APPENDIX A: ANTHROPOGENIC ALTERATIONS TO HABITAT-FORMING PROCESSES

Introduction

Our approach to assessing root causes of habitat degradation by directly analyzing land use changes to habitat-forming processes is based on two well-founded assumptions: (1) salmon are adapted to local habitat conditions, and (2) habitat conditions vary in space and time as a function of landscape processes (as such as soil erosion or riparian functions) (Beechie and Bolton 1999). In combination these two statements suggest that sustainable restoration of salmon habitat must focus on restoring processes that create and maintain ecosystems that support salmon. Process-based restoration actions will ultimately restore the range of habitat conditions that historically sustained abundant and diverse salmon populations, and sustain those conditions without repeated and costly intervention by land managers.

To analyze deviations from natural conditions for each process and identify where restoration actions are likely to benefit salmon, for each process we estimate (1) historical rates, (2) current rates, and (3) the change in rate (Beechie and Bolton 1999). In our analyses we use coarse resolution data and process-based assessment approaches to identify locations where habitat-forming processes have likely been significantly altered by land uses (Beechie *et al.* 2002, Beechie *et al.* 2003). We analyze a suite of candidate limiting factors (Table 1) that can be grouped into (1) processes that form and sustain aquatic habitats, and (2) habitat conditions created by those processes (Figure 1). Our analyses focus on habitat-forming processes rather than instream conditions because coarse resolution data sets available can be most directly related to landscape processes, whereas instream habitat conditions are more reliably analyzed using field inventories.

Where previous analyses (e.g., ICBEMP [Quigley and Abelbide1997] estimated historical and current rates or conditions of landscape processes, we summarize those results and avoid redundant analyses. Where previous efforts did not answer our specific questions, we focus on simple, process-based analyses that are easily related to land management practices that alter habitat-forming processes. To account for ecoregional variation in dominant processes (Table 2) we tailor analyses separately to the relatively dry interior Columbia River basin and the wetter lower Columbia River (Beechie *et al.* 2003). Where possible, we use available aerial photograph or field data to assess potential errors in coarse resolution analyses.

The Columbia River basin encompasses a wide range of physical and ecological conditions that are classified as a hierarchical set of ecoregions (CEC 1997) (Figure 2). Level II ecoregions identify three areas within which climate, lithology, topography, and ecosystems are generally similar: marine coastal forests, drier western forested mountains, and semi-arid to arid western deserts (CEC 1997, USEPA 2000). Level II ecoregions are largely determined by topography, precipitation, and vegetation patterns, as illustrated in Figure 2. Western deserts tend to be lower and drier than adjacent forested ecoregions, and wettest areas are found west of the Cascade Mountain crest.

Physical and climatic characteristics drive many of the landscape processes that form and sustain salmonid habitats in streams (Beechie and Bolton 1999). Therefore, our assessments rely heavily on several common data sets: digital elevation data (10m and 30m DEMs from USGS), vegetation and land cover classification (historical data from ICBEMP [Quigley and Arbelbide 1997], recent data from NHI and/or USGS) (Figure 3), and hydrography generated from DEMs (see Appendix C for complete description). In general, the same categories of assessments must be conducted regardless of ecoregion (e.g., sediment supply, riparian functions, isolated habitats), but the specific processes or mechanisms addressed may vary from one ecoregion to another. For example, sediment supply to the stream network should be evaluated in any watershed, but certain processes of sediment supply may be emphasized depending on location.

Analyses of each candidate habitat factor (listed in Table 1) produce maps of historical rates or conditions, current rates or conditions, and ratios of current rates or conditions to historical rates or conditions. The degree of change estimated from the analysis is our primary indicator of habitat change within each 6th field HUC, and its association with each of the populations identified by the TRT provides a simple indicator of where various classes of restoration actions are likely necessary to improve population performance.

Riparian functions

Overview

This analysis is conducted in two parts: riparian areas without significant floodplain areas (*Riparian Screen*), and riparian areas within floodplains (*Floodplain Screen*). The floodplain screen applies to streams falling within floodplain areas, as determined by FEMA floodplain maps, whereas the riparian screen applies only to streams not covered in the floodplain screen.

Each screen consists of 2 steps. Step 1 uses geospatial data in GIS to evaluate how land use conditions bordering streams have changed from historical conditions as a result of human influence, with the results summarized by 6th field HUC. Step 2 involves random spot-checking throughout the Basin using aerial photos to assess differences in riparian conditions in impaired and natural areas. In addition, we provide a summary of the literature on riparian functions at different buffer widths that should be useful when translating results of the screens into management decisions.

Products

1. A literature review of variation in riparian functions with increasing buffer width.
2. For each screen, Riparian and Floodplain, across the Columbia River Basin:
   - Map of percent of streams running through agricultural/urban land. Summarized as subwatersheds (6th field HUCs) having 0%, <25%, 25-50%, and >75% of streams or floodplains in agriculture/urban.
   - Summary data tables from aerial photo interpretation for each stratum (agriculture/urban, forest, shrub/grass), giving mean buffer widths, nearest vegetation type (type closest to the stream), and dominant vegetation type (dominant through 100m on each side), as well as some descriptive statistics (e.g., cumulative frequency distributions of buffer widths in each stratum, and tests for differences in buffer widths among strata).

Literature Review: Riparian Functions in the Columbia River Basin

Riparian areas provide many functions that contribute to habitat that is suitable for viability of salmonids, as well as the integrity of the stream network itself (Table 3). In order to build and interpret coarse screens, we felt that we needed to have a basic understanding of which riparian functions were likely to be important to salmon populations in the Columbia River Basin, and how such functions differed with buffer width.

Forested Areas

For forested areas west of the Cascades, bank stability (e.g., root strength), organic matter input (e.g., large woody debris and litter fall), and stream temperature control (e.g., shading) have been considered important riparian functions driving stream condition. FEMAT (1993) related these functions to buffer widths in terms of site potential tree heights, but this has also been represented in terms of buffer width (Beechie *et al.* 2003; Figure 4). A similar set of curves can be drawn for the same functions in eastern Washington, based on an adjustment in average site potential tree height between eastern and western Washington (Figure 5). This set of curves may be used for forested areas of the Interior Columbia, in the absence of local data.

The curve for large woody debris input, as the function requiring the widest buffer for complete functionality, may be used to determine percent riparian function at a given buffer width in forested areas.

Non-forested Areas

For non-forested areas of the Interior Columbia River Basin (e.g., shrub-steppe, grasslands), we developed riparian function curves similar to those developed for forested areas. Important functions of non-forest riparian areas include bank stability (e.g., root strength), stream temperature control (e.g., shading), and filtration of sediment, nutrients, and pollutants from runoff. Large woody debris input is unlikely to be as important in non-forested areas of the Interior. Note however, that downstream input of wood from forested areas may have historically been important to non-forested areas, particularly near forest/nonforest edges. Thus for such areas where local stream conditions could be shaped by wood deposition from upstream sources, large woody debris input may be an important riparian function that is unaccounted for in our analysis. Our curves illustrate the change in each of these functions as buffer width increases, as described below.

Filtration— Figure 6 illustrates that percent sediment removal increases with increasing buffer width and decreases with increasing hill slope. Nitrogen and phosphorus (in their various forms) removal functions appeared similar (Figure 7).

Temperature control— Figure 8 shows the relationship of percent shading vs. buffer width, based on the studies we consulted. Although our dataset was limited to buffers <50m, shading tended to be greater in streams with wider buffers.

Bank stability— There were insufficient data in the literature to examine variation in root strength with increasing buffer width. However, we did surmise that fine root biomass and root strength of wetland vegetation (e.g., sedges, rushes) and grasses is greater than that of trees (Dunaway et al. 1994, Micheli and Kirchner 2002, Simon et al. 2002), and that of trees and grass is greater than that of agricultural plants (Tufekcioglu et al. 1999, Waldron and Dakessian 1982).

Overlaying the threshold curves for these three functions indicates that sediment filtration is the function requiring the widest riparian buffer for complete functionality. Thus, we suggest that this curve may be the best one to use to determine percent riparian function at a given buffer width for non-forested areas.

Approach and Methods

Step 1: Coarse Scale Screens

For each test basin and across the entire Columbia River Basin, we summarized the percentage of streams that run through each of three categories of land cover, as follows: (a) agriculture/urban, (b) grass/shrub, and (c) forested. We then determined the percentage of current riparian areas that have likely been converted to agriculture/urban. For the Riparian Screen, we divided converted stream length (e.g., streams running through the agriculture/urban habitat category) by total stream length. Results for each were summarized by subwatershed (6th field HUC), and mapped (e.g., none, <25%, 25-50%, >75% converted). For the Floodplain Screen, we divided converted floodplain area by total floodplain area in order to identify the amount of floodplain in every subwatershed that has been converted to agricultural land and other types.

Methodology:

DATA INPUTS:

1) *Current Land Cover layer*
   Current and historic wildlife habitat type layers are available for the entire Columbia River basin (Northwest Habitat Institute 2003; IBIS 2003). These data have been created by the Northwest Habitat Institute in cooperation with the Fish and Wildlife 2000 Program of the Northwest Power and Conservation Council (NWPCC). Current wildlife habitat types were derived from 1996 Landsat TM data, ancillary data (e.g., roads, streams, National Wetland inventory, other habitat inventories by local/state/federal agencies), and extensive field mapping.

2) *DEM-routed stream network*. Streams were generated from a DEM, with stream gradient and channel width estimated for each stream segment. (See Appendix C for complete description of stream layer generation.)

3) *6th field HUC layer*. Boundaries of 6th field hydrologic units (USGS), used for summary purposes. Source: Northwest Habitat Institute (2003).

4) *FEMA Floodplain maps*. Boundaries of the 100- year floodplain mapped by the Federal Emergency Management Administration. Boundaries in some cases exceed the area of the geomorphic floodplain, and in other areas omit portions of the floodplain behind levees.

5) USGS Land use/land cover layer. Land uses/land cover classes within the Columbia River basin.

STEPS:
To identify the area encompassed by the riparian analysis, we first selected all streams with gradients between 1 and 4% to remove reaches that will be covered under the floodplain screen (≤1%) and areas above the anadromous zone (≥4%). Within the selected stream area, we identified stream segments within any agricultural or urban land cover class, and calculated the proportion of stream length within each HUC6 that has been converted to agriculture or urban use.

To identify areas encompassed in the floodplain analysis, we selected all 100-yr FEMA floodplains that intersected stream reaches potentially accessible to salmon. Within those floodplains, we identified the proportion of floodplain area converted to any agricultural or urban land cover class, and calculated the proportion of floodplain area within each HUC6 that has been converted to agriculture or urban use.

Step 2: Aerial Photograph Analysis

To describe the condition of riparian buffers within each land cover category (agriculture/urban, forested, or grass/shrub), we randomly selected over 200 sample reaches throughout the Columbia River Basin (at least 100 reaches for riparian areas along non-floodplain streams, and at least 100 reaches within floodplains). At each sample site we measured buffer characteristics at 5 cross-valley transects from recent digital orthophotographs (most photographs taken in the late 1990s) (Figure 9). We summarized average buffer widths for left, right, and both (average of right and left) stream banks (Table 5). We also identified the vegetation type that most frequently occurred next to streams, and the dominant type of vegetation within a 100-m buffer on each side of the stream. Based on bootstrap analysis of preliminary data in the test basin, we determined that 20 photos in agricultural areas and 10 each in forested and shrub/grass areas allowed us to estimate the mean buffer width with a precision of less than +/-15%. Thus for the Columbia Basin, we decided to sample at least 50 photos in agriculture/urban areas, and at least 25 photos in each of the natural areas (forested and shrub/grass). Post-hoc bootstrap analyses suggested that these sample sizes were sufficient for confidence limits below +/-10%. Additionally, we performed some calibration of aerial photo interpretation by driving to sites and visually assessing what we saw. Planned future additional aerial photo samples and further ground-truthing should only increase our accuracy.

Analyses to Determine Sample Size

Using the Grand Ronde test basin dataset, we performed some analyses to determine (1) how many transects we need to sample per photo, and (2) how many photos to sample per stratum (agriculture/urban, forested, and shrub/grass).

Number of Transects Per Photo

We summarized the data for the Grande Ronde using both 10 and 5 transects per photo. To drop half of the transects, we omitted data from every other transect such that data included only transects 1, 3, 5, 7, and 9. We then compared summaries for both banks from this dataset having only 5 transects per photo to summaries for both banks from the original dataset having 10 transects per photo. We compared mean buffer widths with paired t-tests, and nearest and dominant vegetation types with 1-way ANOVAs. Results were nearly identical (Table 6), thus we reduced the number of transects sampled per photo for photo interpretation of the Columbia River Basin dataset.

Number of Photos per Stratum

To determine how many photos we should sample in each stratum, we performed a bootstrap analysis on mean buffer width using data for both banks (mean of left and right buffer widths from all transects within a photo). We began with the agriculture/urban stratum dataset, which was based on interpretation of 20 aerial photos. The bootstrap analysis consisted of several steps. First, we drew 20 samples randomly from our dataset, with replacement. Then we calculated the average of the first two samples, the first three samples, and so on until all 20 samples were included. An example of the cumulative mean and confidence intervals plotted against sample size from one randomly drawn dataset is shown in Figure 10A. Next, we randomly drew another 20 samples and calculated the cumulative average for that dataset. We performed the bootstrapping 1000 times, and calculated 95% confidence intervals for the entire dataset (Figure 10B). The 95% confidence interval (at α=0.05) shrinks to around +/- 14% at a sample size of 20. Thus, we can assume that with 20 photos in the agriculture/urban stratum, the true mean buffer width lies somewhere between 33.3 and 61.9 m. We performed a similar analysis for the forested and shrub/grass strata as well (Figure 11A and 11B, respectively). Although we analyzed only 10 photos for each of these strata, the results are similar to those for the agricultural stratum, indicating that we can analyze less photos for estimating buffer widths in these strata and still have similar confidence, likely because the agriculture/urban data were more variable.

Columbia River Basin Dataset

Based on results from the Grande Ronde test basin, we sampled at minimum of 50 photos in the agriculture/urban habitat stratum, 25 in the forested, and 25 in the shrub/grass stratum for both the riparian and floodplain screens. Post-hoc bootstrap analyses of these datasets indicate that our confidence intervals should be around +/-10% in all cases (Figure 12).

Statistical Analyses

We analyzed average buffer widths measured in aerial photos separately for the riparian and floodplain screens. Additionally, within each screen, we conducted separate analyses based on whether habitat strata (e.g., agricultural/urban, forested, or shrub/grass) were determined by the Northwest Habitat Institute data layer or by the dominant habitat type observed in each aerial photo (see the Results section for a discussion on misclassification rates and rationale for this decision). At this time, we did not statistically analyze differences in nearest and dominant habitat types observed in aerial photos.

The assumption of normality and heterogeneity of variance, basic underlying assumptions in an analysis of variance, were tested using standard procedures. A Shapiro-Wilks normality test was completed for each habitat stratum, determined by NHI data or aerial photo analysis, within both riparian and floodplain areas (Zar 1999). The results of this analysis indicated that the data departed significantly from a normal distribution ($P < 0.05$), and thus violated the assumption of normality (Table 7). Both Bartlett's test and Fligner's test (less sensitive to departures from normality) of heterogeneity of variance were completed in the same fashion (Zar 1999). These results indicated that the observed variance in average buffer widths within floodplain zones significantly departed from the assumption of homogeneity of variance ($P < 0.05$), while the observed variance in average buffer widths within riparian zones did not significantly depart from the assumption of homogeneity of variance ($P > 0.05$) (Table 8).

Given the results of these preliminary tests, we chose to use a non-parametric analysis because it should provide a more robust statistical analysis as compared to a parametric analysis of variance. Likewise, a Kruskal-Wallis rank sum test was used to determine statistically significant differences between habitat types within riparian and floodplain zones, and by NHI data or aerial photo analysis. Each of these tests was followed with a pairwise Wilcoxon rank sum test with P-values adjusted using the Holms method to determine where differences occurred (Zar 1999).

Results

Estimated percentages of riparian (non-floodplain) and floodplain areas converted to agricultural or urban land uses within each HUC6 in the Columbia River Basin are shown in Figures 10 and 11, respectively. Error matrices examining misclassification rates between the Northwest Habitat Institute land type classification and aerial photograph interpretation are shown in Table 9. There is generally close agreement between the two methods in forested lands. However, the error matrix indicates substantial uncertainty in the classification of agricultural lands and natural grass and shrub lands. In other words, areas identified as grass or shrub lands by aerial photos were frequently identified as agricultural areas by the NHI layer, and vice versa. Most commonly the mis-match of classifications occurs in grassy areas, which may be either natural or agricultural areas. As, neither aerial photography nor the NHI classification can be considered 'true' is not possible to assess which classification is in error without ground truthing. For more information on how the NHI data layer was created, see the metadata (http://www.nwhi.org/ibis/mapping/gisdata/docs/crb/crbcurhab_meta.htm), and reports on base layers used for Washington and Oregon (Johnson and O'Neill 2001) and Idaho (Scott *et al.* 2002).

Because misclassification rates were relatively high, we summarized riparian conditions separately for each classification method. Analyses of riparian conditions by the NHI classification describe the range of riparian conditions within each land cover type as mapped and summarized in the coarse-resolution GIS analysis (step 1). Analyses of riparian conditions in each land cover class observed in aerial photos are similar, but cannot be directly related to the coarse scale analysis. Results of aerial photograph interpretation for the riparian and floodplain screens are summarized in Table 10. Cumulative distributions of buffer widths within each category analyzed are shown in Figure 15.

The Kruskal-Wallis rank sum test indicated that there were statistically significant differences in average buffer widths between habitat strata among groups for each test (e.g., analysis of riparian buffer widths using aerial photo habitat classification) ($P < 0.05$) (Table 11). Pairwise post-hoc tests indicated that there were statistically significant differences in average buffer widths ($P < 0.05$) between agriculture/urban and forest/mixed, and agriculture/urban and shrub/grass strata within riparian and floodplain zones by NHI data and by aerial photo analysis (Table 12). However, with one exception, there were not statistically significant differences in average buffer widths ($P > 0.05$) between natural habitat categories (e.g., forest/mixed and shrub/grass) within riparian and floodplain zones by NHI data or aerial photo analysis (Table 12). Differences in average buffer widths between forest/mixed and shrub/grass habitats were statistically significant ($P < 0.05$) within the floodplain zone when classified by NHI data (Table 12).

Thus, regardless of whether we stratified habitat type by the NHI layer or by the dominant observations in aerial photos, buffer widths were narrower in the agricultural/urban stratum than in natural categories (Figure 15). Further, in the riparian screen (e.g., areas not in floodplains), both the nearest and dominant vegetation type in agricultural/urban areas and in forested areas was forest cover, whereas shrubs dominated in natural shrub/grasslands. This may suggest that areas that have been converted to agricultural or urban areas were historically predominantly forested, rather than shrub or grasslands. In riparian areas within the floodplain zone, the nearest and dominant categories were more often shrub or mixed in both the agricultural/urban and forested strata, but shrub or grass within the natural shrub/grass strata. Thus, in floodplain areas, most land use conversion likely occurred in shrub- or grasslands rather than forested areas.

Surface erosion on non-forested lands

Erosion on non-forested lands of the Columbia River basin is dominated by surface erosion and gullying processes, with relatively little contribution from mass wasting. Spatial variation in surface erosion rate is governed by several natural factors including hillslope angle, soil erosivity, rainfall intensity, and vegetation cover. Agricultural practices typically increase surface erosion by reducing vegetation cover and exposing more of the soil surface to rainfall impact and overland flow. The following analysis uses the long-standing universal soil equation as the basic model for estimating changes in surface erosion on non-forested lands as a function of conversion from grass or shrub cover to agriculture. It is necessarily a coarse resolution analysis (in order to have similar data quality across the entire basin), relying on geospatial datasets from ICBEMP and USGS to run the model. Results are summarized in an index of change in surface erosion rate for each HUC6 within the basin.

Products

1. Map of historical sediment supply ratings at HUC6 resolution
2. Map of current sediment supply ratings at HUC6 resolution
3. Map of difference between current and historical (divide historical by current to get percent increase in sediment supply rating) at HUC6 resolution

Approach and Methods

The equation at the basis of our approach is the Universal Soil Loss Equation (Wischmeier and Smith 1978), updated as the Revised USLE (Renard *et al.* 1996):

$$A = RKLSCP$$

Where,
A is the soil loss per unit area,
R is the rainfall and runoff factor,
K is the soil erodibility factor,
L is the slope factor,
S is the slope steepness factor,
C is the cover factor (also called the cropping practice factor),
and P is the support practice factor (representing conservation tillage practices).

Accounting for those values that are held constant for both historical and current estimates of sediment production within a grid cell (R), that do not vary spatially (L), or that we cannot estimate with sufficient detail (P) an index of erosion (E) is

$$E = K_p I_{slope} C$$

Where,
K_p is the soil erodibility weighting factor,

I_{slope} is an index of the change in erosion rate as a function of slope,
C is the weighting factor for vegetation cover.

Because all values except C are constant for a given cell in both the current and historical estimates, the ratio of $(E)_{ag}/(E)_{natveg}$ is an index of the change in erosion rate for a grid cell. Values are summarized at the HUC6 level by averaging all cells within each HUC6 to generate maps of mean sediment supply ratings for historical conditions, current conditions, and the ratio of current/historical.

