Lewis River Case Study Final Report

A decision-support tool for assessing watershed-scale habitat recovery strategies for ESA-listed salmonids

Lewis River Case Study

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Introduction

Watershed scale management for threatened or endangered Pacific salmon is essential for coordinated salmon recovery efforts, yet developing an efficient and effective habitat management strategy over large spatial extents presents new challenges (Beechie et al. 2003). Limitations in our understanding of how landscapes impact in-stream habitats and in how fish populations respond to those habitats are magnified as we move from the reach to the watershed scale. Much of fisheries research, particularly in restoration monitoring, is conducted at the reach scale and there are few tools available for appropriately scaling results (Urban 2005). A further complication is that within a watershed, more than one ESA-listed species is often the target of a particular management strategy. The ultimate goal of any watershed management strategy for ESA-listed salmonids is to improve future habitat conditions in such a way as to increase the likelihood of salmon population persistence. Therefore, watershed management strategy selection should be based on predictions of how the suite of restoration and protection actions described by that strategy are predicted to impact future watershed condition or population performance. Identifying how alterative watershed management strategies may impact future conditions across the watershed is a key to making the best habitat management decisions now.

Pacific salmon are wide-ranging species that spawn and often rear in freshwater. Populations in the Pacific Northwest have declined to a fraction of their historical abundance (Meengs and Lackey 2005). Chum salmon, chinook salmon, and steelhead in the Lower Columbia River were listed as threatened by NOAA Fisheries under the ESA in 1998 and 1999. Coho salmon were listed as threatened in 2005. In response to these declines, a great deal of money has been and will likely continue to be spent on actions to restore and protect their freshwater habitats (NOAA 2004). While each species has unique habitat needs, characteristic spawning habitats are low gradient, cobble-based channels and characteristic rearing habitats are smaller channels with some habitat complexity in the form of pools and overhanging banks; channel gradient preferences during rearing vary widely by species. During all life history phases, salmon require adequate cool and clean water (Groot and Margolis 1991).

During the years 2000 to 2003, the Pacific Coastal Salmon Recovery Fund and Pacific Northwest states allocated approximately $500 million to salmon recovery (NOAA 2004). Common habitat restoration and protection activities include road decommissioning and upgrades to reduce sediment inputs to headwater streams and reduce peak run-off from storm events; culvert or small dam replacements and modifications to improve fish passage and open currently inaccessible habitats; riparian plantings and harvest protections to provide shade, bank protection, and sources of large wood that can increase channel complexity; side channel reconstruction and dike removal to increase and improve floodplain habitats; and, placement of in-stream structures to increase habitat complexity, decrease stream power, and reduce transportation of sediment. Choosing the appropriate suite of actions and the most efficient locations for the actions is both difficult and essential. There is vast literature on identifying restoration actions and locations within a watershed (e.g., Beechie et al. 2003, Roni 2004), but there is little research predicting the cumulative impact of multiple restoration actions within a watershed on habitat conditions and on salmon population performance.

The decision support system designed and applied in the Lewis River watershed can predict the future landscapes that would result from alternative watershed-scale management strategies. We predict the impacts of 6 alternative watershed management strategies and evaluate those potential

future landscapes with a suite of physical and biological response models. There are four main steps in the application of the decision support system. First, we generate a series of potential watershed management strategies. Next we identify and model specific actions that would result from the application of each strategy. And, we model the physical habitat impacts of those actions, creating 6 potential future landscapes. Third, we quantify habitat quality and distribution for each potential future landscape and predict the biological implications for multiple species. And, fourth, we synthesize results using metrics that summarize predicted physical conditions and biological responses for each of the watershed management strategies. The outcomes of our analyses are predictions of the benefits and trade-offs across the watershed of each of the 6 modeled strategies. These predictions can help to guide the development of an on-the-ground watershed management strategy for the Lewis River basin.

