. Schabenberger. 2005. SAS for mixed models, 2nd edition. SAS Institute, Cary, North Carolina.

Luce, C. H. 2002. Hydrological processes and pathways affected by forest roads: what do we still need to learn? Hydrological Processes 16:2901–2904.

Marsh, D. M., and P. C. Trenham. 2008. Current trends in plant and animal population monitoring. Conservation Biology 22:647–655.

May, C. L., and R. E. Gresswell. 2003. Processes and rates of sediment and wood accumulation in headwater streams of the Oregon Coast range, USA. Earth Surface Processes and Landforms 28:409–424.

McIntosh, B. A., J. R. Sedell, R. F. Thurow, S. E. Clarke, and G. L. Chandler. 2000. Historical changes in pool habitats in the Columbia River basin. Ecological Applications 10:1478–1496.

Minshall, G. W., R. C. Petersen, K. W. Cummins, T. L. Bott, J. R. Sedell, C. E. Cushing, and R. L. Vannote. 1983. Interbiome comparison of stream ecosystem dynamics. Ecological Monographs 53:1–25.

Minshall, G. W., C. T. Robinson, and D. E. Lawrence. 1997. Postfire responses of lotic ecosystems in Yellowstone National Park, USA. Canadian Journal of Fisheries and Aquatic Sciences 54:2509–2525.

Montgomery, D. R., J. M. Buffington, R. D. Smith, K. M. Schmidt, and G. Pess. 1995. Pool spacing in forest channels. Water Resources Research 31:1097–1105.

Montgomery, D. R., and L. H. MacDonald. 2002. Diagnostic approach to stream channel assessment and monitoring. Journal of the American Water Resources Association 38:1–16.

Muhlfeld, C. C., and B. Marotz. 2005. Seasonal movement and habitat use by subadult bull trout in the upper Flathead River system, Montana. North American Journal of Fisheries Management 25:797–810.

Naiman, R. J., H. Decamps, and M. E. McClain. 2005. Riparia: ecology, conservation, and management of streamside communities. Elsevier Academic, Burlington, Massachusetts.

Neter, J., W. Wasserman, and M. H. Kutner. 1983. Applied linear regression models. R. D. Irwin, Homewood, Illinois.

Nickelson, T. E., and P. W. Lawson. 1998. Population viability of coho salmon, *Oncorhynchus kisutch*, in Oregon coastal basins: applications of a habitat-based life cycle model. Canadian Journal of Fisheries and Aquatic Sciences 55:2383–2392.

Ouren, D. S., C. Haas, C. P. Melcher, S. C. Stewart, P. D. Ponds, N. R. Sexton, L. Burris, T. Fancher, and Z. H. Bowen. 2007. Environmental effects of off-highway vehicles on Bureau of Land Management lands: a literature synthesis, annotated bibliographies, extensive bibliographies, and internet resources: U.S. Geological Survey, Open-File Report 2007–1353.

Paller, M. H. 2002. Temporal variability in fish assemblages from disturbed and undisturbed streams. Journal of Aquatic Ecosystems Stress and Recovery 9:149–158.

Platts, W. S., and R. L. Nelson. 1985. Impacts of rest-rotation grazing on stream banks in forested watershed in Idaho. North American Journal of Fisheries Management 5:547–556.

R Development Core Team. 2004. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. Available: www.r-project.org. (July 2010).

Reeves, G. H., F. H. Everest, and J. R. Sedell. 1993. Diversity of juvenile anadromous salmonid assemblages in coastal Oregon basins with different levels of timber harvest. Transactions of the American Fisheries Society 122:309–317.

Richards, C., L. B. Johnson, and G. E. Host. 1996. Landscape influences on stream habitats and biota. Canadian Journal of Fisheries and Aquatic Sciences 53:295–311.

Rieman, B. E., and J. Clayton. 1997. Wildfire and native fish: issues of forest health and conservation of native fishes. Fisheries 22(11):6–15.