To estimate E we required values for K_p, I_{slope}, and C. Because we do not have K values for individual soils in the ICBEMP soils layer (we have the percentage of soils with K>0.37 in each soil type polygon), we used a weighting factor for K_p that is a function of the percentage of soils with K greater than 0.37 (Table 13). Values of I_{slope} are calculated as a function of grid cell slope (from the 10m DEM) using the equation

$$I_{slope} = 3.41(S^2) + 0.93(S),$$

which we derived (using the USLE) to analyze sensitivity of erosion rate to hillslope gradient (m/m) (Figure 16). Normative parameters for the sensitivity analysis were R = 25 (an intermediate value for interior rangelands from Figure 2-15 in Renard *et al.* 1996), K = 0.37 (arbitrary value based on the ICBEMP), L = 72 (length of standard slope from Wischmeier and Smith 1978), S = slope of grid cell, LS = ((L/72.6)^0.5) * (65.41*((SIN(S))^2) + (4.56*SIN(S)) + 0.065) (a non-linear function of L and S based on Wischmeier and Smith 1978, p. 13), C = 0.01, (value for 80% grass cover from Dunne and Leopold 1978, p 529), and P = 1.

Weighting factors for vegetation cover (C) were selected from various sources as indicated in Table 13. These factors are chosen to represent average relative changes in erosion under various cover types, and are not intended to predict actual erosion rates. Note that we did not have sufficient detail on erosion control practices to incorporate a value of P into the analysis. Literature on the subject indicates that erosion control practices (e.g., no–till seeding, strip cropping) can in some cases substantially reduce erosion rates (e.g., Wischmeier and Smith 1978, Ebbert and Roe 1998 [USGS Fact Sheet FS-069-98]).

Final ratings for current and historical conditions (for each grid cell) were derived by multiplying the three weighting factors together as in the RUSLE. Examples of erosion rating combining all of these factors are shown in Table 15. Values were summarized at HUC6 resolution by first calculating the ratio of current/historical rate for each grid cell, then calculating the area-weighted average ratio for each HUC6.

Results

As one would expect from the structure of the model, the highest erosion ratings for historical conditions are found in areas with steeper hillslopes and greater area in shrublands (Figure 17). Current sediment supply ratings (Figure 18) and the change in ratings from historical conditions (Figure 19) are driven predominantly by the location of agricultural land uses. Changes in sediment supply ratings are highest where historical grasslands (which had relatively low surface erosion rates) have been converted to agricultural land uses. Across the entire analysis area, the largest changes are concentrated in the Palouse region where relatively steep slopes of the loess deposits have been converted from grasslands to agriculture (mainly small grains).

Mass wasting and surface erosion on forested lands

A substantial literature concerning effects of forest practices (e.g., logging and road building) on mass wasting processes has established that clearcut-cut logging and road building significantly alter sediment supply rates from landsliding (e.g., see summaries in Sidle *et al.* 1985 and Meehan 1991). Based on 12 landslide studies in the Pacific Northwest summarized by Sidle *et al.* (1985), landslides in clearcut produce sediment at 2 to 41 times the rate from forested areas, and road-related landslides produce sediment at 26 to 212 times the rate from forested areas. However, only one of those studies was east of the Cascade Mountains. Sediment supply rates from landsliding in mature forest areas range from 11 to 87 m^3/ km^2/yr in wetter forests west of the Cascade Mountains, but drier forests east of the cascades tend to have lower rates for landsliding from forested areas (6 to 21 m^3/ km^2/yr). However, sediment supply rates from clearcut areas may be higher east of the Cascades than to the west (138-3193 m^3/ km^2/yr compared to 25-322 m^3/ km^2/yr), while sediment supply rates from roads are similar across the two regions (1585-15,565 m^3/ km^2/yr west of the Cascades compared to 1315-11,316 m^3/ km^2/yr east of the cascades). Intense, stand-replacing fires can dramatically increases erosion rates in forested areas of the Columbia basin (Megahan *et al.* 1995, Meyer *et al.* 2001), and much of that increase is due to elevated rates of mass wasting.

We generate a simple landslide erosion rate index based on empirical rates of landslide sediment production from watersheds within the Columbia River basin. The erosion rate index is a simple function of forest ownership class and length of roads, which estimates the order of magnitude of changes in sediment supply among land use classes (e.g., 10^0x, 10^1x or 10^2x). We elected to base the index on order magnitude values because sediment supply rates vary by more than one order of magnitude among landslide studies, and there are few data points from which to calculate mean or median values (Table 16). There are also too few studies to empirically estimate how natural factors (e.g., lithology or terrain) affect sediment supply rates across the basin. Hence, we were forced to assume that natural factors affecting erosion rates do not substantially alter the ratios of sediment production from various land use classes, even though natural rates may vary. We were unable to estimate the influences of recent fires or shifts in fire regime on recent sediment supply rates due mainly to the lack of consistent fire maps across the Columbia basin.

Products

1. Map of estimated difference between current and historical sediment supply (historical values are all 1 in this analysis) at HUC6 resolution.

Approach and Methods

The modeling area is only that area that was historically forested. The mass wasting assessment was further restricted to areas with slope greater than 25 degrees, which encompasses the majority of landslide prone areas (Sidle *et al.* 1985). Within the mass wasting assessment area, we estimated the average area in forest and clearcuts based on data from a study of Washington forest practices (Collins 1996). Average clearcut areas in any one year were calculated based on the median annual harvest rates in each ownership class (Table 17). Clearcut areas were then calculated for each ownership class in each 6th field HUC. To calculate road areas we multiplied road length by 10 m (the approximate width of forest roads including cut and fill slopes).

Each cover class for the mass wasting assessment was then assigned the erosion rate factor (eastside forest = 1, westside forest = 3, clearcut = 10, road = 100, other lands eastside = 1, and other lands westside = 3). Surface erosion on roads was estimated using the area of roads in all forested lands, and an erosion weighting factor of 100. The value of 100 is the same order of magnitude as proportional increases in surface erosion from roads documented in Megahan and Kidd (1974) (220 times more sediment from roads) and Reid and Dunne (1984) (130 times more sediment on a heavily used road). For each 6th field HUC in the interior Columbia basin, the weighted average erosion rating under current forest practice regimes (E_{curr}) then calculated as

$$E_{curr} = \frac{\left(A_{forest>25} \times 1\right) + \left(A_{clearcut>25} \times 10\right) + \left(A_{road>25} \times 100\right) + \left(A_{other} \times 1\right) + \left(A_{road,all} \times 100\right)}{A_{total}}$$

For each HUC6 in the lower Columbia basin, the weighted average erosion rating under current forest practice regimes (E_{curr}) then calculated as

$$E_{curr} = \frac{\left(A_{forest>25} \times 3\right) + \left(A_{clearcut>25} \times 10\right) + \left(A_{road>25} \times 100\right) + \left(A_{other} \times 3\right) + \left(A_{road,all} \times 100\right)}{A_{total}}$$

Results

The highest predicted increases in sediment supply are concentrated in the lower Columbia River basin (the Cascade Mountains and Coast Ranges) (Figure 20), where there is a high proportion of forest land, high average timber harvest rates, and high road densities. Within the Interior Columbia basin, the largest predicted increases in sediment supply extend across all forested mountain ranges, with the notable exception of wilderness areas. The Salmon River basin, for example, is largely forested, but extensive tracts of wilderness area have relatively little logging or road building, resulting in low estimated erosion rates.

Our ratings do not account for the effects of recent fires, which can substantially increase rates of surface erosion and gullying (e.g., Meyer *et al.* 2001). The National Fire Occurrence Database (www.fs.fed.us/fire/fuelman) records point locations and areas of fires, but lacks fire perimeter information. Without fire perimeter maps, we were unable link area burned with hillslope data or land management class in order to estimate effects on erosion rates. Local, specific fire information may be used within the sub-basin planning process to indicate where fires have significantly increased sediment supply, and to identify specific restoration actions that may help reduce sediment supply to streams.

Water quality

Pesticides are frequently detected in salmon habitat throughout the Columbia Basin. For example, 50 different pesticides were recently detected by the U.S. Geological Survey in the Willamette basin (Wentz *et al.* 1998), and 43 different pesticides have been detected in the lower Yakima River (Rinella *et al.* 1999). Sub-lethal effects of these pesticides on salmon survival and reproductive health are largely unknown, especially when they enter streams in complex mixtures. Trace metals and petroleum-based products also enter surface waters in high concentrations in urban areas (Wentz *et al.* 1998), and their effects on salmon are also poorly understood. Recent studies indicate that at least some of these compounds dramatically alter olfactory-mediated behaviors in salmon (Scholz *et al.* 2000), which can result in increased mortality during juvenile life stages. The potential for increased mortality combined with high exposure potential creates a critical uncertainty in our ability to identify actions necessary to improve population status.

In light of the potentially large impact on in-stream survival, the water quality analysis focuses ranking the relative exposure of stream reaches to non-point pollutants (mainly current-use herbicides, insecticides and fungicides, as well as trace metals and petroleum-based products). We rank the relative potential exposure based on recent studies that document concentrations of pollutants in surface waters (e.g., Wentz *et al.* 1998, Rinella *et al.* 1999). High exposure potential indicates a greater likelihood that salmon may be affected by pollutants, but does not necessarily equate to a change in survival rates because effects of most pollutants are poorly understood at present.

Products

1. Map of ranked exposure to pesticides and urban runoff at HUC6 resolution.

Approach and methods

We ranked exposure to non-point source pollutants (here referring to pesticides or urban runoff harmful to aquatic invertebrates or fishes) based on relative application and detection rates in varying land cover classes. In general, pesticide applications on dryland crops are roughly an order of magnitude lower than on irrigated crops of the Central Columbia Pleateau (Wagner *et al.* 1996, Ebbert and Embrey 2002), and certain trace metals are found in surface waters at concentrations 10 times higher in urban areas than in other land uses (Wentz *et al.* 1998). Concentrations of many compounds in the Columbia basin exceed criteria for protection of aquatic life, including 4 pesticides in the Palouse basin (Roberts and Wagner 1996), 6 pesticides in the Yakima basin (Ebbert and Embrey 1992), 6 pesticides in the lower Clackamas basin (Carpenter 2004), and 10 pesticides in the Willamette basin (Wentz *et al.* 1998).

Because of the many uncertainties in determining the relative impact of non-point pollutants on salmon, we did not attempt to rate exposure potential in proportion to its effect on salmon. Rather, we rated the relative potential exposure to salmon with a simple ranking system, assigning values of 1 to low potential exposure, 2 to moderate potential exposure, and 10 to high potential exposure. This procedure is analogous to, but simpler than, the methodology developed by Black *et al.* (2000). Intensive agriculture (e.g., orchards, vineyards, row crops) and urban areas are associated with very high concentrations of pesticides and other pollutants, usually at least an order of magnitude higher than that found in streams bordered by other land uses (Wentz *et al.* 1998). Hence intensive agriculture and urban areas were given the highest value (a value of 10). Dryland agriculture was assigned a value of 2 to indicate higher potential exposure to pesticides than non-agriculture and non-urban areas. All other areas were assigned a value of 1. Values were assigned based on USGS land use and land cover data as shown in Table 18. For each HUC6, we record the dominant rating and the area-weighted average rating to indicate the relative potential for exposure of salmon to non-point source pollutants.

Results

The distribution of likely exposure to non-point source pollutants largely mirrors land uses within the Columbia River basin, with most likely exposure in areas dominated by intensive (usually irrigated) agriculture and urban development (Figure 21). Non-irrigated agriculture (usually small grains) is predicted to present some level of risk, and forested areas present a low potential exposure to harmful current-use pesticides.

HYDROLOGICAL FLOW AND DIVERSION LIMITATIONS WITHIN INTERIOR COLUMBIA BASIN STEELHEAD AND CHINOOK ESUs

OVERVIEW

Goal:

Evaluate and quantify the likely relative degree of alteration for instream flows and entrainment due to water withdrawls as related to Steelhead and Chinook populations within the Interior Columbia Basin ESUs.

Products:

1. (i) Tables summarizing flow and diversion limiting factors by population sub-basins and reach use structure, (ii) and maps illustrating the results of these summaries with populations classified by hierarchical rankings.

Background:

Streamflow is a fundamental component supporting hydrological systems (Hortness and Berenbrock 2001). Moreover, it is a crucial factor in sustaining salmon and steelhead viability throughout their riverine habitats. Sufficient flows must be present for both the upstream journey of adults to their natal streams and the out-migration of juveniles to marine environments. While historic flow regimes have been dictated primarily by physiographic and seasonal climatic variations, the diversion of water resources for industrial, agricultural, and household uses has played an ever-increasing role in determining patterns of streamflow. As a result, stream dewatering has become a seasonally endemic problem, impairing vital elements within the anadromous lifecycle (Chapman 1993). Indeed, many cases of extirpation have been explained by the over-utilization of instream water (Fulton 1968, 1970).

In addition to their general impact upon streamflow, diversions also impinge on salmon and steelhead survival by entraining juveniles into irrigation systems (Neeley 2000). These effects are entirely related to the exact planar location of diversion points along salmonid occupied streams, whereas instream flow conditions are a result of accumulated water withdrawls in all upstream habitats. Untold numbers of out-migrants have perished on their seaward journey after being diverted from migratory corridors. Recalling the observations of a traveler near the mouth of the Umatilla River around 1900, Lichatowich (1999, pg. 71)) writes:

> "Years later, Campbell would vividly remember the view from that little hotel. In the spring and summer, he looked out over the nearby farmland and saw cultivated fields covered with a shimmering layer of silver where millions of salmon smolts lay drying in the desert sun. The irrigation diversions in the Umatilla River intercepted young steelhead and salmon as they migrated downstream toward the sea and sent them into feeder ditches that eventually carried the fish onto the farmers' fields, where they died."

In recent times, entrainment risks have been tempered through selected utilization of diversion screens and bypass sytems, but even these mechanisms may not entirely mitigate for juvenile mortality (Neeley 2003). Our goal is to quantify current diversion data as it relates to extant steelhead and chinook populations within the Interior Columbia Basin ESUs. Rather than a direct appraisal of mortality, our focus will be to measure the relative differences of diversion density, and its effects on streamflow within populations. The summaries produced from our analyses will provide a general assessment of flow and diversion risks in order to prioritize recovery needs.

Approach:

We employed two distinct analytical approaches for quantifying diversion and flow related impacts. First, we developed a general water-right and flow summary for each independent chinook and steelhead population. This approach quantified diversion attributes within a population's hydrological basin, independent from its geographic reach structure, and was intended to illustrate the general condition of flow affected by instream diversions. In our second analysis, we incorporated a population's hydrographic network for quantifying diverted flow as it specifically relates to salmon and steelhead stream use. Diversion and flow data were collected for population reaches inhabited by any part of the anadromous lifecycle, as well as for the migration pathways downstream from the population's sub-basin.

Both analyses required spatial data that describe the distribution of diversions, streamflow, and salmonid population structures. We used two primary sources for flow and diversion information, and both required varying degrees of preparation and conversion from their native formats. Multiple sources were synthesized to create the population spatial themes. Data production efforts were the most resource intensive aspect of our analytical investigations.

Data Development

Our most challenging data production scheme was development of the water-right diversion GIS coverage. This spatial theme was built from a 1997 National Marine Fisheries Service (NMFS) study that compiled and reduced non-spatial water-right data from Washington, Oregon, and Idaho. The NMFS's effort produced a single database for each state, and contained information describing the legal rates of withdraw (in CFS) and location of each unique water-right. Locations were expressed as legal descriptions referencing the Public Land Survey System (PLSS). The PLSS is a national property survey composed of individual township, range, and section lines (TRS), and each water-right identified its TRS coordinate. In addition, the legal descriptions often included quarter, and sub-quarter locations (NW, NE, SW, SE). By using an existing PLSS polygonal coverage as a reference theme, we were able to geo-code the water-right features using the TRS spatial coordinates. This process translated over 77,000 database records into centroids representing their associated PLSS section.

Using the previously described methodology, our water-right placement accuracy was approximately 800 meters, recognizing the dimensions of a PLSS section are one mile (~1.6km) per side. In an effort to improve precision, we developed a script that shifted the water-right point locations to their quarter, and sub-quarter TRS positions. This meant that records with quarter sections identified had their positional accuracy improved to 400 meters, and where sub-quarter sections were described it was improved to 200 meters. The Oregon and Idaho databases contain partial quarter and sub-quarter coordinates parsed from their legal descriptions, and were identified as unique fields. However, even with the improved precision from the quarter and sub-quarter information, the overall accuracy limited useful analysis since the water-right locations were not necessarily spatially coincident with the hydrological features identified by their attribute table. In order to correct these geographic discrepancies, we used both automatic and manual theme editing techniques to snap water-right points to their associated stream feature. The final theme contained only diversions within NMFS recovery domains and included 11,147 data points.

In addition to having accurate spatial data describing water-right diversions, our analyses made it necessary to obtain streamflow values for hydrological features within the Interior Columbia ESUs. The EPA's River File Version 1 (RF1) (EPA 1982) was the only source that provided consistent data throughout the entire study area. Developed in 1982, this dataset contains flow values that were modeled by analyzing the relationship between USGS gage station measurements and upstream basin size. The EPA calculated both mean and low flow (70% excedance) for over 60,000 independent stream reaches in the continuous United States. Although the EPA describes limitations associated with their data, it is currently the only continuous streamflow information available for multi-state extents. Ultimately, we transferred the RF1 streamflow attributes to the water-right diversion theme using proximity analyses. The three nearest stream features to each diversion were identified, and a final join was performed using the hydrological feature name and proximity. We used various attribute manipulation techniques in order to match feature abbreviations (e.g. W FK = West Fork, R = River) and case structure so that attribute tables could be joined based on feature names. Both a proximity analysis and feature name field join were necessary due to instances where a diversion's closest stream feature did not match its corresponding stream name. This problem was most evident near stream intersections and in areas with closely parallel stream features.

In order to determine how streamflow and diversions impact salmon and steelhead, various spatial themes were developed for describing population structure. As a first step, we represented independent anadromous populations as hydrological basins. Population independence was determined by the Interior Columbia Technical Recovery Team (ICTRT), and was based on unique genetic, life history, morphological, and spatial characteristics within the interior Columbia Basin steelhead and chinook ESUs (ICTRT 2003).

We delineated watersheds containing chinook and steelhead populations by using standard GIS hydrographic analysis tools and USGS 1:24,000-scale level 2 Digital Elevation Models (DEM). Once we located the downstream extent for our populations, and defined these locations as hydrological pour points, watershed development scripts were utilized for representing populations as polygonal drainage features. Additional population boundary enhancements have been accomplished by edge matching polygons with other commonly used watershed layers. These included the 5th and 6th field watersheds developed by the Natural Resource Conservation Service (NRCS), the

Regional Ecosystem Office (REO), and the Idaho Department of Water Resources (IDWR), and watersheds produced for the British Columbia Watershed Atlas by the Ministries of Sustainable Resource Management and Water, Land & Air Protection. If further enhancements to this spatial data layer are necessary, we will likely use these existing sources to better coincide with industry standard watershed themes.

As a complement to our sub-basin defined populations, we also developed reach, lifecycle, and species specific data layers. Themes describing the primary anadromous lifecycle stages were necessary in order to determine all possible stream interactions between salmon and diversion structures. Digital spatial themes were compiled from existing sources, including Streamnet, the Oregon Department of Fish and Wildlife (ODFW), the Washington Department of Fish and Wildlife (WDFW), and the Idaho Department of Fish and Game (IDFG). Using this state agency data, we created use defined themes describing spawning, rearing, and migration corridors for summer and winter steelhead, and summer, spring, and fall chinook. Spatial layer scales were inherited from the sponsoring agencies, and were left unaltered in our data development efforts. Once the various lifecycle and species specific themes were separated for each state, we reduced and standardized the attribute tables so that GIS layer merging techniques could be applied. The appended outputs were re-projected into a common coordinate system and combined to form continuous feature layers within the study area. These spatial themes illustrate a presence/absence view, in regards to the various lifecycles, of the population structure within the sub-basins.

Analysis

INSTREAM FLOW

Using our diversion and distribution spatial data, we quantified flow limitations within occupied sub-basins and reaches. Basin areas were used as the summary unit for our analysis, recognizing that the reduction of streamflow within anadromous populations is a result of accumulated water withdrawls in all upstream areas within a watershed regardless of salmon and steelhead use. Our preliminary effort included summaries for population basins and the number of diversions, total legal rate of withdraws, and the sum of the percent of flow diverted. The percent of flow diverted (for mean and low flow) was calculated for each water-right by dividing the legal rate with the EPA derived streamflow values. We also included the number of diversions within the 90th percentile (greatest 10%) of the legal rates of withdraw for each population. Only water-rights contained completely within population watersheds were included, and summaries were reported by ICTRT defined steelhead and chinook populations. It should be noted that summaries based upon population sub-basins provide a general evaluation of diverted flow, and related limiting factors, but do not provide a complete analysis between specific use extents and diversion locations.