Recovery planning of listed salmon and steelhead populations in the Lower Columbia and Willamette regions is required by the Endangered Species Act (ESA). The Lewis River project was designed as a forum for working through the technical issues involved in identifying the factors currently limiting recovery of ESA-listed salmonids and in identifying suites of actions that will address those factors. It has evolved into a decision-support system that can serve as an example of how multiple models can be incorporated into large-scale habitat recovery planning and of how science-based habitat recovery planning can occur over entire watersheds. While NOAA Fisheries is ultimately responsible for producing plans that address ESA requirements, the agency hopes to rely on locally developed watershed-scale and regional plans as building blocks for ESA recovery plans to the extent possible. It is hoped that the case study will illuminate these technical issues for the Lower Columbia Technical Recovery Team (LCR – TRT) and for other watershed planning entities. Since many of the technical issues confronting recovery planning have never been satisfactorily addressed, working through an example is the best way to gain insight into the required analyses. This main body of this report provides a short synthesis of the project. Detailed descriptions of models used in the DSS are provided in the appendices or in other published material.

The Lewis River Watershed

The Lewis River was identified as the case study watershed because it contains a mix of the issues confronting all of the watersheds in the Lower Columbia River. It is the only watershed in the Lower Columbia River to contain all races of listed salmon and steelhead. Population status in the watershed ranges from relatively healthy (bright fall chinook salmon) to extirpated (spring chinook salmon). The Lewis River also contains habitat types and land ownerships representative of the Lower Columbia domain. In addition to selecting the Lewis River because of its representative mix of technical issues, the river's current management environment is conducive to the case study. The LCFRB is actively engaged in the technical work of recovery planning and interested in collaborating with the WLC-TRT; the hydro-system on the Lewis River is currently involved in re-licensing through the Federal Energy Regulatory Commission (FERC); and ongoing analyses are being developed for a habitat conservation plan in the Lower East Fork of the Lewis. We were able to build on existing limiting factors analyses (LFA) (Wade 2000) and to work in cooperation with restoration planning analyses underway by the Lower Columbia Fish Recovery Board (LCFRB 2004a; LCFRB 2004b).

The Lewis River watershed in southwestern Washington State, USA, encompasses 2,760 square km and drains the western slope of the Cascade Mountain range, emptying into the Columbia

River 140 km upstream of the mouth. Average annual precipitation in the lower watershed is nearly 200 cm (LCFRB 2004a). The hydrograph has two main peaks. The largest of these is in the fall and results from rain-on-snow events and the other is in the spring and results from snowmelt. There are three large, impassible dams on the North Fork of the Lewis River. The lowest of these, Merwin Dam (RKM 31.4), was completed in 1931 and is currently a barrier to all anadromous fish (Figure 1; Appendix A).

The landscape has historically been influenced by logging, fire, and volcanic activity. A detailed report on historical watershed conditions, land-use history, and recent changes in watershed condition is provided in Appendix B. Current conditions in the Lewis River watershed are summarized in the LFA and by LCFRB (Wade 2000; LCFRB 2004a). The majority of the headwaters of the basin are forested and in public ownership; active logging was common until the 1980s. Currently, logging activities are greatly reduced, particularly on federal lands. All riparian areas on upland forested lands are protected by the Washington Forest Practices Board (2004) and the U.S. Forest Service Northwest Forest Plan (USDA/USDI 1994). Stand replacement fires were common in the basin in the early part of the 20th century. Hydrology, sediment transport, and vegetation continue to show impacts from these historical fires especially in the East Fork Lewis. The main tributaries on the north side of the Lewis basin drain the slopes of Mt. St. Helens, which erupted in 1980 (Figure 1). Very fine sediments originating as volcanic ash from past eruptions characterize the northern subwatersheds of the upper Lewis drainage.