Rieman, B. E., D. C. Lee, and R. F. Thurow. 1997. Distribution, status, and likely future trends of bull trout within the Columbia River and Klamath basins. North American Journal of Fisheries Management 17:1111–1125.

Riley, S. C., and K. D. Fausch. 1995. Trout population response to habitat enhancement in six northern Colorado streams. Canadian Journal of Fisheries and Aquatic Sciences 52:34–53.

Roni, P. 2003. Responses of benthic fishes and giant salamanders to placement of large woody debris in small Pacific Northwest streams. North American Journal of Fisheries Management 23:1087–1097.

Roper, B. B., J. M. Buffington, S. Bennett, S. H. Lanigan, E. Archer, S. T. Downie, J. Faustini, T. W. Hillman, S. Hubler, K. Jones, C. Jordan, P. R. Kaufmann, G. Merritt, C. Moyer, and A. Pleus. 2010.A Comparison of the performance and compatibility of protocols used by seven monitoring groups to measure stream habitat in the Pacific Northwest. North American Journal of Fisheries Management 30:565–587.

Roper, B. B., B. Jarvis, and J. L. Kershner. 2007. The role of natural vegetative disturbance in determining reach characteristics in central Idaho and western Montana. Northwest Science 81:224–238.

SAS Institute. 2004. SAS OnlineDoc 9.1.3. SAS Institute, Cary, North Carolina.

Schlosser, I. J. 1991. Stream fish ecology: a landscape perspective. BioScience 41:704–712.

Southwood, T. R. E. 1977. Habitat, the template for ecological strategies? Journal of Animal Ecology 46:337–365.

Tonina, D., C. H. Luce, B. Rieman, J. M. Buffington, P. Goodwin, S. R. Clayton, S. M. Ali, J. J. Barry, and C. Berenbock. 2008. Hydrological response to timber harvest in northern Idaho: implications for channel scour and persistence of salmonids. Hydrological Processes 22:3223–3235.

Trombulak, S. C., and C. A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology 14:18–30.

Urquhart, N. S., S. G. Paulsen, and D. P. Larsen. 1998. Monitoring for policy-relevant trends over time. Ecological Applications 8:246–257.

Wang, L., P. W. Seelbach, and J. Lyons. 2006. Effects of levels of human disturbance on the influence of catchment, riparian, and reach-scale factors of fish assemblages. Pages 199–219 in R. M. Hughes, L. Wang, and P. W. Seelbach, editors. Landscape influence on stream habitats and biological assemblages. American Fisheries Society, Symposium 48, Bethesda, Maryand.

Wenger, S. J., C. H. Luce, A. F. Hamlet, D. J. Isaak, and H. Neville. 2010. Macroscale hydrologic modeling of ecologically relevant flow metrics. Water Resources Research 46. DOI: 10.1029/2009/WR008839.

Westerling, A. L., H. G. Hidalso, D. R. Cayan, and T. W. Swetnam. 2006. Warming and earlier spring increases western U.S. forest wildfire activity. Science 313:940–943.

Whitacre, H. W., B. B. Roper, and J. L. Kershner. 2007.A comparison of protocols and observer precision for measuring physical stream attributes. Journal of the American Water Resources Association 43:923–937.

Wohl, E., and K. Jaeger. 2009.A conceptual model for the longitudinal distribution of wood in mountains streams. Earth Surface Processes and Landforms 34:329–344.

Woodsmith, R. D., J. R. Noel, and M. L. Dilger. 2005. An approach to effectiveness monitoring floodplain channel aquatic habitat: channel condition assessment. Landscape and Urban Planning 72:177–204.

Young, K. A. 2000. Riparian zone management in the Pacific Northwest: who's cutting what? Environmental Management 26:131–144.

APPENDIX: STREAM HABITAT ATTRIBUTES AT STUDY SITES

TABLE A.1. Average yearly estimates (SD in parenthesis) of percent fine sediment (<6 mm) in pool tails, residual pool depth, median particle size (d_{50}), percent pool habitat, large woody debris (LWD) frequency, and bank angle and the number of years of sampling (n) for each of the 47 sites in the interior Columbia River basin.