DIVERSION ENTRAINMENT

By utilizing the reach specific use data, we evaluated diversions in relation to salmon and steelhead stream distribution. Similar to the population sub-basin approach, this summary included the total number of water-rights, legal rates in CFS, percent of flow diverted, and the number of diversions in the 90th percentile. However, only diversions within reaches used for spawning, rearing, and/or migration were included in the analysis. This was done because salmonid entrainment occurs specifically at the point where individuals intersect the location of a diversion structure, and diversions outside of occupied streams would not pose any risk. With this in mind, we only queried stream diversions that were spatially coincident with the utilized reaches as well as all diversions that would be encountered downstream from a population's watershed outlet. Downstream intersections were determined through use of GIS hydrological tools and the flow direction and accumulation themes developed from the USGS DEM. Our team developed a script that delineated flow paths using the watershed pour points and the DEM generated themes, and then selected the water-rights entrained on those paths. Summaries were calculated for these diversions and then added to the within population values. The final water-right summarizations were presented by ICTRT defined populations.

Results

Within chinook sub-basins (Table 1), the total number of diversions ranged from 1 within the Loon Creek (MFLOO) and Minam River (GRMIN) populations to 950 within the Lemhi River population (SRLEM). For steelhead populations (Table 2), the range was from 2 within the Lolo Creek (CRLOL-s), Lochsa River (CRLOC-s), Rock Creek (MCROC-s), and Secesh River (SFSEC-s) populations to 950 within the Lemhi River population (SRLEM). The sum of percent of flow diverted (mean flow) for chinook populations ranged from 0% at the Secesh River (MFSEC) to 52% at the Lemhi River (SRLEM). For steelhead, the values ranged from 0% at the Lochsa River (CRLOC-s), Lolo Creek (CRLOL-s), the Deschutes River Westside Tributaries (DRWST-s), Rock Creek (MCROC-s), Secesh River (SRSEC-s), and Hell's Canyon Tributaries (SNHCT-s) to 69% within the Okanogan River (UCOKA-s).

Our summaries illustrate a varying distribution of flow and diversion impacts within the interior Columbia Basin recovery domains. It is not surprising that water-right diversions are most concentrated in watersheds with high agricultural production, and conversely the fewest diversions appear in wilderness, or near-wilderness dominated landscapes. Even though these results do not challenge generally accepted paradigms, they do quantify limiting factors in a more meaningful way.

In our second analysis, which quantified diversions based on salmon and steelhead use and downstream migration corridors, we also found notable differences between populations. For steelhead ESUs (Table 4), the values ranged from 47 diversions within the Rock Creek population (MCROC-s) to 964 in the Walla Walla River population

(WWMAI-s). Within chinook ESUs (Table 3), the number of diversions ranged from 304 at the Wenaha River (GRWEN) to 891 at the Lemhi River (SRLEM). The total number of diversions for the Lemhi River population (for example) is different in this analysis compared to the sub-basin summary because it only includes reaches occupied by salmon and steelhead, rather than all diversions within the watershed. Similar to our first analysis, variation in the distribution of diversions appears to be related to land use types. However, it is also influenced by their position relative to the most downstream point in our study area and the resultant lengthening of the migration pathways. As migration distances increase, so does the opportunity for entrainment. It is probable that survival rates during out-migration are directly related to the number of diversion encounters.

Discussion and Conclusion

Some limitations were encountered during our analyses. First, we made no distinction between diversion types, or if screening or bypass systems were present, and both characteristics have been shown to effect survival (Neeley 2003). Ideally, water-right diversions would be weighted by these factors so that their relative impacts could be calculated. However, currently available data sources do not include this information. Secondly, chinook and steelhead stream use was assumed to be equally distributed within populations, and diversions were not weighted by current salmon and steelhead densities. In the future, we would assign greater weight to diversions within high density areas (in terms of fish use) by incorporating escapement data. Thirdly, due to time constraints we did account for the potentially discontinuous temporal aspects of water-right utilization and stream use. More thought should be given to when irrigation pressures coincide with chinook and steelhead use. For example, diversion impacts in August are far different than April due to variations in streamflow and irrigation needs, and these realities should be considered. Recognizing these limitations, and certainly others, we still believe a general accounting of how diversions are distributed throughout the interior Columbia Basin ESUs is a valuable exercise by highlighting the relative differences of flow and entrainment impacts between anadromous populations.

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Table 1. Diversion summaries for Chinook population watersheds.

Population	Total Diversions	Total CFS	Sum of % of Flow Diverted (low flow)	Sum of % of Flow Diverted (mean flow)	Number of Diversions in 90th Percentile (low flow)	Number of Diversions in 90th Percentile (mean flow)
GRCAT	240	341	198.1	9.6	135	19
GRLOO	4	72	344.0	20.4	3	3
GRLOS	286	482	32.3	4.3	7	10
GRMIN	1	100	2.6	0.2	1	1
GRUMA	52	154	62.6	0.9	17	3
IRBSH	14	50	3.1	0.4	2	2
IRMAI	39	52	3.6	0.4	2	2
MFBIG	11	17	2.2	0.4	0	1
MFCAM	3	10	1.6	0.1	1	0
MFLMA	3	9	2.7	0.2	1	1
MFLOO	1	1	0.1	0.0	0	0
MFMAR	4	11	0.6	0.1	0	0
MFUMA	4	252	1.9	0.4	1	1
SFEFS	7	23	1.0	0.2	0	1
SFMAI	33	262	8.4	2.3	3	4
SFSEC	2	6	0.2	0.0	0	0
SNASO	40	5	0.5	0.1	0	0
SNTUC	91	120	35.6	1.4	31	1
SRCHA	6	12	1.6	0.3	1	1
SREFS	65	115	12.9	2.3	1	4
SRLEM	950	1732	227.7	52.2	54	126
SRLMA	512	1838	127.2	36.9	25	48
SRLSR	339	1342	76.7	12.1	16	17
SRNFS	59	62	13.0	2.4	5	5
SRPAH	225	968	169.7	34.2	39	79
SRPAN	38	19	7.1	1.3	2	3
SRUMA	66	327	24.9	3.1	4	7
SRVAL	38	88	4.8	1.1	1	1
SRYFS	9	36	1.3	0.2	0	0
UCENT	112	62	2.3	0.5	0	0
UCMET	337	708	35.6	3.7	5	5
UCWER	234	695	1440.4	1.7	22	3

Table 2. Diversion summaries for Steelhead population watersheds.

Population	Total Diversions	Total CFS	Sum of % of Flow Diverted (low flow)	Sum of % of Flow Diverted (mean flow)	Number of Diversions in 90th Percentile (low flow)	Number of Diversions in 90th Percentile (mean flow)
CRLMA-s	347	5312	528.8	4.2	23	8
CRLOC-s	2	3	0.0	0.0	0	0
CRLOL-s	2	0	0.3	0.0	0	0
CRSEL-s	9	201	0.6	0.1	0	0
CRSFC-s	27	157	14.1	1.5	3	4
DREST-s	195	1316	72.9	14.0	17	26
DRWST-s	7	2	0.0	0.0	0	0
GRJOS-s	21	17	2.7	0.3	0	0
GRLMT-s	40	106	130.3	3.6	3	2
GRUMA-s	430	793	1544.1	51.4	89	67
GRWAL-s	287	582	34.9	4.5	8	12
IRMAI-s	53	103	6.7	0.8	4	5
JDLMT-s	386	369	2296.9	15.5	49	33
JDMFJ-s	37	42	142.5	2.6	16	9
JDNFJ-s	181	167	793.7	2.2	34	7
JDSFJ-s	54	23	54.0	1.7	0	6
JDUMA-s	455	437	242.9	14.2	18	21
MCFIF-s	241	154	4.5	1.4	0	2
MCKLI-s	130	444	62.4	2.9	19	8
MCROC-s	2	0	1.5	0.0	1	0
MCUMA-s	564	1416	2651.7	15.2	100	64
MCWSA-s	40	124	0.7	0.2	0	0
MFBIG-s	13	30	5.4	0.6	2	3
MFUMA-s	9	265	2.8	0.6	1	1
SFSEC-s	2	6	0.2	0.0	0	0
SFSFS-s	19	66	3.3	0.6	1	3
SNASO-s	161	35	117.9	0.8	29	0
SNHCT-s	5	3	0.2	0.0	0	0
SNTUC-s	94	123	40.0	1.4	3	2
SRCHA-s	25	230	7.8	2.2	2	4
SREFS-s	422	1463	88.8	24.7	13	43
SRLEM-s	950	1732	227.7	52.2	51	183
SRLSR-s	368	1350	77.0	12.2	16	20
SRNFS-s	59	62	13.0	2.4	5	7
SRPAH-s	327	1354	202.9	44.8	46	122
SRPAN-s	42	24	8.0	1.5	2	5
SRUMA-s	166	555	49.1	8.3	9	21
UCENT-s	112	62	2.3	0.5	0	0
UCMET-s	337	708	35.6	3.7	5	7
UCOKA-s	460	2886	23982.8	69.3	73	4
UCWEN-s	234	695	1440.4	1.7	9	4
WWMAI-s	872	1218	17267.1	37.4	261	81
WWTOU-s	367	1289	198.5	6.5	5	6

YRNAC-s	333	1946	79.0	10.0	11	12
YRTOS-s	9	15	10.3	0.1	1	0
YRUMA-s	526	3700	167.9	33.8	35	52

Table 3. Diversion summaries for Chinook population reach structure (population diversion encounters).

Population	Total Diversions	Total CFS	Sum of % of Flow Diverted (low flow)	Sum of % of Flow Diverted (mean flow)	Number of Diversions in 90th Percentile (mean flow)
GRCAT	595	9248	122.6	9.7	47
GRLOO	308	8897	333.6	19.8	2
GRLOS	535	9244	19.9	3.1	13
GRUMA	390	9014	74.0	1.0	5
GRWEN	304	8824	0.3	0.1	0
IRBSH	319	8869	3.4	0.5	3
IRMAI	330	8869	3.9	0.5	3
MFBEA	348	9104	2.7	0.6	2
MFBIG	354	8867	2.3	0.4	3
MFCAM	348	8860	2.8	0.3	2
MFLMA	348	8861	3.2	0.4	3
MFLOO	347	8855	0.9	0.2	1
MFMAR	352	9115	3.3	0.7	4
MFSUL	347	8854	0.8	0.2	1
MFUMA	349	6106	2.7	0.6	2
SFEFS	352	8874	1.3	0.3	3
SFMAI	370	9109	8.5	2.3	5
SFSEC	348	8858	0.5	0.1	0
SNASO	318	8819	0.7	0.2	1
SNTUC	340	8825	5.9	1.5	7
SRCHA	347	8861	1.5	0.3	1
SREFS	625	10207	18.7	3.9	17
SRLEM	891	9940	36.8	11.4	50
SRLMA	804	10393	53.0	15.0	76
SRLSR	479	9849	40.5	7.0	12
SRNFS	413	8914	12.7	2.3	12
SRPAH	574	9913	47.2	9.6	59
SRPAN	367	8857	1.3	0.3	2
SRUMA	658	10444	30.4	4.6	28
SRVAL	625	10198	10.4	2.6	14
SRYFS	585	10316	6.8	1.7	9
UCENT	580	820814	29.6	7.3	12
UCMET	840	1261619	66.8	13.6	22
UCWER	581	611355	1444.7	6.2	15

Table 4. Diversion summaries for Steelhead population reach structure (population diversion encounters – includes downstream diversions).

Population	Total Diversions	Total CFS	Sum of % of Flow Diverted (low flow)	Sum of % of Flow Diverted (mean flow)	Number of Diversions in 90th Percentile (mean flow)
CRLMA-s	581	13916	49.1	1.6	8
CRLOC-s	381	13306	3.7	0.3	1
CRLOL-s	336	13281	3.7	0.3	1
CRNFC-s	322	13370	3.7	0.4	1
CRSEL-s	388	13504	4.3	0.4	2
CRSFC-s	429	13893	16.1	1.7	6
DREST-s	95	2500	22.5	3.7	7
DRWST-s	57	1761	0.2	0.1	1
GRJOS-s	308	8832	2.6	0.3	3
GRLMT-s	313	8831	1.2	0.2	1
GRUMA-s	720	9585	1377.7	50.2	102
GRWAL-s	536	9344	22.4	3.3	14
IRMAI-s	343	8920	6.9	0.9	6
JDLMT-s	412	2147	2142.9	10.6	54
JDMFJ-s	389	2132	179.3	2.8	14
JDNFJ-s	404	2132	617.3	1.9	14
JDSFJ-s	329	2019	27.3	0.4	2
JDUMA-s	743	2444	119.2	13.8	49
MCFIF-s	231	1836	3.9	1.3	5
MCKLI-s	76	459	23.2	1.1	2
MCROC-s	47	1796	0.0	0.0	0
MCUMA-s	476	5371	1992.1	6.4	41
MCWSA-s	30	94	0.3	0.1	0
MFBIG-s	356	8874	4.5	0.6	4
MFUMA-s	353	9117	3.3	0.7	4
SFSEC-s	349	8859	0.6	0.1	0
SFSFS-s	361	8917	3.6	0.7	5
SNASO-s	415	8845	107.5	0.8	4
SNHCT-s	292	8818	0.3	0.1	0
SNTUC-s	343	8828	10.3	1.5	7
SRCHA-s	362	9076	7.4	2.2	4
SREFS-s	801	10403	35.3	9.2	55
SRLEM-s	893	9941	37.5	11.6	52
SRLSR-s	494	9853	40.6	7.1	12
SRNFS-s	413	8914	12.6	2.3	12
SRPAH-s	594	9948	58.5	13.6	77
SRPAN-s	368	8859	1.6	0.3	3
SRUMA-s	721	10623	54.0	9.6	47
UCENT-s	580	820814	29.6	7.3	12
UCMET-s	840	1261619	66.7	13.6	22
UCOKA-s	903	1261531	191.6	11.3	12
UCWEN-s	581	611355	1444.7	6.2	15
WWMAI-s	964	8016	17129.2	24.5	81
WWTOU-s	552	8353	178.5	6.5	10
YRNAC-s	660	18807	215.8	10.6	28
YRTOS-s	315	10603	19.9	0.9	4
YRUMA-s	823	20562	289.9	30.1	95

NATURAL AND ANTHROPOGENIC BARRIER IMPAIRMENTS IN THE INTERIOR COLUMBIA RIVER BASIN *(DRAFT)*

Overview:

Columbia River salmon and steelhead evolved in systems without structural anthropogenic barriers (e.g., dams and culverts) and were dependent upon the river current to aid in their migration to the ocean (NPPC, 2003). During the last century numerous anthropogenic barriers have been constructed within the Columbia River Basin that impede salmonid migration to varying extents. Within the range of all threatened and endangered evolutionary significant units (ESUs), chinook salmon (Federal Register, 1998) and steelhead that occupy similar regions face a multitude of barriers that limit the access of juvenile and adult fish to essential freshwater habitats. The effects of physical barriers on salmonids include: loss of spawning habitat for adults, and for juveniles the inability to reach overwintering sites or thermal refugia, the loss of summer rearing habitat, and increased vulnerability to predation (PSMFC, 1999).

The purpose of this analysis is to: (1) identify how many stream kilometers are blocked by anthropogenic barriers; (2) determine the worse-case scenario, in which we assume that all anthropogenic features that have an unknown barrier status are complete barriers; and (3) quantify the percentage of low, medium, and high quality habitat blocked by anthropogenic barriers using the results of a historical intrinsic production potential analysis, which used methods described in Appendix C. This analysis was performed on major independent populations of chinook and steelhead for endangered species act (ESA) listed ESUs within the Interior Columbia Basin, as defined by Interior Columbia Basin Technical Recovery Team (2003).

A word of caution: this analysis was based upon readily available statewide data sets. The quality and the extent of data varied across states. For example, blockage extents for dams in Idaho were not available.

Approach and Methods:

Statewide datasets containing information on anthropogenic barriers to salmon migration were obtained for Washington (WDFW, 2003), Oregon (ODFW, 2003), and Idaho (IDFW, 2003). This analysis utilized readily available statewide data only. Local datasets containing additional barriers are available in some locations and a comparative analysis including local data is forthcoming.

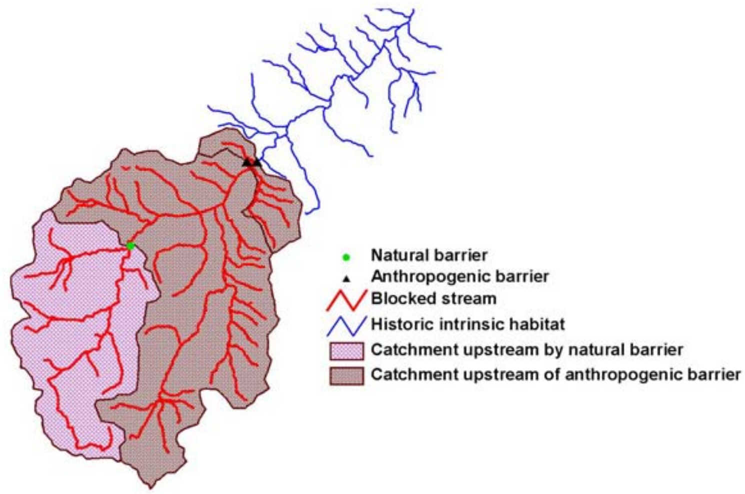
The datasets for the three states were merged and records for barriers lying within the Interior Columbia River Basin were selected, creating a new dataset containing only these barriers. In a number of instances, values for codes in the datasets, such as blockage type and blockage extent, differed for each state. In these cases, codes were standardized.

Structural anthropogenic barriers were analyzed in this study to quantify the amount for stream habitat currently blocked in each independent population of chinook and steelhead in the Interior Columbia River Basin. Structural anthropogenic barriers were recorded as partial, complete, or unknown for blockage extent. The number of potentially usable stream kilometers that are currently blocked by complete barriers was determined (i.e., those barriers in which no fish can pass). We also analyzed a "worst-case scenario," in which we assumed all of the barriers of unknown blockage extent were complete barriers. Where current fish distribution data show fish populations above unknown barriers, unknown barriers were coded as partial barriers. No analysis using the partial barriers was conducted at this time.

In order to develop an estimate of how much stream habitat is being blocked by anthropogenic barriers, ESRI ArcView 3.2 and Spatial Analyst were used. A customized script and a 10 m digital elevation model (DEM) enabled us to delineate the catchments upstream of each barrier. An ArcView shapefile with polygons representing the catchments above each barrier was created. Essentially, these polygons define the regions that are inaccessible to fish due to the presence of barriers.

The catchments upstream of the natural barriers (i.e., falls and gradients only) were intersected with a GIS layer containing 200 m segments representing historic salmon habitat in the Columbia Basin assumed to be usable by an analysis using methods described in Appendix C. Those 200 m segments that were at least 50 percent within a catchment upstream of a natural barrier were recoded as non-usable habitat and eliminated from further analysis. Then, the stream layer was intersected in the same manner with the catchments above the anthropogenic complete barriers. The 200 m segments that were at least 50 percent within the catchments above the anthropogenic complete barriers were tallied to provide the length of stream blocked. The number of stream kilometers blocked in each distinct population was then computed. Figure 1 demonstrates the basic approach.

Figure 1. Example of how stream length blocked by natural and anthropogenic barriers was computed.



Each 200 m segment of stream in the GIS layer has a habitat quality rating consistent with the "capacity metric" in fish population status (i.e., low = 0.25, medium = 0.5, and high = 1.0). We present our results as total stream kilometers blocked, stream kilometers blocked reduced by the weighting factors shown, and percentages of total stream kilometers blocked for each independent chinook and steelhead population.

Results:

Anthropogenic barriers in the Interior Columbia River Basin were found to completely block varying extents of chinook and steelhead habitat in several major population groupings. The worst-case scenarios using statewide sources increased the blockage extent of the complete barriers in several major population groupings. See Table 1 for chinook data and Table 2 for steelhead data.

Relatively few stream kilometers were completely blocked to chinook populations (see Figures 2). Of the 35 independent chinook populations in the Interior Columbia Basin, 20 have no identified complete barriers. The highest percentage of completely blocked stream kilometers for the "known" analysis using statewide data occurs in the Upper Columbia Entiat River (UCENT) with 43.41 percent (141.82 km) of the stream completely blocked. When the habitat weighting factors are implemented, the estimate of completely blocked habitat in this population increases to 58.10 percent (31.92 km) of total weighted habitat. For the worst-case scenario, the Upper Columbia Wenatchee River (UCWEN) has the highest percentage of unweighted stream kilometers completely blocked (44.05 percent). Catherine Creek in the Grande Ronde had the highest percentage change in the worst-case scenario, with a 9.12 percent increase in completely blocked weighted stream habitat.

A greater percentage of stream kilometers were completely blocked to steelhead populations (see Figures 3). Of the 47 independent steelhead populations in the Interior Columbia Basin, 18 have no identified complete barriers. The highest percentage of completely blocked stream kilometers for the “known” analysis using statewide data occurs in the Snake River North Fork Clearwater steelhead population (CRNFC-s) and the Middle Columbia White Salmon River (MCWSA-s) with 100 percent the stream habitat completely blocked to these populations. When the habitat weighting factors are implemented, there is little change in the percentage of kilometers blocked (see Table 1 and 2). Note that CRNFC-s and MCWSA-s are considered historic populations. The Upper Columbia Wenatchee (UCWEN-s) and Entiat Rivers (UCENT-s) have the greatest proportion of stream kilometers completely blocked, when considering current populations and disregarding those that are historic. For the worst-case scenario using statewide data, UCWEN-s and the Middle Columbia Umatilla River (MCUMA-s) have the greatest proportion of weighted stream kilometers completely blocked when disregarding historic populations.