Small hobby farms, newer low-density residences, and agriculture dominate lowland areas. Gravel mining occurs in the lower parts of the East Fork of the Lewis River, and the mainstems of both the East and North Forks of the river are heavily channelized. Historically, the mainstem river was characterized by anastomosing channels on a wide, active floodplain that supported large deciduous trees (R2 Resources 2004). Using aerial photographs, we estimated that, historically, the East Fork Lewis River had 0.5 km of side channel for every kilometer of river. The human population in the watershed is relatively low, 14,157 people in 2002, and is concentrated in Woodland, Washington near the mouth of the river (U.S. Census Bureau 2000).

Four species of salmonids inhabit the Lewis River watershed: chinook, steelhead, coho, and chum salmon. These include 10 populations of ESA-listed salmonids: Lewis River fall run chinook salmon, Lewis River late fall run chinook salmon, Lewis River spring run chinook salmon, North Fork Lewis River summer run steelhead, East Fork Lewis River summer run steelhead, North Fork Lewis River winter run steelhead, East Fork Lewis River winter run steelhead, North Fork Lewis River coho salmon, East Fork Lewis River coho salmon, and Lewis River fall run chum salmon. Only a remnant population of spring chinook remains in the basin. Historically, spring chinook spawned primarily in the upper watershed. Currently they spawn predominantly in the mainstem directly below Merwin Dam. Early fall chinook populations are relatively abundant and spawn primarily in the mainstem sections of the East Fork Lewis River with some spawning in the mainstem North Fork Lewis River downstream of Merwin Dam. Steelhead populations are intermediate in abundance and spawn primarily in Cedar Creek, a tributary to the North Fork Lewis below Merwin Dam (Figure 1). Coho historically spawned throughout the basin and currently spawn in the main tributaries of the lower watershed. Chum historically used mainstem habitats including what is now Lake Merwin, but did not use the upper basin. Currently, chum use habitat in the lower North Fork and East Fork Lewis rivers (LCFRB 2004a, NOAA 2005). Detailed descriptions of estimation procedures for current and historical fish distribution are provided in the fish distribution section in Appendix A.

The Decision Support System

The Decision Support System spatial environmental data were produced in a GIS, and summarized by stream reach into tabular form. Thus, every piece of data used in the analyses can be linked to an individual stream reach. All summarized tabular data were stored locally on an internal network in an ArcGIS Spatial Data Engine (SDE) Oracle database. The analyses in this framework can be performed entirely in GIS; however, to reduce the possibility of user error and to expedite processing by multiple users, we automated part of the procedure into a self-contained system (the DSS tool) consisting of tabular data, models, analyses, and summary metrics. The automated DSS tool is housed in Microsoft Access and analysis procedures are written with structured query language (SQL) and Visual Basic (VB). The DSS tool in Access links directly to the SDE Oracle tables, processes analyses locally, and returns results back to the SDE database to be stored permanently for spatial summary and viewing. Figure 2 and Figure 3 provide a schematic of the DSS framework and how models relate to one another.

After all data were generated for each of the base scenarios or modified for potential conservation strategies, we ran watershed process models, the routing model, and habitat and fish response models on the original or strategy-modified datasets. Finally, reach-specific results were permanently stored for each strategy, and summary metrics were produced for later comparisons. All of these steps were automated to enable batch processing of strategies.

Setting the current and historical landscapes

An essential piece of the DSS approach was creating spatial representations of the current and historical (or unaltered) conditions for the primary natural and anthropogenic features of the Lewis River watershed. Because these templates were used to run scenarios and to measure the effects of restoration scenarios, it was necessary to create base current and historical landscapes that are as accurate as possible, using the best data and methodologies available to us. This includes upland vegetation, riparian conditions, hydrology, urban and agricultural land use, fish distribution, and potential fish habitat. Current conditions in the Lewis River watershed were estimated using GIS data layers describing vegetation, road distribution, fish distribution, and land ownership (Table 1). Detailed descriptions of the base data used in these analyses are provided in Appendix A. In some cases (fish distribution, barriers, fish potential models), the landscape varies by fish species. Current conditions were used as the template on which restoration sites were selected, and restoration strategies modeled. To estimate conditions in 2003 not described in earlier GIS layers, restoration actions completed in the basin between 1998 and 2003, such as road decommissioning or barrier removals were identified and mapped (REO 2003; NOAA 2003; WDFW 2004; SRFB 2003) (Figure 1). The landscape on which all watershed management strategies were modeled was created after incorporating the impacts of these real restoration actions. Because so many of the landscape evaluation models require stream width, we developed a customized model of stream width from field measurements (Table 2, Appendix G). Base landscapes are illustrated in Figure 4 through Figure 9. The historical base scenario represents our best estimate of landscape and habitat conditions before European settlement (Appendix B).