Stream	Fine sediment (%)	Residual pool depth (cm)	d_{50} (mm)	Percent pool (%)	LWD frequency (pieces/km)	Bank angle (°)	n
Andrews Creek	27.1 (10.1)	42.8 (7.0)	70 (13)	37.7 (12.0)	924.5 (288.4)	104.8 (15.0)	4
Barron Creek	18.7 (9.7)	40.4 (3.8)	21 (5)	70.6 (8.5)	383.5 (89.9)	95.6 (7.2)	4
Bayley Creek	100.0 (0.0)	29.0 (6.3)	3 (1)	58.4 (15.0)	2.9 (3.3)	128.5 (7.2)	4
Bearskin Creek	80.9 (10.1)	70.7 (4.2)	6 (1)	84.5 (9.9)	19.4 (5.9)	97.7 (9.2)	5
Cottonwood Creek	13.4 (6.1)	16.8 (1.6)	52 (11)	20.5 (8.4)	71.6 (27.9)	139.4 (7.2)	5
Dutch Flat Creek	80 (13.0)	57.7 (10.8)	3 (3)	83.1 (12.1)	1,234.3 (319.3)	82.4 (10.6)	6
East Fork Canyon Creek	9.3 (1.4)	34.1 (4.9)	27 (7)	39.2 (10.3)	260.5 (103.7)	124.0 (9.0)	6
Eagle Creek	29.7 (11.4)	76.4 (17.4)	25 (10)	51.9 (7.1)	640.1 (156.1)	88.3 (11.3)	7
Eightmile Creek	24.8 (20.3)	40.5 (8.3)	55 (5)	25.3 (6.1)	188.6 (12.9)	97.8 (11.9)	4
Elk Creek, ID	86.2 (7.4)	50.8 (7.7)	4 (4)	77.4 (16.0)	311.9 (73.4)	66.0 (5.9)	5
Elk, Creek, OR	23.9 (4.3)	25.3 (3.9)	20 (5)	38.8 (8.2)	254.2 (88.3)	98.7 (9.2)	6
Emerson Creek	19.4 (8.5)	24.5 (4.1)	39 (9)	37.5 (7.4)	138.9 (13.5)	99.7 (10.1)	6
Goat Creek	40.0 (18.8)	47.3 (4.3)	9 (1)	52.5 (8.5)	45.2 (3.4)	75.0 (15.9)	7
Gorge Creek	9.3 (5.5)	59.0 (6.0)	40 (4)	57.3 (8.6)	734.9 (301.1)	80.7 (9.2)	7
Griffin Creek	19.0 (23.6)	36.5 (4.0)	34 (4)	71.4 (13.4)	274 (75.9)	87.0 (6.6)	6
Grimes Creek	26.0 (11.3)	50.0 (5.6)	35 (20)	77.3 (6.7)	15.3 (8.8)	93.0 (15.0)	5
Huckleberry Creek	18.5 (13.7)	30.4 (2.7)	26 (6)	37.6 (12.2)	105.8 (22.3)	93.4 (9.0)	8
Hughes Creek	23.9 (8.0)	107.9 (7.4)	14 (4)	80.8 (3.2)	212.7 (28.9)	77.9 (5.6)	7
John Day River	14.5 (6.1)	50.2 (5.0)	17 (4)	79.2 (9.0)	120.6 (22.5)	85.0 (6.7)	5
Kenney Creek	7.9 (2.8)	21.7 (4.0)	37 (8)	30.1 (13.5)	185.6 (55.7)	116.6 (15.3)	6
Lamb Creek	71.5 (21.2)	38.6 (4.2)	3 (1)	64.8 (12.0)	179.7 (48.9)	83.9 (10.2)	7
Little Goose Creek	16.6 (6.3)	24.5 (1.6)	49 (12)	52.5 (9.9)	144.8 (30.0)	100.7 (4.4)	6
Little Minam River	0.8 (0.7)	36.8 (10.0)	103 (24)	22.8 (13.6)	204.8 (70.2)	133.0 (6.2)	5