Table 1. Current and Worst Case Chinook Stream Kilometers Blocked by Anthropogenic Structures and Weighted by Habitat Quality

Major Population Grouping	Current Population Code	Population Name	Historical Length Available (km)	Current Length Blocked (km)	Worst Case Length Blocked (km)	Historical Length Available Weighted by Habitat Quality (km)	Current Length Blocked Weighted by Habitat Quality (km)	Worst Case Length Blocked Weighted by Habitat Quality (km)	Percent Current Length Blocked (km)	Percent Current Length Blocked Weighted by Habitat Quality (km)	Percent Worst Case Length Blocked (km)	Percent Worst Case Length Blocked Weighted by Habitat Quality (km)
Lower Snake River												
	SNASO	Asotin River	627.48	0.00	0.00	42.60	0.00	0.00	0.00	0.00	0.00	0.00
	SNTUC	Tucannon River	982.78	0.00	0.00	146.50	0.00	0.00	0.00	0.00	0.00	0.00
Grande Ronde / Imnaha												
	GRWEN	Wenaha River	369.71	0.00	0.00	58.18	0.00	0.00	0.00	0.00	0.00	0.00
	GRLOS	Wallowa/Lostine Rivers	786.16	9.14	9.14	212.29	6.44	6.44	1.16	3.03	1.16	3.03
	GRLOO	*Lookingglass Creek (Historic)	128.30	8.09	16.04	21.98	0.00	0.00	6.31	0.00	12.50	0.00
	GRMIN	Minam River	181.92			64.32			0.00	0.00	0.00	0.00
	GRCAT	Catherine Creek	497.53	79.45	113.42	165.78	4.85	19.98	15.97	2.93	22.80	12.05
	GRUMA	Upper Grande Ronde River	1,525.90	1.55	54.23	267.40	0.00	2.75	0.10	0.00	3.55	1.03
	IRMAI	Imnaha River	576.75	0.00	0.00	98.86	0.00	0.00	0.00	0.00	0.00	0.00
	IRBSH	Big Sheep Creek	450.03	0.00	0.00	53.00	0.00	0.00	0.00	0.00	0.00	0.00
South Fork Salmon River												
	SRLSR	Little Salmon River	526.06	3.40	3.40	86.35	0.00	0.00	0.65	0.00	0.65	0.00
	SFMAI	South Fork Salmon River	779.34	0.00	0.00	165.35	0.00	0.00	0.00	0.00	0.00	0.00
	SFSEC	Secesh River	250.32	0.00	0.00	0.00	0.00	0.00	0.00		0.00	
	SFEFS	E Fk S Fk Salmon River	223.65	0.00	0.00	74.41	0.00	0.00	0.00	0.00	0.00	0.00
Middle Fork Salmon River												
	SRCHA	Chamberlain Creek	417.91	0.00	0.00	98.01	0.00	0.00	0.00	0.00	0.00	0.00
	MFBIG	Big Creek	667.37	0.00	0.00	144.96	0.00	0.00	0.00	0.00	0.00	0.00
	MFLMA	Lower Middle Fork Salmon River	228.84	0.00	0.00	32.66	0.00	0.00	0.00	0.00	0.00	0.00
	MFCAM	Camas Creek	294.85	40.90	42.11	61.84	9.71	10.16	13.87	15.70	14.28	16.43
	MFLOO	Loon Creek	262.06	0.00	0.80	53.99	0.00	0.00	0.00	0.00	0.31	0.00
	MFUMA	Upper Middle Fork Salmon River	559.24	0.00	0.00	130.85	0.00	0.00	0.00	0.00	0.00	0.00
	MFSUL	Sulphur Creek	53.33	0.00	0.00	20.12	0.00	0.00	0.00	0.00	0.00	0.00
	MFBEA	Bear Valley Creek	217.52	0.00	0.00	131.08	0.00	0.00	0.00	0.00	0.00	0.00

	MFMAR	Marsh Creek	187.99	0.00	0.00	85.22	0.00	0.00	0.00	0.00	0.00	0.00
Upper Salmon River												
	SRPAN	Panther Creek (Historic)	470.15	23.50	30.05	88.10	1.40	1.60	5.00	1.59	6.39	1.82
	SRNFS	N Fk Salmon River	338.26	46.88	46.88	56.43	0.65	0.65	13.86	1.15	13.86	1.15
	SRLEM	Lemhi River	1,028.70	15.37	70.48	293.94	0.65	8.42	1.49	0.22	6.85	2.86
	SRLMA	Lower Salmon River	1,018.75	33.85	84.28	199.16	1.10	4.80	3.32	0.55	8.27	2.41
	SRPAH	Pahsimeroi River	411.83	0.00	0.00	181.90	0.00	0.00	0.00	0.00	0.00	0.00
	SREFS	E Fk Salmon River	292.09	0.00	0.00	75.49	0.00	0.00	0.00	0.00	0.00	0.00
	SRYFS	Yankee Fork	180.89	0.00	0.00	49.37	0.00	0.00	0.00	0.00	0.00	0.00
	SRVAL	Valley Creek	176.78	5.41	5.41	87.51	4.91	4.91	3.06	5.61	3.06	5.61
	SRUMA	Upper Salmon River	307.64	0.00	0.00	157.54	0.00	0.00	0.00	0.00	0.00	0.00
Upper Columbia												
	UCWEN	Wenatchee River	1,223.99	481.95	539.17	301.43	86.03	93.09	39.38	28.54	44.05	30.88
	UCENT	Entiat River	326.67	141.82	141.82	54.94	31.92	31.92	43.41	58.10	43.41	58.10
	UCMET	Methow River	1,358.96	454.07	454.07	226.66	71.69	71.69	33.41	31.63	33.41	31.63

Table 2. Current and Worst Case Steelhead Stream Kilometers Blocked by Anthropogenic Structures and Weighted by Habitat Quality

Major Population Grouping	Current Population Code	Population Name	Historical Length Available (km)	Current Length Blocked (km)	Worst Case Length Blocked (km)	Historical Length Available Weighted by Habitat Quality (km)	Current Length Blocked Weighted by Habitat Quality (km)	Worst Case Length Blocked Weighted by Habitat Quality (km)	Percent Current Length Blocked (km)	Percent Current Length Blocked Weighted by Habitat Quality (km)	Percent Worst Case Length Blocked (km)	Percent Worst Case Length Blocked Weighted by Habitat Quality (km)
Cascade Eastern Slope Tributaries												
	MCWSA-s	White Salmon River (Historic)	406.40	399.19	399.19	286.74	284.33	284.33	98.23	99.16	98.23	69.96
	MCKLI-s	Klickitat River	1,956.36	30.81	158.50	1,418.13	23.98	113.81	1.57	1.69	8.10	5.82
	MCFIF-s	Fifteen Mile Creek (winters)	1,084.46	37.96	38.45	915.10	32.46	32.62	3.50	3.55	3.55	3.01
	DREST-s	Deschutes River, Eastside	2,362.67	185.28	204.15	1,807.05	155.24	169.76	7.84	8.59	8.64	7.19
	DRWST-s	Deschutes River, Westside	1,142.12	0.00	35.22	831.68	0.00	31.04	0.00	0.00	3.08	2.72
	MCROC-s	Rock Creek	361.31	0.00	0.00	300.12	0.00	0.00	0.00	0.00	0.00	0.00
John Day River												
	JDLMT-s	John Day River lower mainstem	6,001.49	42.11	218.69	4,685.85	36.36	180.57	0.70	0.78	3.64	3.01
	JDNFJ-s	North Fork John Day River	3,207.20	24.76	29.83	2,518.12	20.83	24.48	0.77	0.83	0.93	0.76
	JDMFJ-s	Middle Fork John Day River	1,490.48	0.00	0.00	1,151.18	0.00	0.00	0.00	0.00	0.00	0.00
	JDSFJ-s	South Fork John Day River	1,084.41	0.00	10.27	848.60	0.00	6.86	0.00	0.00	0.95	0.63
	JDUMA-s	John Day upper mainstem	1,539.17	35.84	37.66	1,184.08	26.79	27.40	2.33	2.26	2.45	1.78
Umatilla and Walla Walla Rivers												
	MCUMA-s	Umatilla River	3,060.00	1,161.96	1,162.36	2,346.94	953.35	953.75	37.97	40.62	37.99	31.17
	WWMAI-s	Walla Walla River	1,729.51	27.44	126.67	1,207.22	22.83	104.91	1.59	1.89	7.32	6.07
	WWTOU-s	Touchet River	1,370.95	0.00	0.00	1,011.12	0.00	0.00	0.00	0.00	0.00	0.00
Yakima River Group												
	YRTOS-s	Toppenish and Satus Creeks	1,831.26	65.83	79.58	1,412.21	49.00	56.62	3.59	3.47	4.35	3.09
	YRNAC-s	Naches River	1,611.13	305.71	340.83	1,046.15	182.12	206.95	18.97	17.41	21.15	12.85
	YRUMA-s	Yakima River upper mainstem	3,157.57	786.06	786.46	2,247.27	537.97	538.37	24.89	23.94	24.91	17.05
Lower Snake												

	SNTUC-s	Tucannon River	1,033.41	0.00	0.00	780.64	0.00	0.00	0.00	0.00	0.00	0.00
	SNASO-s	Asotin Creek	1,892.34	17.04	17.04	1,486.66	13.76	13.76	0.90	0.93	0.90	0.73
Clearwater River												
	CRLMA-s	Clearwater lower mainstem	1,742.44	1.85	28.69	1,189.39	1.65	17.98	0.11	0.14	1.65	1.03
	CRNFC-s	North Fork Clearwater (Historic)	2,335.63	2,335.63	2,335.63	1,571.91	1,571.91	1,571.91	100.00	100.00	100.00	67.30
	CRLOL-s	Lolo Creek	345.44	0.00	0.00	229.84	0.00	0.00	0.00	0.00	0.00	0.00
	CRLOC-s	Lochsa River	1,015.42	0.55	0.55	685.81	0.40	0.40	0.05	0.06	0.05	0.04
	CRSEL-s	Selway River	1,335.53	0.00	0.00	910.79	0.00	0.00	0.00	0.00	0.00	0.00
	CRSFC-s	South Fork Clearwater River	960.96	0.00	0.00	682.91	0.00	0.00	0.00	0.00	0.00	0.00
Grande Ronde River												
	GRLMT-s	Grande Ronde lower mainstem tribs	1,393.94	0.00	7.49	980.58	0.00	4.76	0.00	0.00	0.54	0.34
	GRJOS-s	Joseph Creek	844.29	0.00	0.00	623.94	0.00	0.00	0.00	0.00	0.00	0.00
	GRWAL-s	Wallowa River	968.09	8.94	9.14	697.10	4.28	4.48	0.92	0.61	0.94	0.46
	GRUMA-s	Grande Ronde upper mainstem	2,696.78	92.31	199.90	2,000.19	69.93	135.77	3.42	3.50	7.41	5.03
Salmon River												
	SRLSR-s	Little Salmon and Rapid Rivers	895.38	3.20	3.20	519.85	1.85	1.85	0.36	0.36	0.36	0.21
	SRCHA-s	Chamberlain Creek	734.08	0.00	0.00	462.24	0.00	0.00	0.00	0.00	0.00	0.00
	SFSEC-s	Secesh River	250.13	0.00	0.00	175.92	0.00	0.00	0.00	0.00	0.00	0.00
	SFSFS-s	South Fork Salmon River	643.40	0.00	0.00	443.97	0.00	0.00	0.00	0.00	0.00	0.00
	SRPAN-s	Panther Creek	657.37	23.09	30.04	461.59	17.83	22.68	3.51	3.86	4.57	3.45
	MFBIG-s	Big, Camas, and Loon Creeks	1,165.63	40.71	42.91	800.13	29.35	30.54	3.49	3.67	3.68	2.62
	MFUMA-s	Middle Fork Salmon River upper mainstem	1,116.76	0.00	0.00	718.38	0.00	0.00	0.00	0.00	0.00	0.00
	SRNFS-s	North Fork Salmon River	338.46	45.69	46.69	222.60	29.31	29.96	13.50	13.17	13.79	8.85
	SRLEM-s	Lemhi River	1,028.51	15.17	70.47	715.43	12.87	53.77	1.47	1.80	6.85	5.23
	SRPAH-s	Pahsimeroi River	648.10	33.44	33.64	426.91	25.13	25.33	5.16	5.89	5.19	3.91
	SREFS-s	East Fork Salmon River	709.21	0.00	21.15	501.93	0.00	16.87	0.00	0.00	2.98	2.38
	SRUMA-s	Salmon River upper mainstem	1,021.05	5.20	34.89	674.52	2.00	20.48	0.51	0.30	3.42	2.01
Hells Canyon												
	SNHCT-s	Snake River Hells Canyon tribs	335.56	0.00	0.00	163.41	0.00	0.00	0.00	0.00	0.00	0.00
Imnaha River												
	IRMAI-s	Imnaha River	1,026.79	0.00	0.00	748.26	0.00	0.00	0.00	0.00	0.00	0.00
Upper Columbia												
	UCWEN-s	Wenatchee River	1,219.38	479.76	537.36	806.20	346.60	383.78	39.34	42.99	44.07	31.47
	UCENT-s	Entiat River	326.68	141.62	141.62	210.30	90.96	90.96	43.35	43.25	43.35	27.84
	UCMET-s	Methow River	1,358.38	452.46	452.66	892.87	316.26	316.46	33.31	35.42	33.32	23.30
	UCOKA-s	Okanogan River	1,176.25	401.80	401.80	802.36	277.15	277.15	34.16	34.54	34.16	23.56

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Table 1. Candidate habitat factors.

Candidate habitat factor	Analysis priority
Landscape processes that form and sustain aquatic habitats	
Barriers to passage	1
Irrigation diversions	1
Flows and water withdrawals	1
Sediment	1
Riparian condition	1
Floodplain interactions (e.g. channelization, off-channel habitat)	1
Fire regime	3
Habitat conditions influenced by landscape processes	
Physical habitat (e.g., pool spacing)	2
Water quality	2
Stream temperature	3
Trophic interactions (e.g., nutrient cycling)	3
Exotic species	3
Predator/competitor interactions	3

Table 2. Regional differences in dominant ecosystem processes or functions in the Pacific Northwest. This table is intended only to illustrate that different processes and assessments should be emphasized in different ecoregions. Important ecosystem processes vary within ecoregions, and watershed-level assessments should target those processes that are locally important within each watershed. (Note that the Columbia River estuary is in the coastal forests ecoregion, but also affects Columbia River stocks in the Western deserts and Western forested mountains.)

Watershed process or function	**Level II ecoregion**		
	Western deserts	Western forested mountains	Coastal forests
Sediment supply	Gullying and surface erosion (especially in agricultural areas)	Mass wasting and gullying	Mass wasting (surface erosion in agricultural lowlands)
Riparian functions	Grasses and shrubs, some forest in floodplains	Sparse forests, shade a dominant function	Dense forests, wood recruitment a dominant function
Floodplain functions	Channel incision common, dikes common	Infrequent channel incision, dikes common	Rare channel incision, dikes common
Habitat connectivity	Culverts, dams, and dikes common; incision and floodplain abandonment common	Culverts, dams, and dikes common	Culverts, dams, and dikes common
Low flow hydrology	Diversions and dams common	Some diversions, dams common	Few diversions, dams relatively common
Flood hydrology	Snowmelt dominated flood regime	Snowmelt dominated flood regime	Rain and rain-on-snow flood regime

Table 3. Possible functions of riparian vegetation. Importance of each function may differ among regions.

General Category	**Specific Riparian Function**
Bank Stability/ Channel morphology	Stabilize stream banks (root strength)
	Help sustain natural channel morphology (prevent channel widening or incision)
	Contribute to habitat complexity (undercut banks, percent pools)
Temperature Control	Maintain stable temperature regime
	Decreased stream temperatures via shading
Organic Matter Supply/Habitat complexity	Provide large woody debris cover
	Provide organic carbon and nutrients to support the aquatic food web
Filtering Capacity	Filter sediment input
	Reduce pollutants and filter runoff, including nutrients (N & P)
	Improve air quality and lower ozone levels
Hydrology Related	Help reduce the severity of floods; maintain stable water flows
	Increase channel-floodplain connections (facilitate exchange of ground- and surface-water)
Other Functions	Provide critical wildlife habitat

Table 4. Example of summary from step one of the riparian coarse scale analysis (data entirely made-up).

HUC6	Stream reach	Stream length (m)	%Forested	%Grass-shrub	%Agric./urban	Length converted	%Converted
1	1	120	56	24	20	24	
1	2	657	12	59	29	190.53	
1	3	246	0	15	85	209.1	
1	4	123	45	0	55	67.65	
1	5	57	0	0	100	57	
summary		**1203**	**113**	**98**	**289**	**548.3**	**45.6**
2	1	168	9	68	23	38.64	
2	2	354	15	8	77	272.58	
2	3	387	19	75	6	23.22	
2	4	129	50	0	50	64.5	
2	5	456	20	0	80	364.8	
summary		**1494**	**113**	**151**	**236**	**763.7**	**51.1**

Table 5. Example datasheet and analysis for three aerial photos. We include up to 4 categories (only 2 shown here for example). NHI = vegetation code used in the Northwest Habitat Institute data layer. "End" refers to any unnatural habitat run into within the 100m transect (e.g., agriculture, urban, road).

		aerial photo L					**aerial photo R**						
Grid ID	**Transect**	**category1**	**dist1 (m)**	**category2**	**dist2 (m)**	**end**	**category**	**dist1 (m)**	**category2**	**dist2 (m)**	**end**	**NHI**	**Comments**
18449	1	mixed	100				forest	21	shrub	79		15	
	2	forest	30	mixed	70		forest	64	mixed	36			
	3	mixed	47	grass	53		forest	37	grass	63			
	4	mixed	43	forest	57		mixed	65	grass	35			
	5	mixed	100				forest	29	grass	71			few trees @ edge
	6	forest	100				forest	47	grass	53			
	7	mixed	100				forest	43	grass	57			
	8	forest	35	mixed	65		forest	39	grass	61			
	9	forest	58	grass	42		forest	68	shrub	32			
	10	forest	100				forest	48	shrub	52			
24116	1	mixed	30			road	forest	20	mixed	80		5	highly shadowed
	2	mixed	26			road	forest	29	mixed	71			
	3	shrub	30			road	mixed	33	shrub	67			
	4	mixed	36			road	mixed	36	shrub	64			
	5	mixed	29	grass	8	road	forest	30	shrub	70			
	6	forest	29	grass	15	road	mixed	27	grass	63			
	7	forest	12	grass	32	road	forest	22	mixed	78			
	8	mixed	21			road	mixed	13	shrub	87			
	9	mixed	9			road	forest	19	shrub	81			included rock in road (RB)
	10	grass	26			road	forest	21	mixed	79			
21626	1	mixed	100				mixed	100				7	
	2	mixed	100				mixed	100					
	3	grass	46	mixed	54		mixed	100					
	4	mixed	100				forest	100					
	5	mixed	100				mixed	100					
	6	mixed	100				shrub	100					
	7	mixed	100				shrub	53	grass	47			
	8	forest	100				shrub	50	grass	50			
	9	forest	100				shrub	44	grass	66			
	10	forest	100				mixed	100					

Table 6. Summary of comparisons of data analyzed with 10 and 5 transects per photo, and the corresponding p-value for each test (α=0.05). For nearest and dominant vegetation categories, numbers shown represent the number of occurrences in each of the following categories, respectively: forest, mixed, shrub, grass, other, and none.

Stratum	**Summary data compared**	**10 transects**	**5 transects**	**P-value**
Agriculture	Mean buffer width	47.3 (33.4)	47.0 (34.2)	0.9806
	Nearest vegetation	3,2,2,8,1,4	3,2,3,7,1,4	1.00
	Dominant vegetation	3,2,1,9,1,4	3,2,1,9,1,4	1.00
Forested	Mean buffer width	86.5 (20.1)	87.1 (18.6)	0.9505
	Nearest vegetation	4,5,0,0,1,0	4,5,0,0,1,0	1.00
	Dominant vegetation	4,5,0,0,1,0	5,5,0,0,0,1	1.00
Shrub/Grass	Mean buffer width	83.6 (19.3)	82.7 (20.6)	0.9154
	Nearest vegetation	4,1,1,4,0,0	3,2,1,4,0,0	1.00
	Dominant vegetation	2,2,3,3,0,0	2,1,3,4,0,0	1.00

Table 7. Shapiro-Wilks Normality Test of Average Buffer Widths.

Zone	NHI Habitat Stratum	P -Value	Photo Habitat Stratum	P -Value
Riparian	Agriculture/Urban	**0.007670**	Agriculture/Urban	**0.02898**
	Forest/Mixed	**0.0001012**	Forest/Mixed	**6.98×10^{-5}**
	Shrub/Grass	**0.0002796**	Shrub/Grass	**0.0004901**
Floodplain	Agriculture/Urban	**0.0001917**	Agriculture/Urban	**0.002569**
	Forest/Mixed	**6.79×10^{-5}**	Forest/Mixed	**0.001204**
	Shrub/Grass	**0.002878**	Shrub/Grass	**0.0008857**

All bold P-Values indicate significance at $\alpha = 0.05$.

Table 8. Heterogeneity of Variance Test of Average Buffer Widths.

Zone	Habitat Classification	Bartlett's Test P -Value	Habitat Classification	Fligner's Test P -Value
Riparian	NHI Data	0.1090	NHI Data	0.1236
	Photo	0.1901	Photo	0.5873
Floodplain	NHI Data	**0.004607**	NHI Data	**0.009278**
	Photo	**0.0002700**	Photo	**0.0001278**

All bold P-Values indicate significance at $\alpha = 0.05$.