Developing watershed management strategies

We predicted future landscape and in-stream conditions after spending about two million dollars using each of six different restoration strategies. The six modeled restoration strategies were selected as examples of those that are commonly used or suggested. A restoration strategy can be thought of as a budget plan, describing how restoration dollars will be allocated both across project types and across the watershed. In this section, we describe each of the modeled restoration strategies. Details on how actions were selected for each strategy are provided in Appendix C and details about how the actions were implemented on the landscape are provided in Appendix D. Future analyses may involve combining or modifying these strategies as well as developing new strategies based on future recommendations.

Total cost spent under each strategy was estimated using a series of economic models (Table 3). Our goal was to spend approximately the same amount of money for each strategy. Because restoration actions require discrete costs, it was not possible for each strategy to spend exactly the same amount of money but all six strategies are within $380K of one another. The hypothetical restoration budget of two million dollars was based on the total dollars spent by the Washington State Salmon Recovery Funding Board during the years 2001-2003, per Water Resource Inventory Area (WRIA), multiplied by 3 to account for other sources of funding. Future analyses will explore the sensitivity of our findings to the total restoration budget used for modeling.

Barriers Strategy – The barriers strategy estimated future landscape and habitat conditions assuming that the total restoration budget was spent on barrier removals or passage upgrades. The modeled effect of a barrier removal or upgrade to allow fish passage is identical. In both cases, the removal opens up new habitat and increases the extent of the current fish distribution. Barriers were prioritized based on the total number of kilometers above the barrier and within the historical fish distribution and the cost of barrier removal. Barriers that were cheaper to remove and that blocked a larger number of km that were historically accessible to salmonids were given a higher priority for removal or upgrade. The three large dams on the North Fork Lewis River, Merwin, Yale and Swift, were not considered for upgrade or removal in this strategy. Using this strategy, all of the money was spent in the two subwatersheds furthest downstream (Figure 1). A total of $1,918,784 was spent under this strategy.

Barriers and Riparian Strategy – A commonly suggested restoration strategy is to protect the best habitat and initiate restoration actions that are most likely to have a positive impact on the target species (e.g., Roni et al. 2004). We followed this strategy by spending 50% of the money on barrier removals (as above) and 50% of the budget on protecting riparian areas that were already estimated to be in good condition (Figure 10; Appendix H) and that bordered stream segments estimated to be of high spawning suitability using the remotely-sensed suitability and capacity model. Riparian protection was limited to public lands that did not already have a protection ordinance. Within stream segments of high spawning suitability and good riparian condition on unprotected public lands, money was spent in the most upstream reaches first. Selection moved downstream until the entire budget was spent. Using this strategy, all of the barriers money was spent in the most downstream subwatershed. The riparian protection money was spent in the middle parts of the North Fork Lewis that drain into the reservoirs (Figure 10). A total of $1,988,638 was spent under this strategy.

Federal Strategy – In this strategy, we estimated how much improvement is possible considering only public lands. We modeled this possibility by spending 50% of the budget on barrier removals or passage upgrades on federal lands and 50% of the budget on road decommissioning on federal lands. We did not include riparian protection actions as all riparian areas on federal lands are already protected (USDA/USDI 1994). Barriers were again selected by cost per newly accessible km. Roads were selected by the amount of modeled sediment entering the stream segment to which that road segment drains. Roads in areas of high sediment yield had the highest priority for decommissioning. In this scenario, all of the funds were spent in the Upper North Fork Lewis River watersheds. Roads were decommissioned in 20 different 6th field HUs (Figure 10). In total, $1,908,093 was spent on this strategy.