Little Queens Creek	12.1 (5.6)	35.6 (5.6)	55 (15)	41.7 (14.5)	184.6 (42.7)	96.1 (13.0)	6
Little Thompson Creek	6.1 (9.2)	19.2 (2.3)	75 (22)	35.5 (10.9)	187.7 (54.3)	114.0 (9.8)	6
Mallory Creek	12.4 (7.3)	18.7 (3.3)	55 (16)	32.6 (7.7)	21.9 (20.0)	143.2 (6.0)	6
Meadow Creek	9.0 (3.5)	39.5 (4.0)	37 (7)	53.6 (3.1)	234.2 (15.9)	103.7 (7.0)	6
Moose Creek	25.8 (10.3)	45.2 (5.2)	32 (14)	73.2 (8.0)	200.7 (57.6)	89.0 (4.3)	5
North Fork Fish Creek	4.7 (3.6)	34 (4.4)	76 (12)	29.2 (7.3)	265.6 (109.2)	117.8 (13.8)	8
Papoose Creek	11.2 (7.4)	39.8 (4.9)	53 (20)	37.5 (6.5)	495.9 (107.5)	110.6 (6.7)	5
Reynolds Creek	14.4 (6.5)	19.0 (3.3)	107 (66)	17.1 (8.4)	369.8 (62.6)	107.4 (7.7)	5
Road Creek	79.4 (21)	31.0 (3.3)	5 (2)	62.1 (18.3)	0.0 (0.0)	92.8 (6.6)	4
Rock Creek	9.7 (2.4)	60.0 (5.8)	46 (14)	72.3 (6.3)	179.0 (17.9)	97.1 (10.6)	8
South Fork Desolation Creek	21.8 (11.2)	47.7 (9.0)	31 (10)	71.9 (14.6)	857.5 (216.3)	91.9 (10.1)	6
Sand Creek	27.5 (9.1)	28.6 (5.8)	18 (6)	62.9 (7.6)	0.8 (2.1)	118.3 (9.4)	7
Sleeping Child Creek	15.2 (7.5)	42.6 (4.0)	58 (18)	49.9 (5.7)	733.9 (217.9)	112.7 (11.5)	6
Sublett Creek	64.3 (12.8)	21.7 (2.6)	2 (1)	41.2 (14.8)	5.9 (13.2)	129.3 (10.7)	6
Sugar Creek	34.4 (2.7)	29.2 (3.5)	12 (3)	71.5 (12.2)	181.2 (32.2)	83.2 (10.1)	5
Swet Creek	33.6 (7.4)	34.7 (3.7)	19 (11)	64.7 (8.9)	708.4 (93.2)	69.0 (12.4)	6
Trapper Creek	22.3 (10.4)	46.4 (5.9)	25 (5)	63.5 (10.3)	914.8 (86.0)	67.6 (14.2)	5
Tucannon River	8.3 (3.2)	46.5 (7.1)	55 (13)	51.8 (10.0)	502.3 (211.4)	125.1 (6.8)	7
Twentymile Creek	51.6 (5.0)	33.7 (4.2)	6 (4)	74.3 (18.8)	510.2 (166.7)	83.3 (8.5)	6
Upper Salmon River	8.6 (6.2)	32.7 (2.8)	60 (18)	47.6 (11.9)	0.0 (0.0)	121.3 (2.8)	7
West Fork Granite Creek	48.1 (6.6)	25.7 (5.2)	11 (5)	54.9 (9.7)	214.1 (79.6)	122.2 (10.7)	6
West Branch Big Creek	20.0 (7.1)	63.8 (9.5)	23 (7)	83.0 (12.9)	826.6 (145.2)	72.8 (12.0)	6
White Sands Creek	6.9 (0.9)	46.8 (5.6)	90 (23)	36.4 (6.9)	466.3 (147.1)	83.8 (9.4)	5
Willow Creek	20.0 (4.6)	19.6 (1.5)	29 (7)	28.2 (10.3)	196.6 (20.8)	109.0 (6.4)	5