Table 9. Error matrix comparing NHI layer-predicted habitat types to the dominant habitat type assessed in aerial photos analyzed for the (A) Non-floodplain and (B) Floodplain areas.

A. Non-floodplain

	Photo							
NHI	**agriculture**	**forest**	**grass**	**shrub**	**urban**	**row total**	**% correct**	**% comission**
agriculture	28	4	12	5		49	57.1	42.9
forest	1	27		3		31	87.1	12.9
grass		1	3			4	75.0	25.0
shrub	2	1	3	15		21	71.4	28.6
urban	1		1		5	7	71.4	28.6
column total	32	33	19	23	5	112	69.6	30.4
% omission	12.5	18.2	84.2	34.8	0.0			

B. Floodplain

	Photo							
NHI	**agriculture**	**forest**	**grass**	**shrub**	**urban**	**row total**	**% correct**	**% comission**
agriculture	43	14	5	7	1	70	61.4	38.6
forest	2	25	3	10	1	41	61.0	39.0
grass	1	1	4	1		7	57.1	42.9
shrub	3		5	15		23	65.2	34.8
urban			2		3	5	60.0	40.0
column total	49	40	19	33	5	146	61.6	38.4
% omission	12.2	37.5	78.9	54.5	40.0			

Table 10. Summary of aerial photo analyses in the Columbia River Basin for riparian buffers in non-floodplain areas (A), and in floodplain areas (B). Habitat classification in strata is based on the NHI data layer in the left column, and on dominant habitat type assessed in aerial photos in the right column. Frequency of photos having a given vegetation type nearest the stream ("near") or dominant vegetation type within the 100m buffer ("dom").

A. Non-Floodplain Areas

	Strata based on NHI layer			**Strata based on aerial photos**		
	agriculture /urban	**forested**	**shrub /grass**	**agriculture /urban**	**forested**	**shrub /grass**
buffer width						
n	56	31	25	37	33	42
med	32.9	92.2	84.1	24.7	92.2	73.5
mean	39.7	81.7	79.1	26.5	82.2	72.4
stdev	28.9	20.3	25.1	20.4	20.1	26.1

vegetation type	*near*	*dom*	*near*	*dom*	*near*	*dom*	*near*	*dom*	*near*	*dom*	*near*	*dom*
forest	23	17	19	17	5	4	12	9	23	22	12	7
mixed	6	7	8	8	3	1	3	3	6	9	8	4
shrub	11	11	2	5	12	12	8	8	1	1	16	19
grass	6	11	2	1	4	7	4	7	3	1	5	11
other	0	0	0	0	0	0	0	0	0	0	0	0
none	10	10	0	0	1	1	10	10	0	0	1	1

B. Floodplain Areas

	Strata based on NHI layer			**Strata based on aerial photos**		
	agriculture /urban	**forested**	**shrub /grass**	**agriculture /urban**	**forested**	**shrub /grass**
buffer width						
n	75	41	30	54	40	52
med	30.0	81.9	66.4	16.1	80.3	69.2
mean	35.4	82.2	62.0	19.7	80.5	69.3
stdev	27.7	19.1	33.3	15.5	17.4	26.4

vegetation type	*near*	*dom*	*near*	*dom*	*near*	*dom*	*near*	*dom*	*near*	*dom*	*near*	*dom*
forest	14	18	14	15	5	2	9	9	22	25	2	1
mixed	24	19	13	13	6	3	16	13	15	15	12	7
shrub	22	15	11	9	16	16	20	10	2	0	27	30
grass	11	9	3	4	3	6	6	6	1	0	10	13
other	1	0	0	0	0	0	0	0	0	0	1	0
none	3	14	0	0	0	3	3	16	0	0	0	1

Table 11. Kruskal-Wallis rank sum test of average buffer widths.

Zone	Habitat Classification	Kruskal-Wallis P-Value
Non-floodplain	NHI Data	**2.20×10^{-16}**
	Photo	**4.55×10^{-13}**
Floodplain	NHI Data	**4.62×10^{-12}**
	Photo	**2.20×10^{-16}**

All bold P-Values indicate significance at $\alpha = 0.05$.

Table 12. Pairwise Wilcoxon rank sum test of average buffer widths.

Zone	Habitat Type Comparison	By NHI P - Value	By Photo P - Value
Non-floodplain	Agriculture/Urban vs. Forest/Mixed	**2.5×10^{-8}**	**7.1×10^{-11}**
	Agriculture/Urban vs. Shrub/Grass	**1.1×10^{-6}**	**1.4×10^{-9}**
	Forest/Mixed vs. Shrub/Grass	9.0×10^{-1}	1.4×10^{-1}
Floodplain	Agriculture/Urban vs. Forest/Mixed	**5.2×10^{-12}**	**6.7×10^{-16}**
	Agriculture/Urban vs. Shrub/Grass	**8.1×10^{-4}**	**1.7×10^{-14}**
	Forest/Mixed vs. Shrub/Grass	**7.1×10^{-3}**	5.3×10^{-2}

All bold P-Values indicate significance at $\alpha = 0.05$.

Table 13. Erosion weighting factors for soil erosivity (based on K values), slope, and land cover factors for the erosion rate index.

Input value	Erosion weighting factor	Explanation
Soil erosivity (ICBEMP:bvbsoil)		
Low - <33% of HUC with K>0.37	1	Simple linear weighting of 6th field HUCs based on proportion of soils with K>0.37.
Moderate - 33-66% of HUC with K>0.37	2	
High - >66% of HUC with K>0.37	3	
Slope (in percent)		
Slope for grid cell from DEM	$I_{slope} = 3.41(S^2) + 0.93(S)$	Equation based on sensitivity analysis of erosion rate to S in the USLE.
Land cover factor (ICBEMP: H2vgO for historical, S1vgO for current)		
Historical - grasses	1	Selected as the normative case for non-forest lands (typical of the Palouse region). Value of C is about 0.01 (Dunne and Leopold 1978, p. 529, Table 15-4, 80% ground cover).
Historical - shrub	4	Shrub lands value of C is about 0.04 (Dunne and Leopold 1978, p. 529, Table 15-4), so rating is 4 times the value of that for grasses.
Agriculture – historically grasses or forest	10	Ag practices on average increase erosion rates by about a factor of one order of magnitude (but with lots of variation), based on Table 15.2 (Dunne and Leopold 1978).
Agriculture – historically shrub	12	Historical rate for shrub was four times higher than for grasses. Ag increases erosion by a factor of about 3.
Rock, water, alpine, and all forest types (see Table 13 for forest classes)	0	No erosion from these surfaces for this rating system.

Table 14. Land cover classes.

Vegetation classes	Interpretation
Cool shrub	Non-forest, shrub
Dry grass	Non-forest, grass
Dry shrub	Non-forest, shrub
Riparian shrub	Non-forest, shrub
Cold forest	Forest
Dry forest	Forest
Moist forest	Forest
Riparian woodland	Forest
Woodland	Forest
Water	Other
Alpine	Other
Rock	Other

Table 15. Erosion ratings for various combinations of hill slope, soil erosivity, and land cover class.

	Hill slope (m/m)					
Soil erosivity (by land cover class)	<.02	.02-.04	.04-.08	.08-.16	.16-.32	>0.32
Natural -grasses						
low - <33% of HUC with K>0.37	0.01	0.03	0.055	0.15	0.45	0.7
moderate - 33-66% of HUC with K>0.37	0.02	0.06	0.11	0.3	0.9	1.4
high - <66% of HUC with K>0.37	0.03	0.09	0.165	0.45	1.35	2.1
Natural - shrub						
low - <33% of HUC with K>0.37	0.04	0.12	0.22	0.6	1.8	2.8
moderate - 33-66% of HUC with K>0.37	0.08	0.24	0.44	1.2	3.6	5.6
high - <66% of HUC with K>0.37	0.12	0.36	0.66	1.8	5.4	8.4
Agriculture - historically grasses						
low - <33% of HUC with K>0.37	0.1	0.3	0.55	1.5	4.5	7
moderate - 33-66% of HUC with K>0.37	0.2	0.6	1.1	3	9	14
high - <66% of HUC with K>0.37	0.3	0.9	1.65	4.5	13.5	21
Agriculture - historically brush						
low - <33% of HUC with K>0.37	0.12	0.36	0.66	1.8	5.4	8.4
moderate - 33-66% of HUC with K>0.37	0.24	0.72	1.32	3.6	10.8	16.8
high - <66% of HUC with K>0.37	0.36	1.08	1.98	5.4	16.2	25.2

Table 16. Summary of sediment budgets for the interior and lower Columbia River basin, and weighting factors used in our analysis. Because data are sparse, weighting factors were chosen to represent the order of magnitude change in sediment supply, and are not used predict actual volumes of sediment supplied to streams.

		Sediment supply ($m^3/km^2/yr$) (factor increase over mature forest rate)			
Location	Years of record	Mature forest	Clear cut	Road	Reference
Idaho batholith	2	7	138 (20)	1315 (188)	Gray and Megahan 1981
Idaho batholith	1-8	6.3[a] (1)	na	na	Megahan 1975
Idaho batholith	6	21	3193 (152)	11316 (539)	Megahan 1975
Idaho batholith	?	7-21[b]	51-65[b] (3-8)		Morgan and Smith 1997
Oregon Cascade Range	25	87	245 (2.8)	2619 (30)	Swanson and Dyrness 1975
Factors used in our analysis		1 (eastside) 3 (westside)	10	100	

a. Value converted from $yd^3/mi^2/yr$ (in original reference) to $m^3/km^2/yr$.
b. Values converted from $t/km^2/yr$ using colluvium bulk density of 1500 kg/m^3 (Meyer *et al.* 2001).

Table 17. Summary of annual clearcut rates in Washington forests by ownership class for selected WRIAs, and median value for all WRIAs (Collins 1996). For our analyses we selected the overall median value (gray highlights at bottom) to represent harvest rates by ownership class, and from that value calculated average percentage of stands less than 20 years old (classified as clearcuts) in any one year.

	Private	Federal	State	Other
Eastern CRB				
Walla Walla (32)	0.03	0.01	0	0
Lower Snake (33)	0.00	N/A	N/A	N/A
Palouse (34)	0	0	0	N/A
Middle Snake (35)	.01	.16	0	N/A
Mean	.01	.06	0	0
Mean % clearcut in any one year	.2%	1.2%	0	0
Western CRB				
Naches (38)	.75	.51	.05	0
Upper Yakima (39)	.8	.17	.23	0
Wenatchee (*45)	.16	.11	0	N/A
Entiat (46)	.08	.39	0	N/A
Chelan (47)	.02	.13	0	N/A
Methow (48)	.1	.21	.0	N/A
Mean	.32	.25	.05	0
Mean % clearcut in any one year	6.4%	5.0%	1%	0
Median value for all eastern Washington WRIAs	.3	.03	.01	0
Mean % clearcut in any one year	6%	.6%	.2%	0
Western Washington				
Willapa (24)	1.52	N/A	.60	N/A
Grays-Elokoman (25)	1.83	N/A	.62	N/A
Cowlitz (26)	1.83	.23	.36	N/A
Lewis (27)	1.61	.28	.41	N/A
Salmon-Washougal (28)	1.31	0	.23	N/A
Mean	1.62	0.17	.44	N/A
Mean % clearcut in any one year	32%	3.4%	8.9%	
Median value for all western Washington WRIAs	1.48	.14	.64	0
Mean % clearcut in any one year	30%	2.8%	12.8 %	0

Table 18. Rating for potential exposure to pesticides or urban runoff by land cover class based on USGS Land Use Land Cover data.

Code	Land cover classification	Rating
11	Open water	1
12	Perennial ice/snow	1
31	Bare rock/sand/clay	1
33	Transitional	1
41	Deciduous forest	1
42	Coniferous forest	1
43	Mixed forest	1
51	Shrubland	1
71	Grasslands/herbaceous	1
91	Woody wetlands	1
92	Emergent herbaceous wetlands	1
81	Pasture/hay	2
83	Small grains	2
84	Fallow	2
21	Low intensity residential	10
22	High intensity residential	10
23	Commercial/industrial/transportation	10
32	Quarries/strip mines/gravel pits	10
61	Orchards/vineyards/other	10
82	Row crops	10
85	Urban/recreational grasses	10

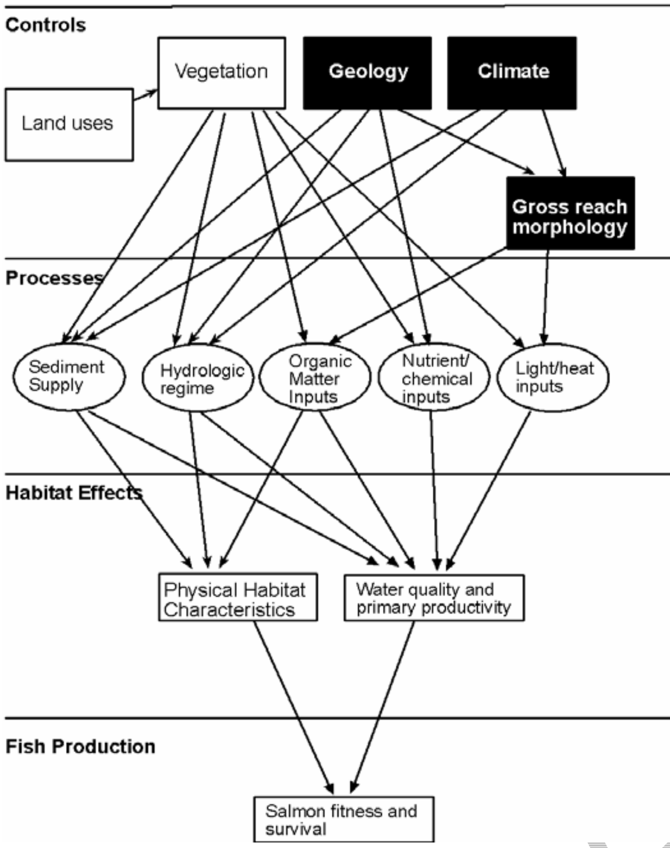


Figure 1. Schematic diagram of relationships between controls on watershed processes, process effects on habitat conditions, and habitat effects on salmon survival and fitness (Beechie *et al.* 1999). Dark boxes in upper row are ultimate controls, which are relatively immune to human actions (with the exception of climate change, which we are not addressing here). Light boxes are proximate controls, which are most directly affected by land use actions. Our analyses make use of these relationships to evaluate where processes have likely been disrupted (based on mapping current and historical states of various controls), and where habitat conditions are likely degraded (based on impairment of important driving processes or field habitat data).

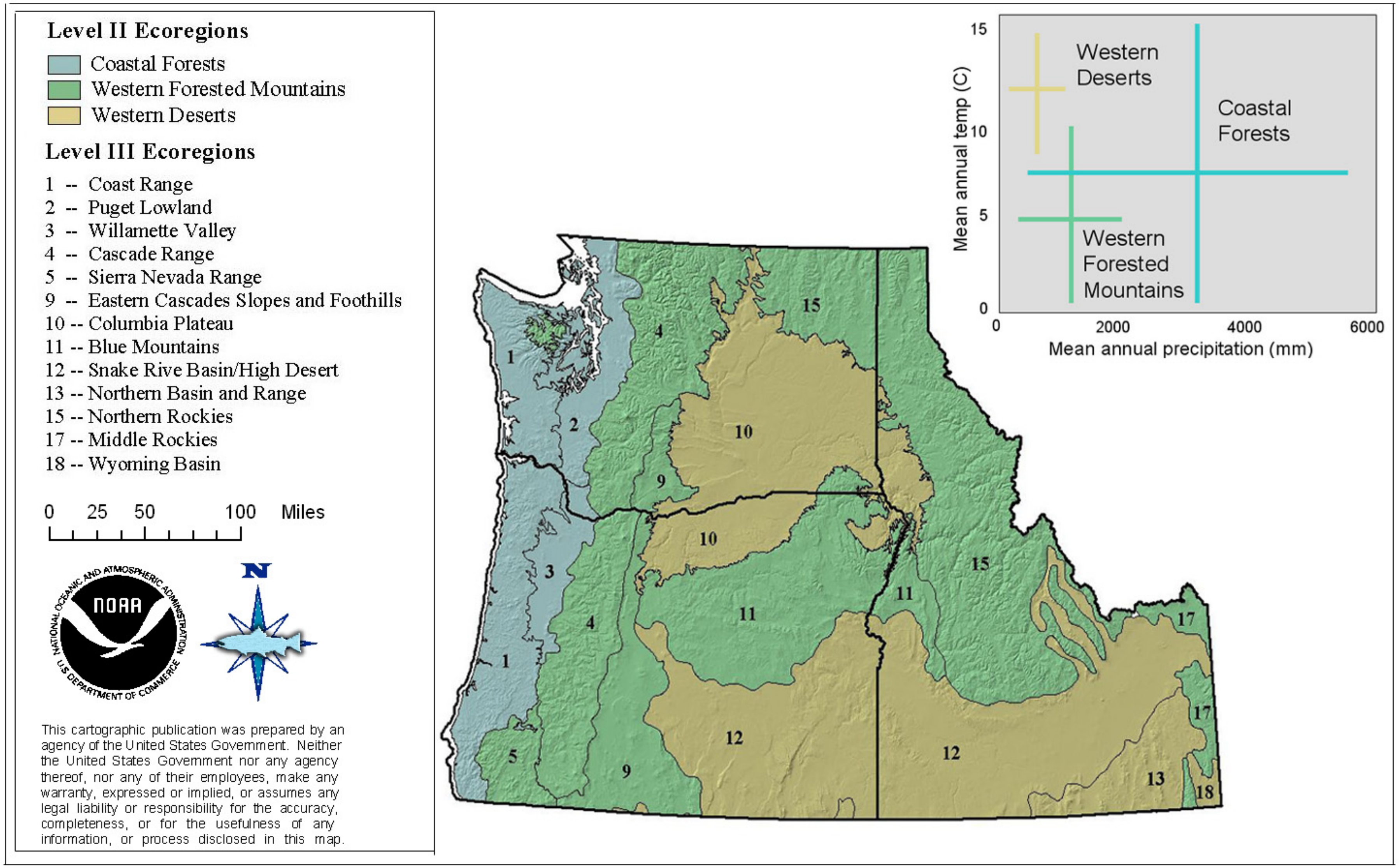


Figure 2. Ecoregions of the Pacific Northwest. Inset shows range of mean annual temperature and mean annual precipitation of each HUC6 within Level II ecoregions.

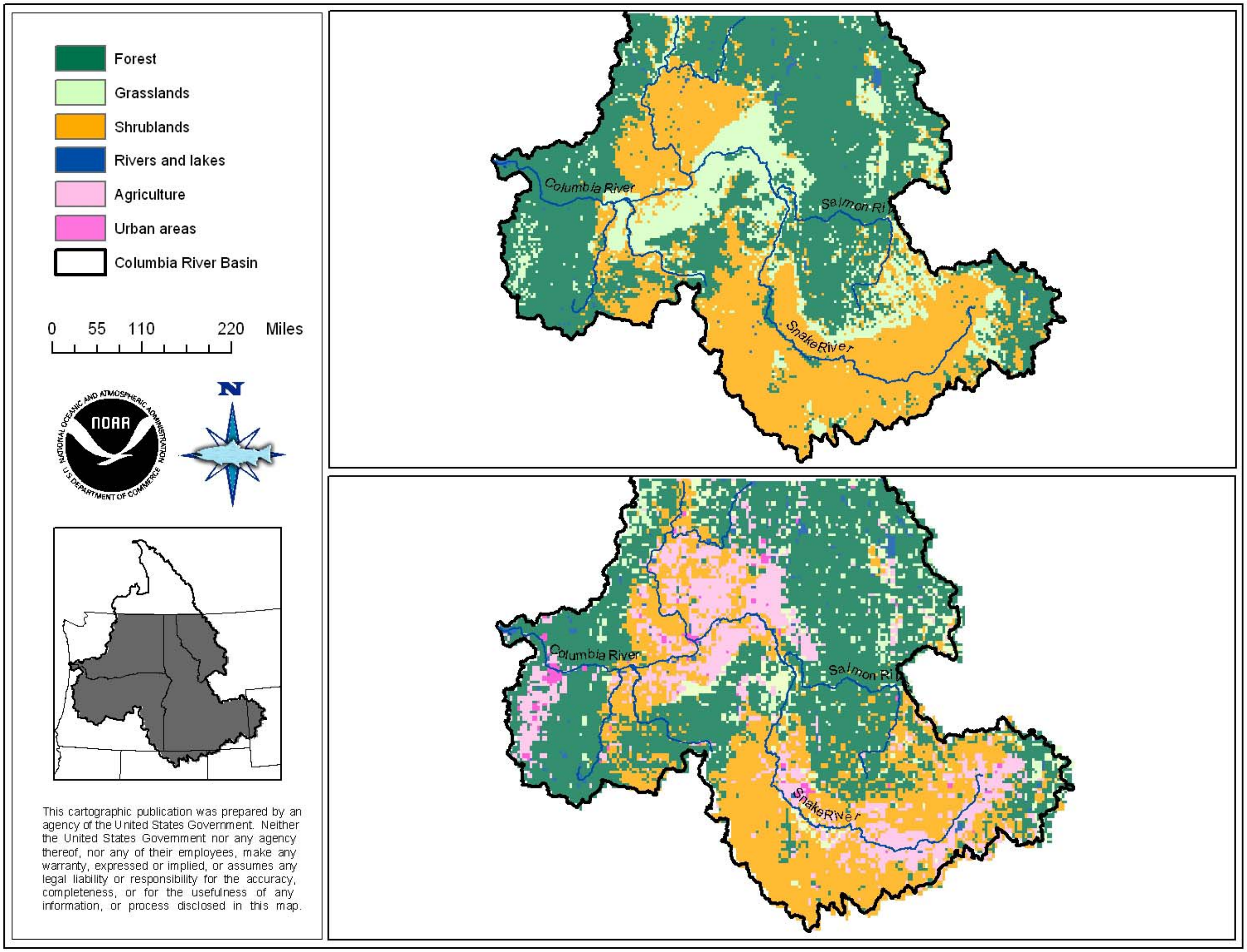


Figure 3. Historical and current land cover maps in the Pacific Northwest (source ICBEMP). Note that for some analyses more detailed and recent current landcover maps were used. Sources for more detailed data were either Northwest Habitat Institute or USGS, depending on requirements of each analysis.