EDT Strategy – The Ecosystem Diagnosis and Treatment (EDT) model developed by Mobrand Biometrics, now Mobrand, Jones and Stokes, has been used in the development of many recovery plans (LCFRB 2004a, Mobrand Biometrics, inc. 2004). The model outputs restoration and protection rankings for each EDT reach in the watershed. For each reach, key habitat elements are also ranked as to their degree of impairment. We created a simple model to translate the EDT output into guidelines for our EDT restoration strategy (Appendix L). In our strategy, 50% of the budget was allocated to restoration and 50% was allocated to protection. All of the money in this strategy was spent in the lowest subwatersheds and most was spent on riparian restoration (Figure 10c). Funds were also spent on in-stream restoration (restore for spawning), floodplain restoration, road decommissioning, and riparian protection. A total of $2,015,401 was spent under this strategy. Note that the EDT strategy is a plan for spending money according to the EDT model output for current conditions. The EDT model was also used to estimate the potential biological response to all 6 of the restoration strategies.

Landscape Strategy – As suggested in the literature (Beechie et al. 2004) and as used in several watershed plans (e.g., LCFRB 2004a), a comparison of current and historical landscape processes (including sediment, hydrology, and riparian condition) can be used to identify subwatersheds for which one or more of these processes is impaired. We conducted landscape screens for sediment, hydrology, and riparian condition (Table 4). To reach consensus about the strategy suggested by these landscape screens, we convened a panel of local and modeling experts. Experts worked in pairs to identify the restoration strategy suggested by the landscape screens. The experts were asked to include local knowledge in the judgment process as these landscape screens are meant to be employed by those familiar with the basin. Each of the 5 resulting strategies was modeled and the average budget distribution describes the strategy suggested by the landscape screens. A wide range of actions was suggested in the lower parts of the Lewis River and a large fraction of the money was spent in the three most downstream subwatersheds. This was the only scenario that included dike removal. Road decommissioning dollars were spread over all the upper watersheds, including both the North and East Forks of the river. Three subwatersheds in the East Fork were targeted for a large expenditure on floodplain restoration. A total of $1,953,674 was spent on the landscape screen strategy.

Expert Strategy – A great source of information is local or expert knowledge about the basin and about areas that are best suited for restoration and preservation actions. Expert opinion is used implicitly or explicitly in most restoration plans. After identifying the landscape screen strategy above, the experts were given all other available information about the Lewis River watershed. The additional information included a summary of the limiting factors analysis (Wade 2000), EDT model output, and output all our other landscape evaluation models applied to current

conditions. Nine experts were grouped into three pairs and one group of three. Again, they were asked to identify a best restoration strategy given their own knowledge and all the available modeled information. Each of the four resulting strategies was modeled and the average budget distribution is used to describe the expert strategy. It was clear that the EDT model output had a strong impact on the strategies suggested by the panel of experts. The experts suggested floodplain restoration actions in the lower parts of the watershed and riparian restoration in the upper reaches of the East Fork Lewis River. Road decommissioning dollars were spent across most of the upper North Fork Lewis River with the Muddy and Clear HUs targeted for extra effort (Figure 10f). A total of $2,023,894 was spent under the expert strategy.

Modeling future landscapes

The second step in the analysis is to translate watershed management strategies into specific restoration and protection actions. For each of the watershed management strategies, specific actions such as road decommissioning or riparian planting were identified and spatially located. The impacts of these actions were modeled (Table 4, Appendix D) and a future landscape was created (Figure 11 through Figure 23, top rows). The effect and cost of all actions were modeled using an instantaneous 50-year time step. For example, the predicted benefits of riparian restoration included 50 years of tree growth. These 6 watershed management strategies resulted in 13 potential future landscapes because the GIS-based assessment strategy and the expert strategy each included multiple future landscapes that were modeled independently.