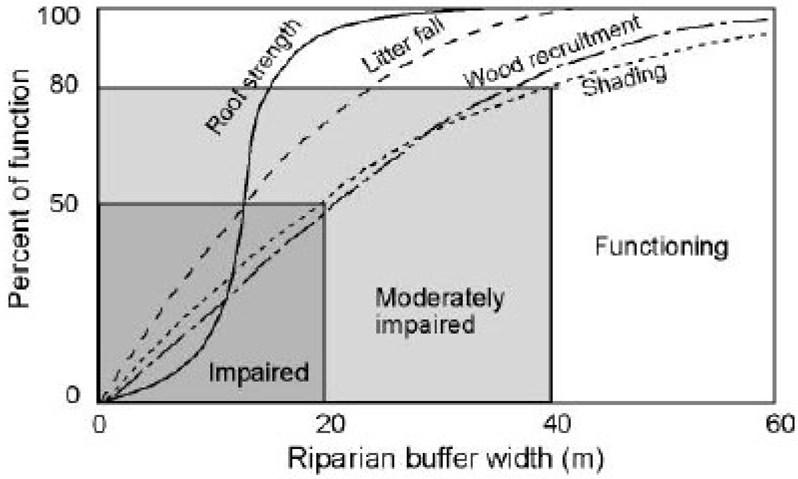


Figure 4. Percent function for four riparian functions important in forested areas west of the Cascades; reproduced from Beechie *et al.* 2003; caption reads "Illustration of change in riparian function with distance from channel (curves adapted from Sedell *et al.* 1997), and the Skagit Watershed Council's (1998) classification of impaired, moderately impaired, and functioning riparian forests."

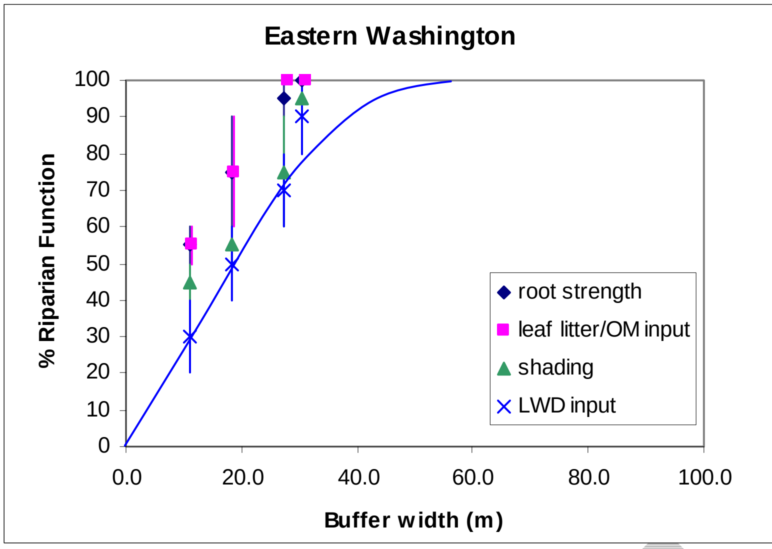


Figure 5. Percent function for the same four riparian functions as in Figure 4, but modified for eastern Washington, based on site potential tree height (west = 175 ft or 53.34 m, east=125 ft or 38.1m). Data are from a table on NOAA Fisheries Northwest Region's website (www.nwr.noaa.gov/1salmon/salmesa/4ddocs/4dws4c.htm), but no citations are provided. Likely, this information comes from Spence *et al.* 1996 (NOAA Fisheries 2003). The curve is drawn for large woody debris (LWD) input, because it is the function requiring the widest buffers for complete functionality.

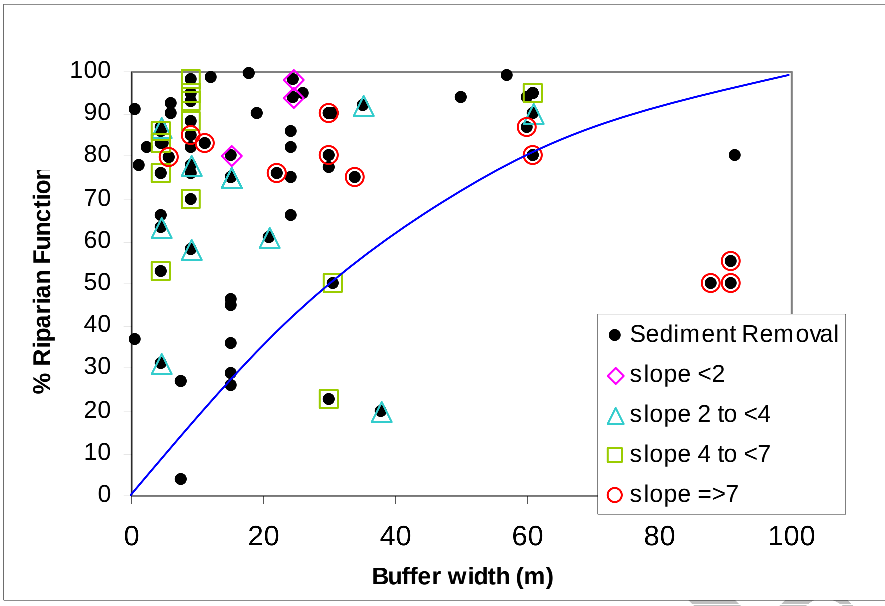
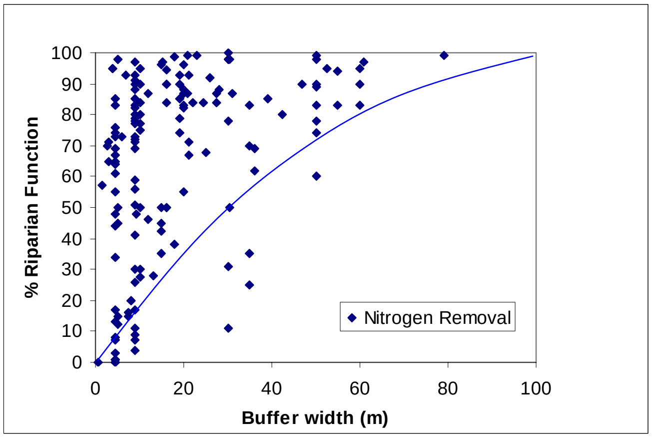


Figure 6. Percent of sediment removal vs. buffer width, based on 80 data points from 39 studies; 3 of these studies included data for forested riparian buffers and 37 included data for presumably non-forested (e.g., grass, shrub, or type not reported) riparian buffers. Only data reported for 100m buffer widths or less are included (5 points fell beyond). Colored symbols separate studies by percent hillslopes of riparian areas, where data were provided in original studies. Sediment forms varied (e.g., total suspended solids, fine sediment), and all forms are included here. The blue line represents a threshold above which 90% of the data points fall.



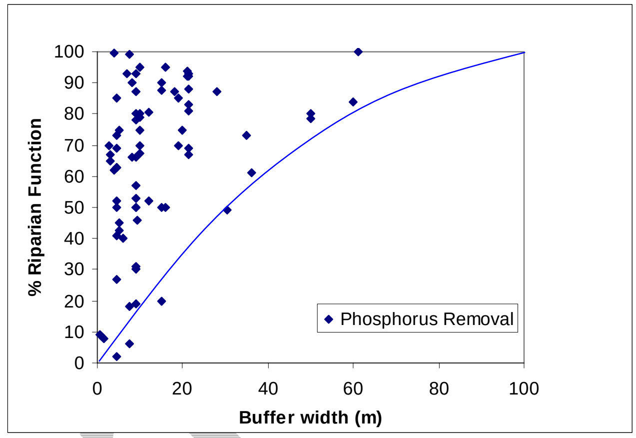


Figure 7. Percent removal of nitrogen (left panel) and phosphorus (right panel) in relation to riparian buffer width. The blue line is the same threshold as derived for sediment removal (Figure 6), overlain for comparison. Data are from 53 and 29 published studies for nitrogen and phosphorus, respectively. Note that 5 and 7 data points, respectively, fell below zero (indicating increases rather than decreases in nutrient amounts); these data points occurred in buffers <10m wide.

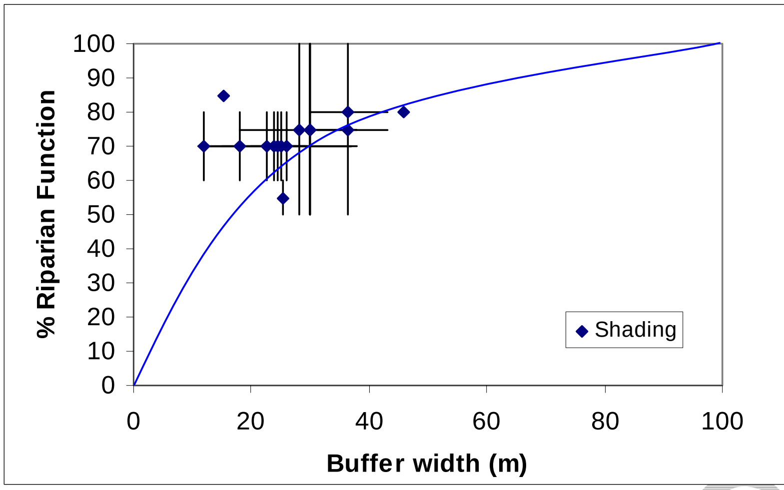


Figure 8. The relationship of percent shading to riparian buffer widths, as reported in 15 published studies. Most studies reported ranges, rather than mean values. Error bars, when present, are ranges of values reported in each study, and points indicate means. No information is available on type of vegetation in buffers reported here. The blue line represents a threshold above which 90% of the data points fall.

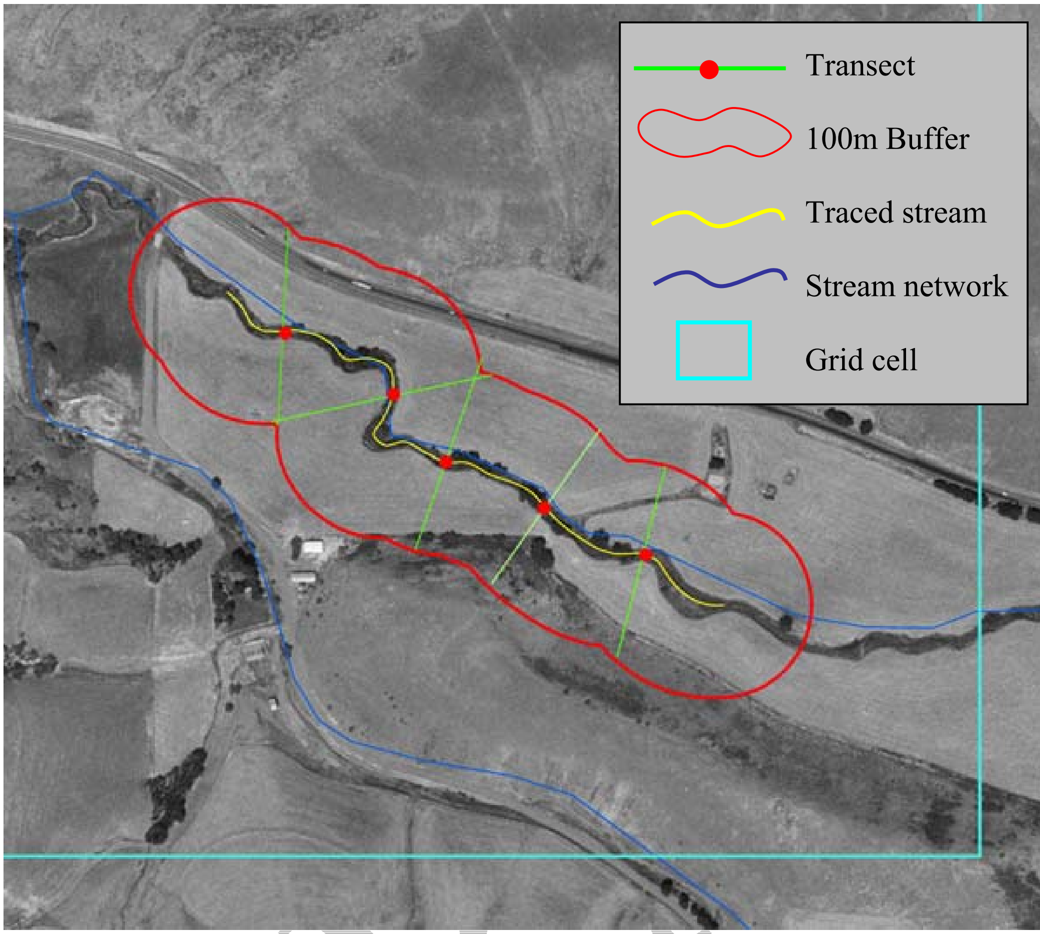


Figure 9. An example aerial photo and sampling transects in an agricultural area.

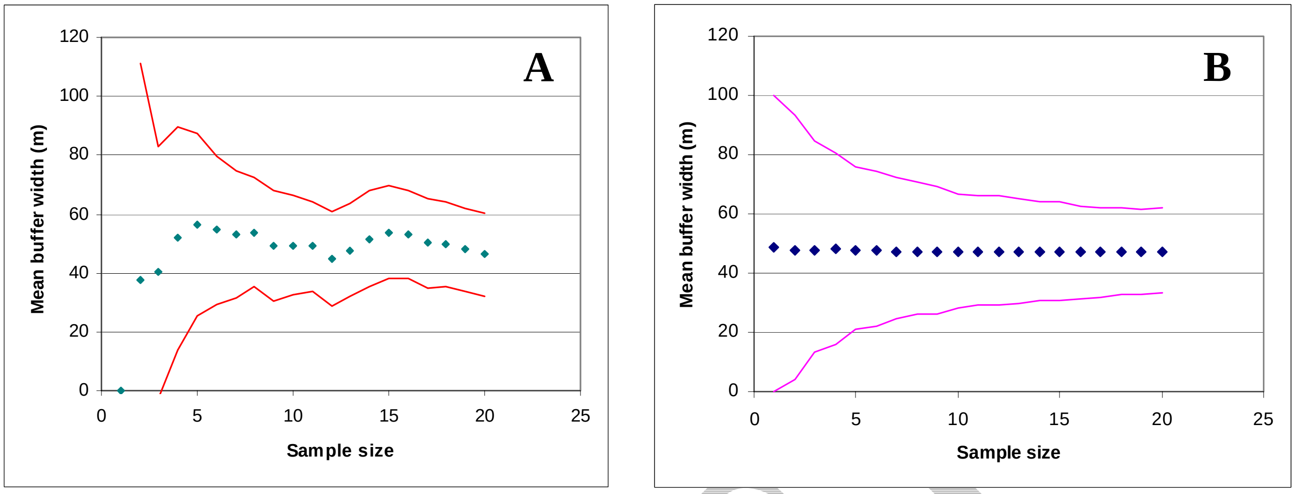


Figure 10. Cumulative mean and confidence intervals plotted against sample size from one randomly drawn dataset (A), and for all 1000 bootstrapped data (B).

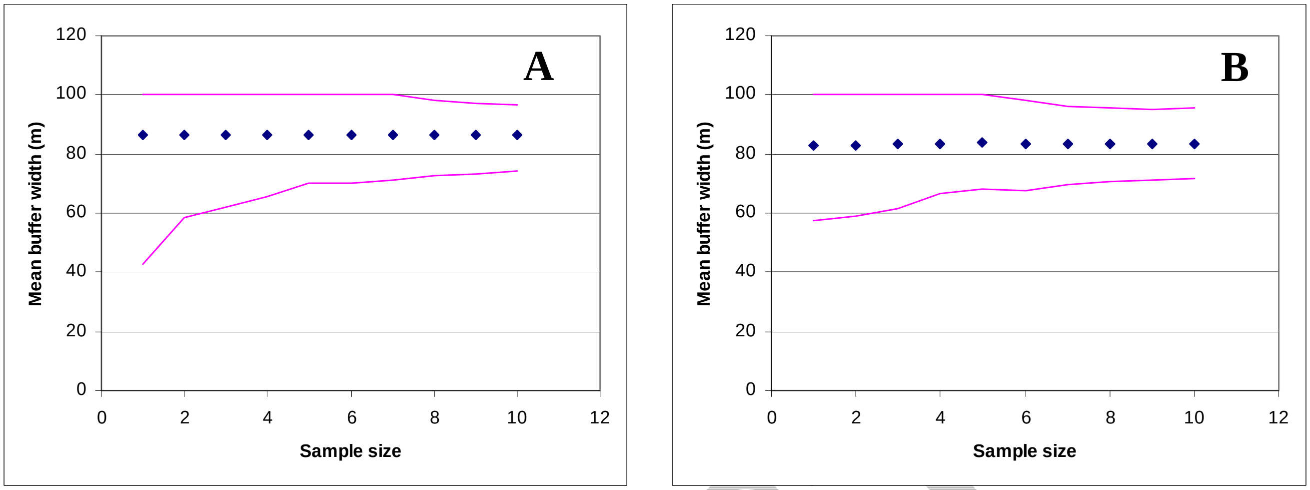
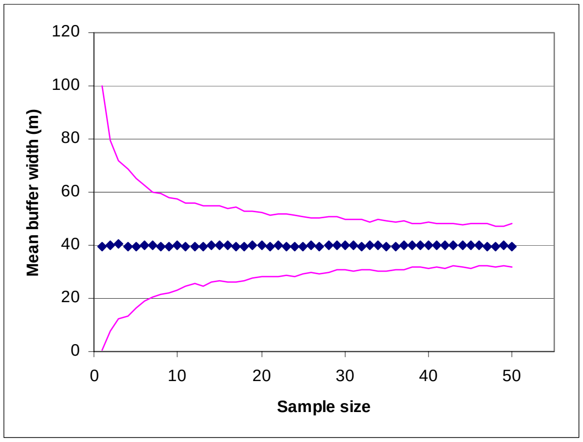


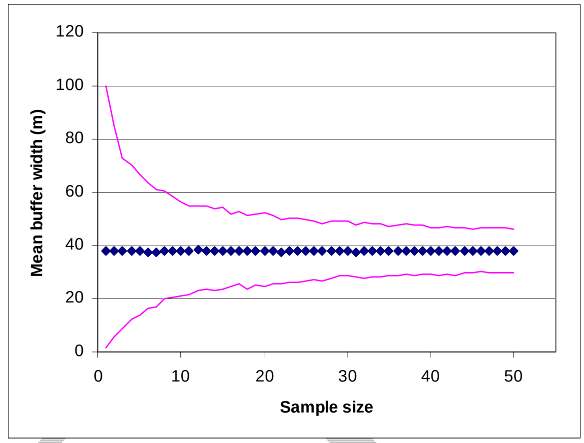
Figure 11. Cumulative mean and confidence intervals of bootstrapped data for the forested stratum (A), and the shrub/grass stratum (B).

RIPARIAN SCREEN **FLOODPLAIN SCREEN**

Agriculture/urban Stratum

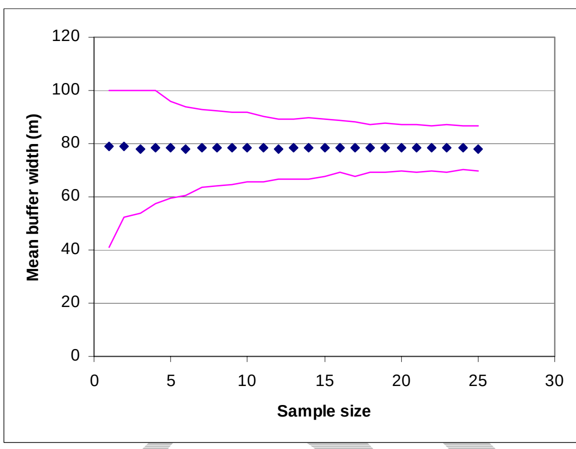


mean = 39.8, ci = +8.4/-7.8, n=50

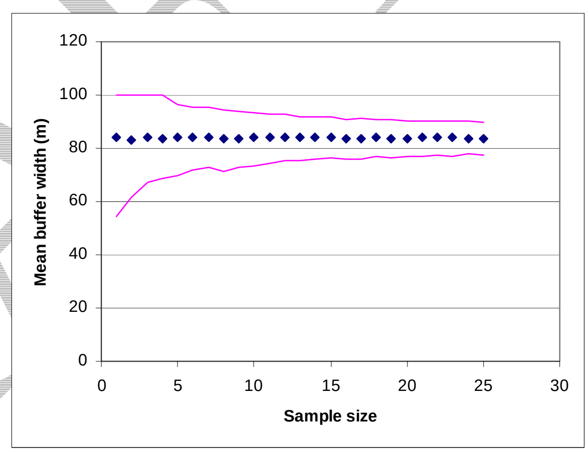


mean = 37.9, ci = +8.1/-8.2, n=50

Forested Stratum

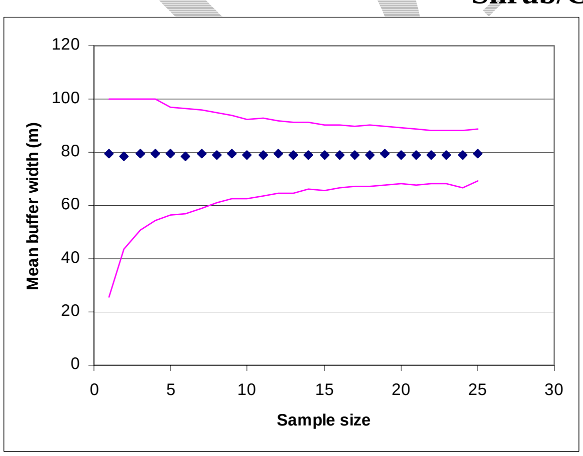


mean = 78.4, ci = +8.4/-8.6, n=25

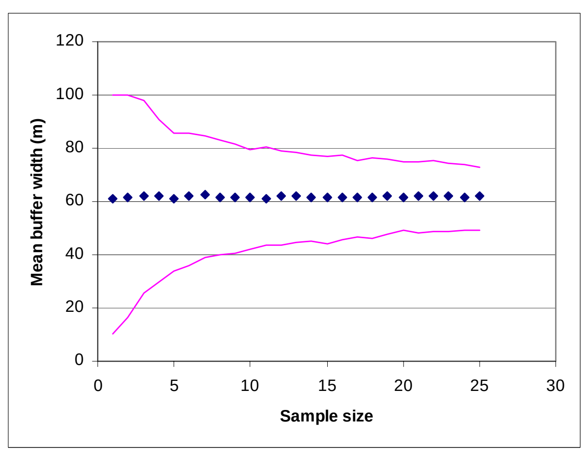


mean = 83.9, ci = +5.9/-6.1, n=25

Shrub/Grass Stratum



mean = 79.1, ci = +9.3/-10.2, n=25



mean = 61.7, ci = +11.0/-12.4, n=25

Figure 12. Bootstrap analyses for aerial photos throughout the Columbia River Basin.

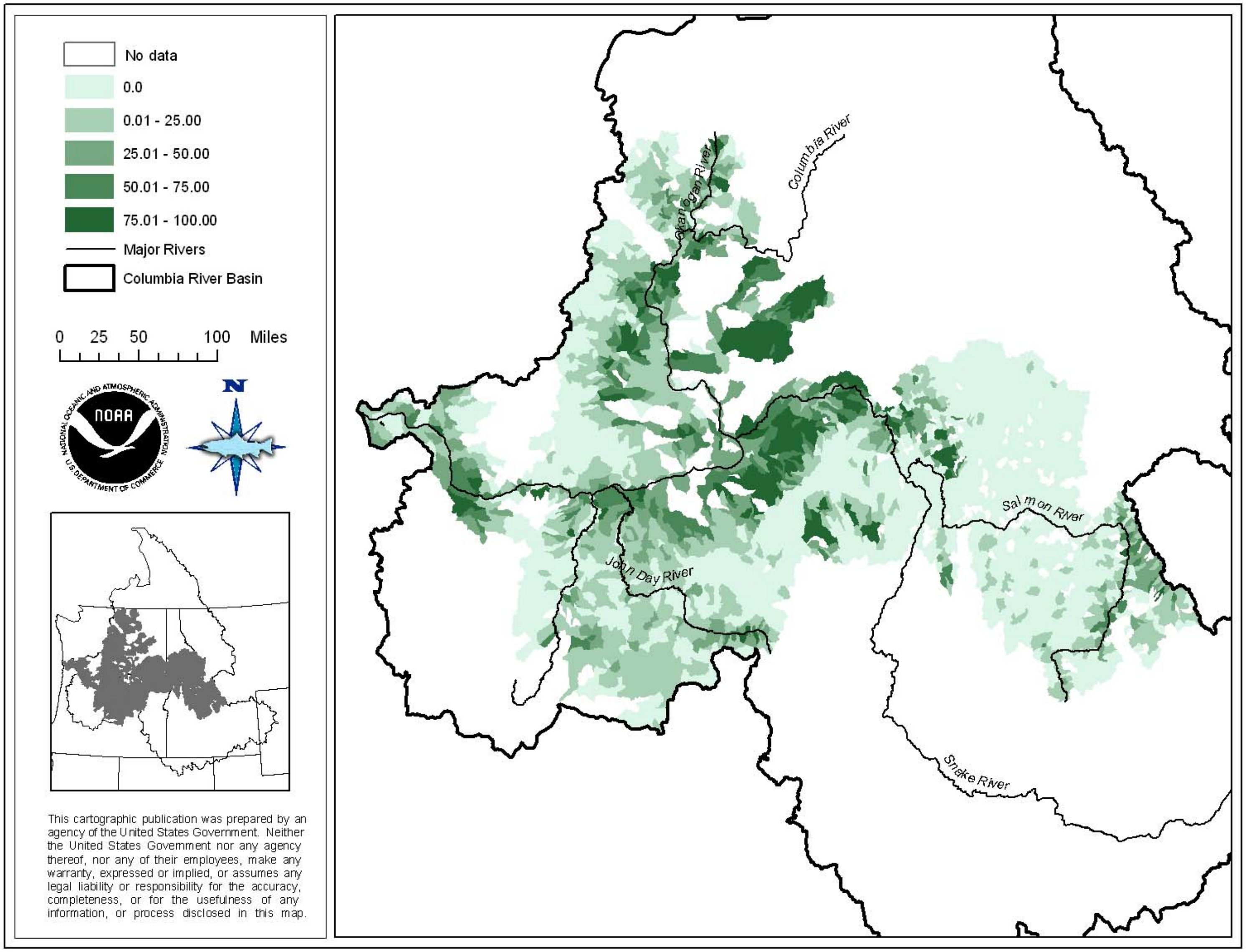


Figure 13. Riparian Coarse Screen: Percent of streams in the Columbia River Basin with gradients between 1 and 4% running through areas classified by the Northwest Habitat Institute layer as agriculture/urban, summarized by 6th field HUCs

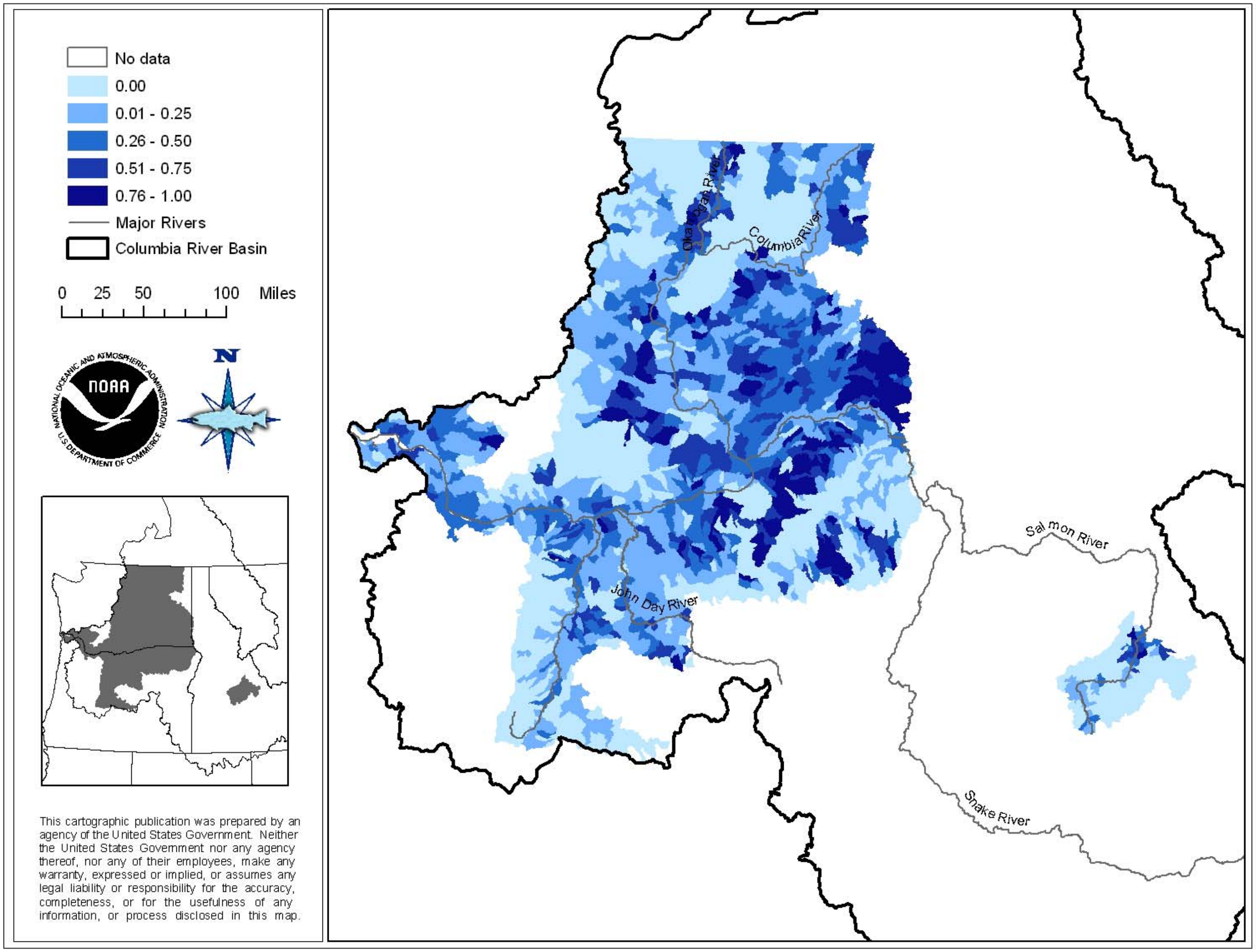


Figure 14. Floodplain Coarse Screen: Proportion of streams in the floodplain areas of the Columbia River Basin running through areas classified as having been converted from natural conditions by humans, summarized by 6th field HUCs

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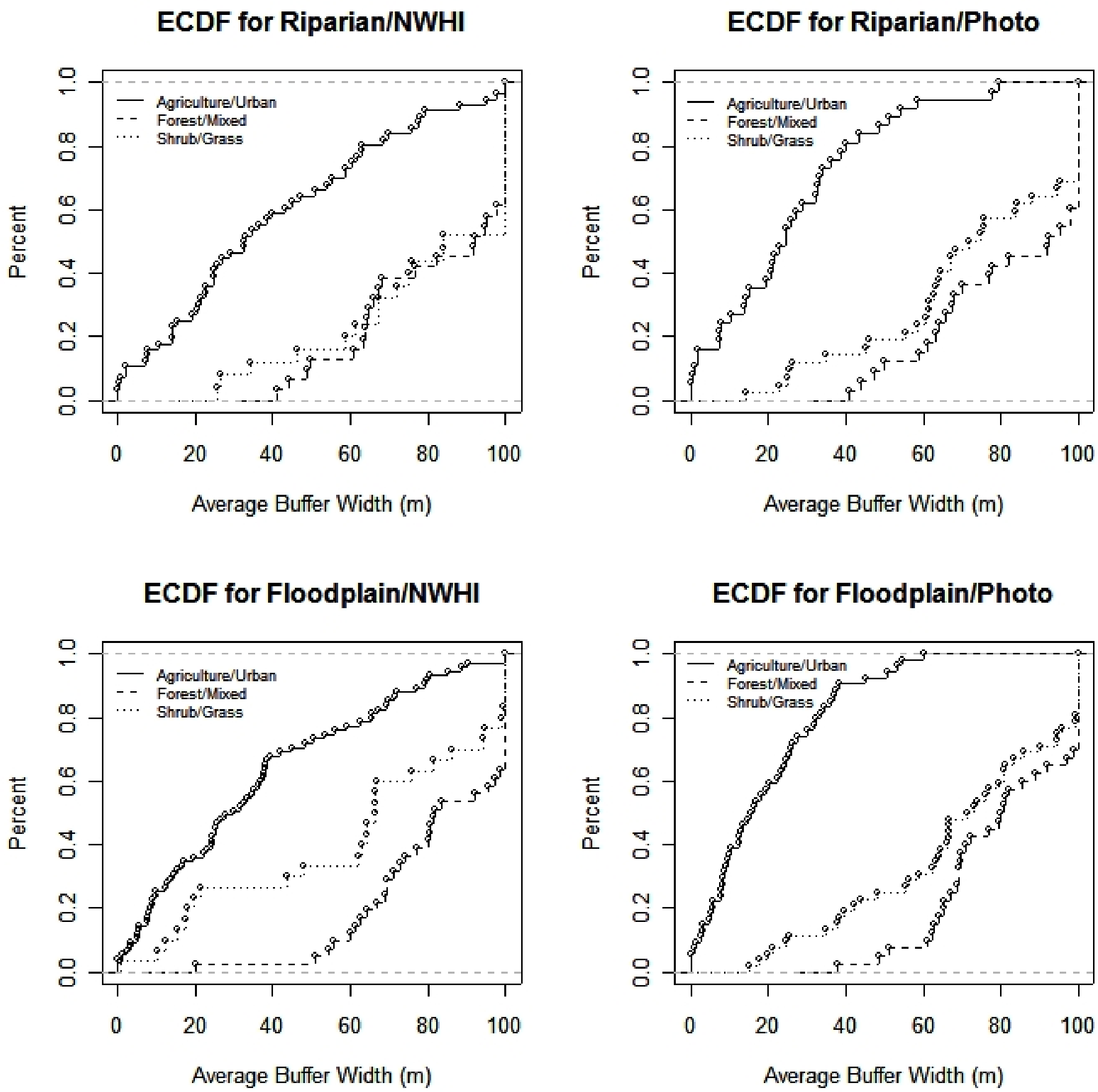

Figure 15. Cumulative frequency distributions of buffer width measured in aerial photos for each combination of data analyzed. Riparian = data used in the riparian screen (e.g., streams with gradients between 1 and 4%); Floodplain = data used in the floodplain screen (e.g., streams falling within designated floodplain areas); NWHI = habitat strata classified by the Northwest Habitat Institute data layer; Photo = habitat strata classified by the dominant type of habitat observed in aerial photos.

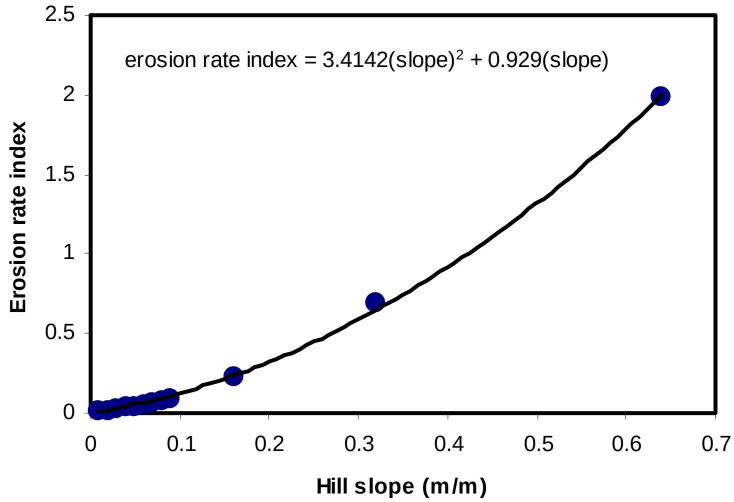

Figure 16. Sensitivity of erosion rate index to hillslope angle.

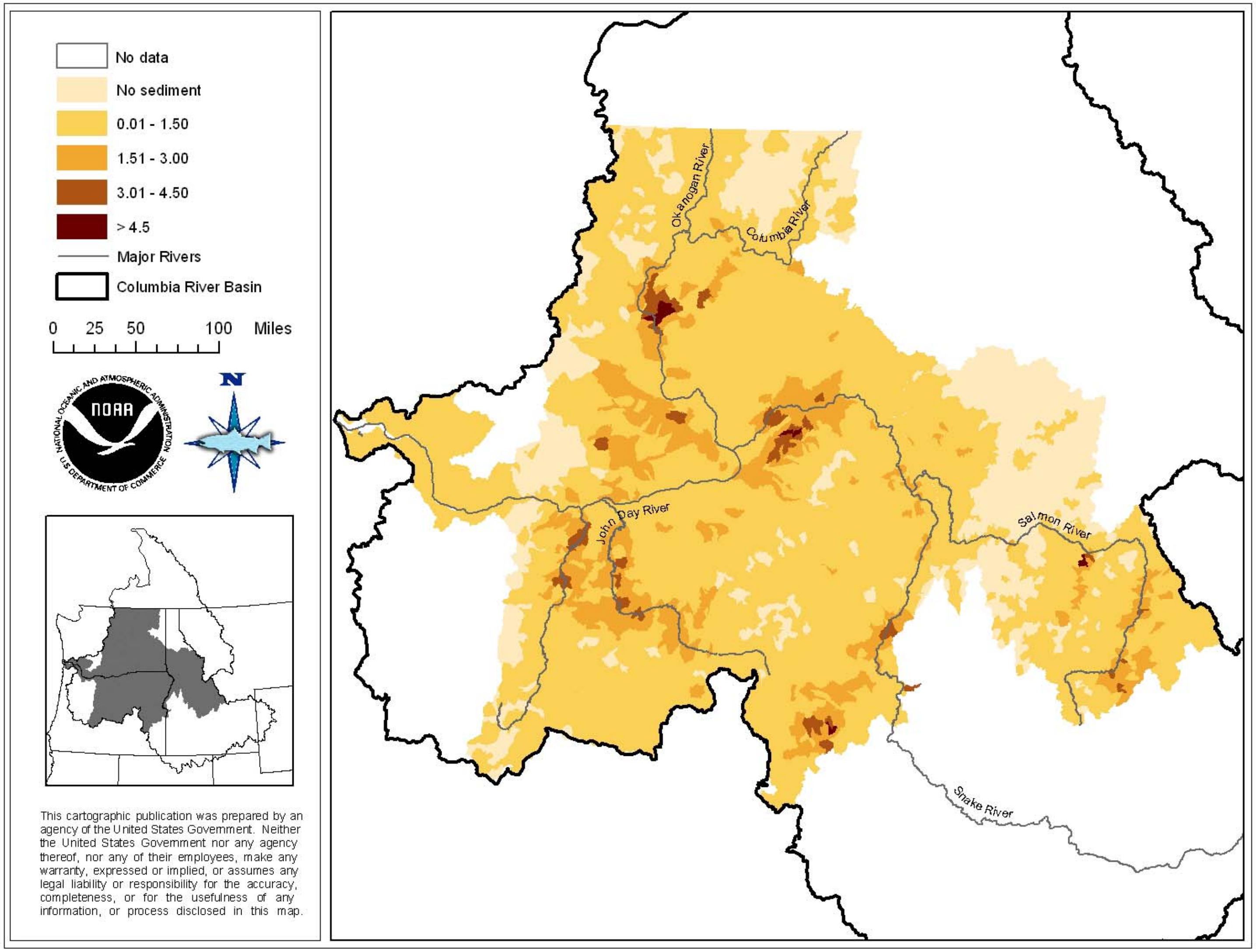

Figure 17. Historical non-forest sediment supply ratings averaged by HUC6 for the Columbia River basin and lower Columbia chum ESU.

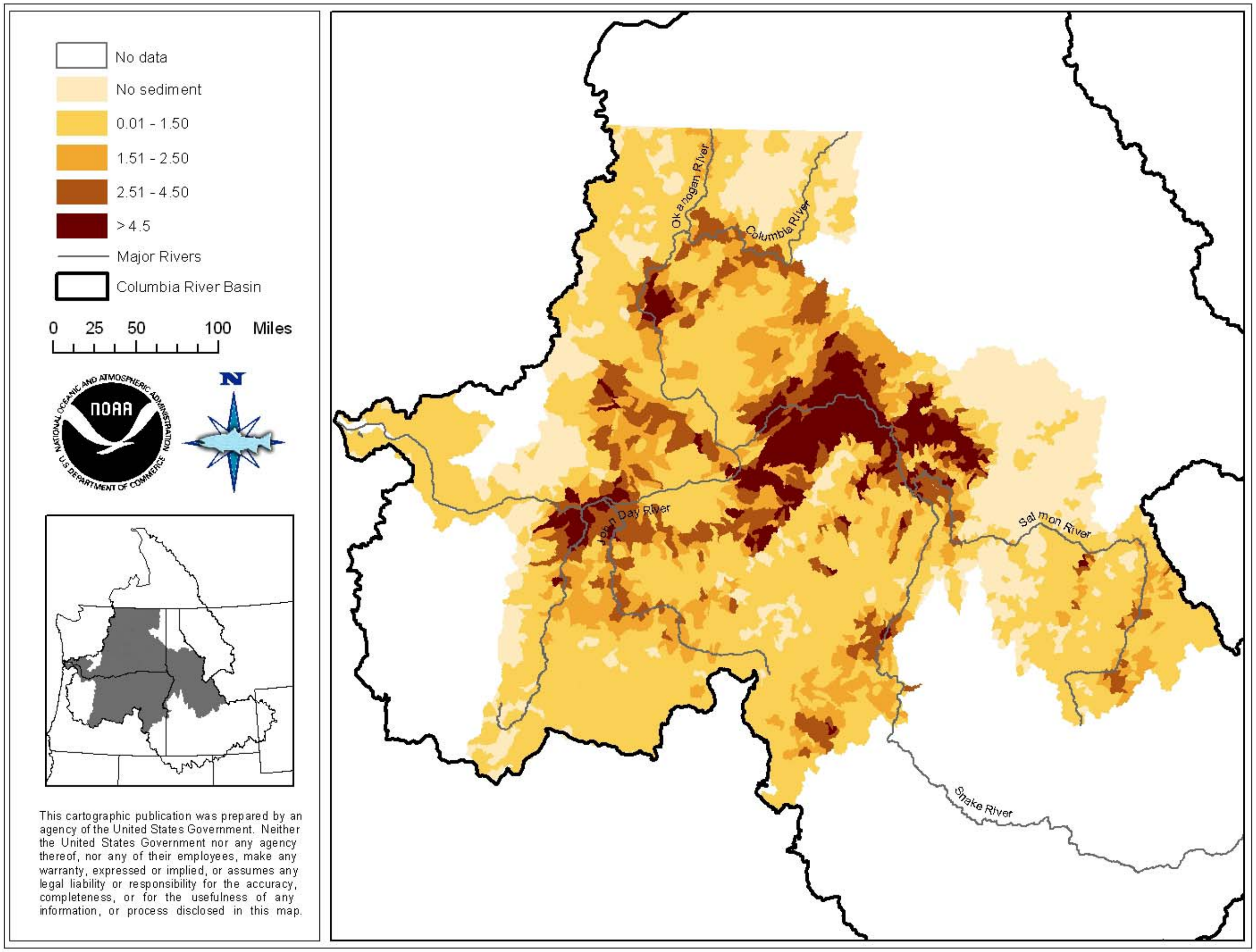

Figure 18. Current non-forest sediment supply ratings averaged by HUC6 for the Columbia River basin and lower Columbia chum ESU.

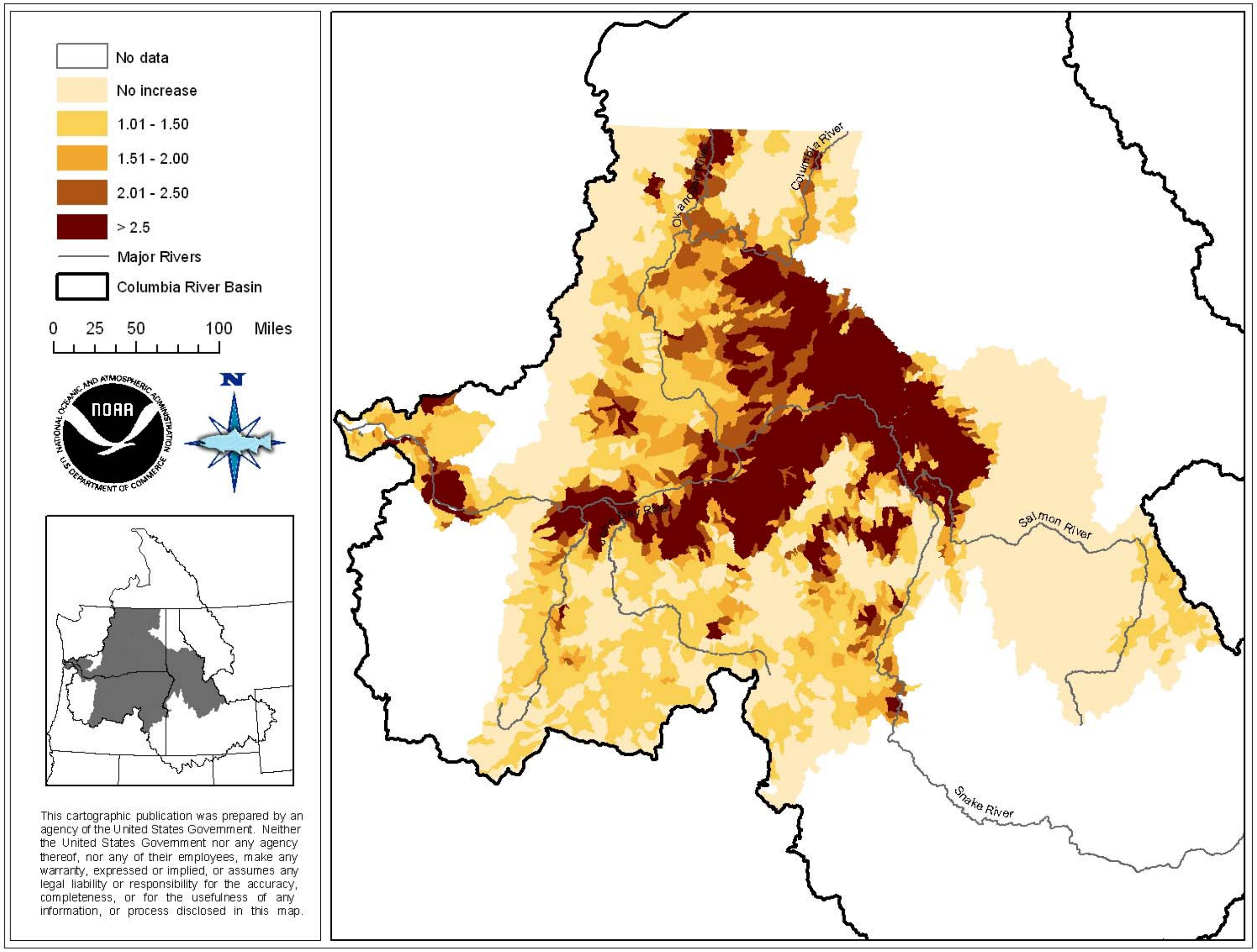

Figure 19. Estimated change in non-forest surface erosion rate from historical conditions in the interior Columbia River basin and lower Columbia chum ESU.

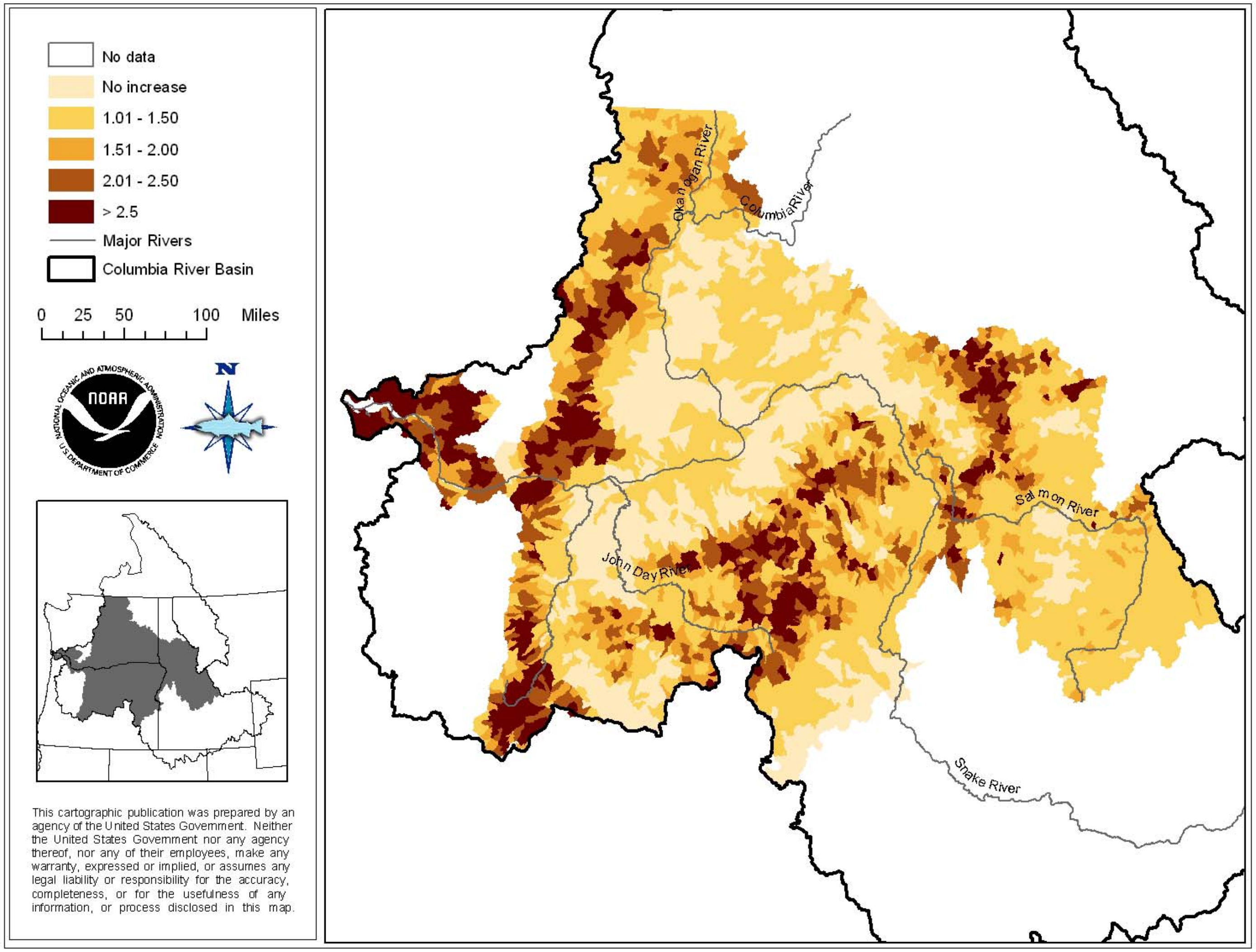

Figure 20. Estimated change in forested-land mass wasting and surface erosion rate from historical conditions in the interior Columbia River basin and lower Columbia chum ESU.

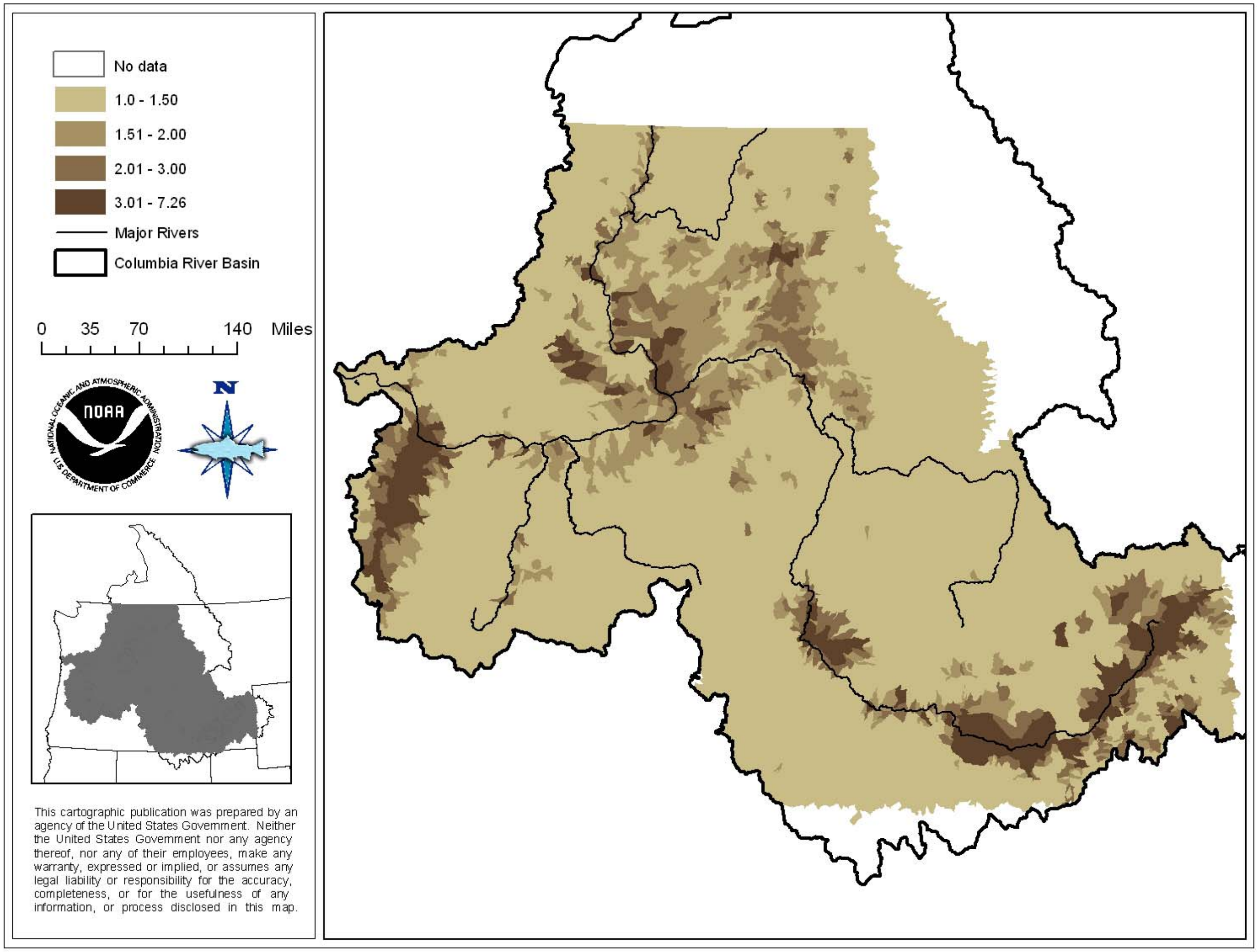

Figure 21. Area weighted average rating of potential exposure to pollutants by HUC 6 in the interior Columbia River basin and lower Columbia chum salmon ESU.

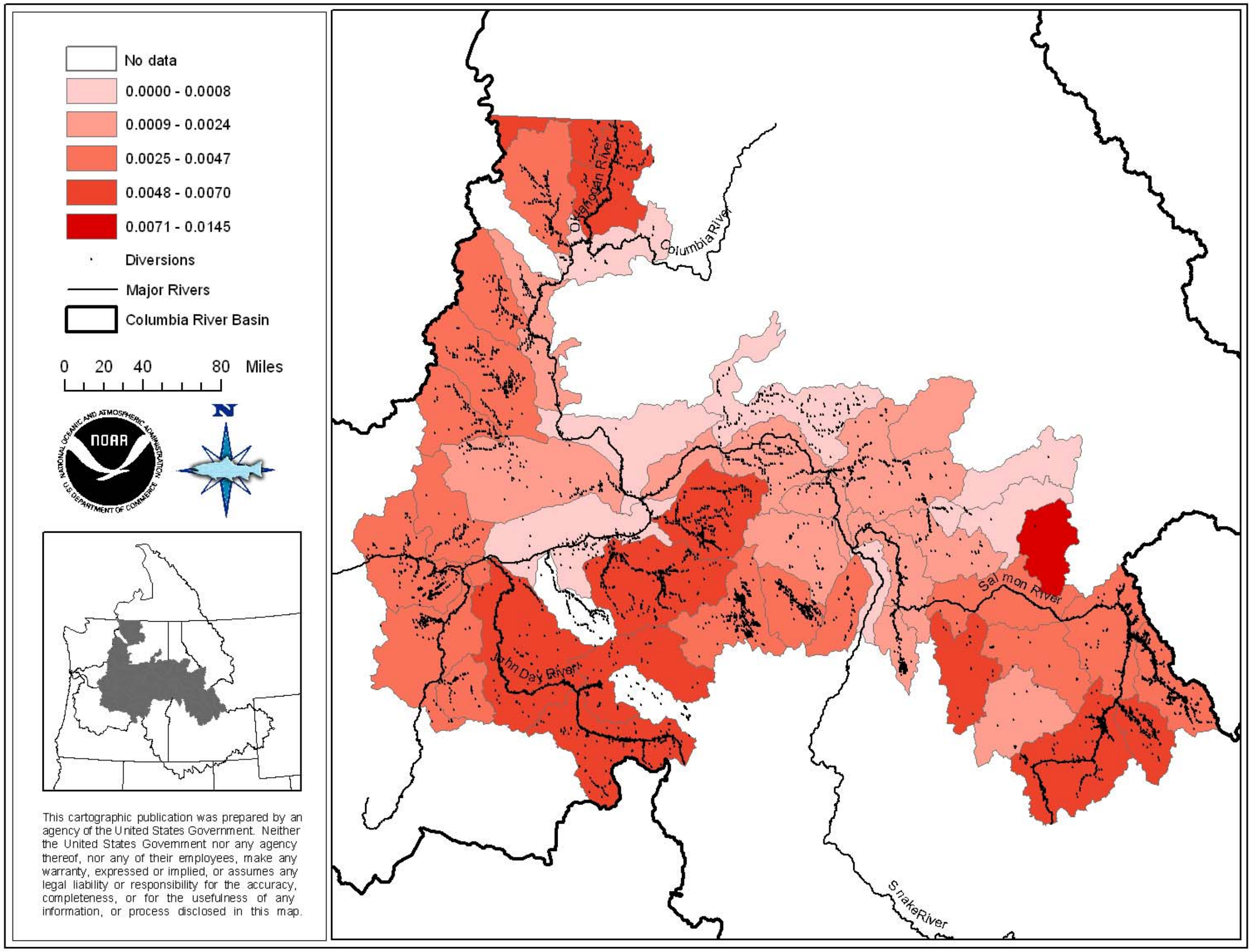

Figure 22. Percent water withdrawal at low flow summarized by HUC4.

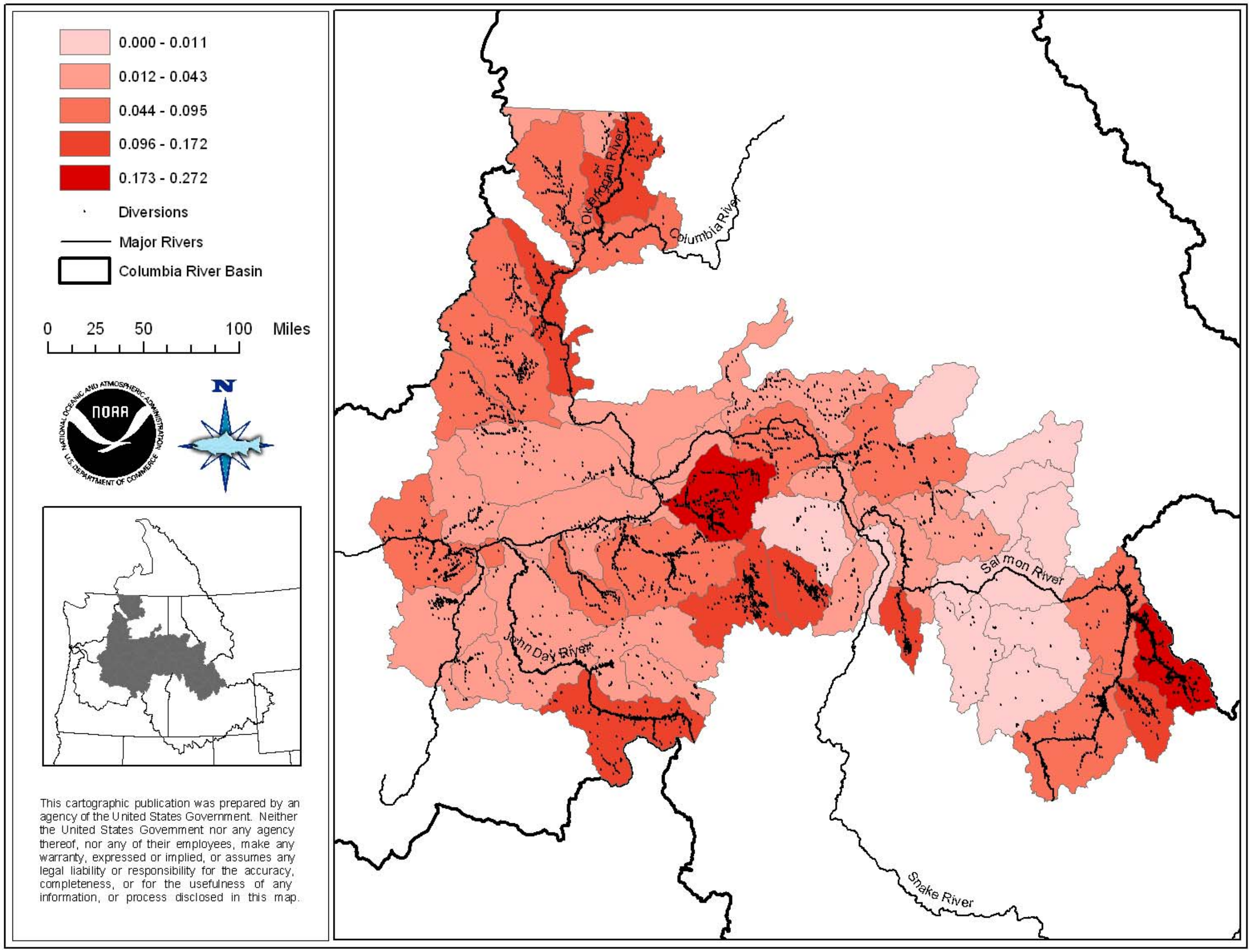

Figure 23. Density of diversions (number of diversions per km^2) at HUC4 resolution.