Evaluating physical and biological response to watershed management strategies

In the third step of our analysis, we quantify habitat quality and distribution and we predict the biological response to these habitat conditions. Eight landscape evaluation models we applied to each of 13 potential future landscapes. Most models provided multiple evaluation metrics. The landscape evaluation models are summarized below.

Riparian condition – Three riparian functions are estimated from remotely sensed vegetation data (Table 4; Appendix H). Model outputs include assessments (good/fair/poor) for shade function, potential large woody debris recruitment, and potential recruitment of pool-forming conifers. Our large-woody debris recruitment model was customized for our application to incorporate deciduous trees, which historically dominated the floodplain and riparian landscape (Rice 1996).

Sediment – The sediment yield model predicts annual yield (kg/yr) of surface, road, and mass wasting sediment delivered to each stream reach (Appendix E). 1) Surface sediment yield was generated through a modified Water Erosion Prediction Procedure (WEPP) model for each 30x30 m pixel in the watershed. Variables used in the WEPP model were land cover, topography (slope) and soil texture (Flanagan and Livingston 1995; Lane et al. 1989). 2) Field data on road sediment yield (PWI 1998, PacifiCorp 2002) was supplemented with data generated through two U.S. Forest Service models, WEPPROAD and XDRAIN (Elliot et al. 1995, Elliot and Hall 1997). Road sediment yields were estimated for all road surfaces and prisms based on underlying soil, road slope, riparian condition, and distance from streams. Riparian condition was used to modify surface and road sediment delivery to streams. On ash soils, fair or good riparian conditions reduced sediment inputs to the stream by 38% and, on non-ash soils, they reduced sediment inputs by 45%. 3) Mass wasting sediment yield was predicted from modified published GIS-based slope stability models (Shaw and Vaugeois 1999, Montgomery and Dietrich 1994).

The modifier variables included soil characteristics, road density, and land cover in adjacent hillslopes.

Hydrology – The hydrology model estimates annual storm runoff (mm H_2O/yr) draining into each reach and 2.3-year flood discharge (cms) for each reach (Appendix E). 1) The Water Erosion Prediction Procedure (WEPP) model was used to estimate the mean annual surface and shallow subsurface storm runoff in the watershed for each 30 x 30 m pixel (Flanagan and Livingston 1995; Lane et al. 1989). Variables used in the model were land cover, topographic slope, and soil texture. As in the sediment model, riparian condition was used to modify surface and road sediment delivery to streams. On ash soils, fair or good riparian conditions reduced runoff volume by 38% and, on non-ash soils, they reduced runoff volume by 45%. 2) The 2.3-year recurrence-interval flood discharge was estimated for each stream reach based on published relationships between gauge data, drainage area, bankfull width and depth, and land use and cover (Dunne and Leopold 1978, Black 1991, Moscrip and Montgomery 1997). The 2.3-year flood was used as an indicator of the mean annual flood and channel forming flow. Flood frequency and sediment transport analysis in the Lewis watershed indicated that the 2.3 year flood is the average flood that initiates bedload transport (PWI 1998, PacifiCorp 2002).

Sediment and hydrology routing – Lateral sediment and runoff delivered to each reach were cumulatively routed through all downstream reaches using the 2.3-year flood as the channel forming flow (Appendix F). The customized routing model provided information on source of sediment and stream response to sediment inputs. Gross morphologic indicators of drainage area, channel gradient, and valley width were used to delineate broad channel types and identify potential zones of transport and deposition (e.g., Montgomery and Buffington 1997). The routine uses a series of variables to estimate the deposition of sediment including contributing area per segment, flood discharge modifications, empirical models for bed textures and fines, estimates of sediment yield per stream reach in (kg/yr) and bed scour. Channel sediment size field data (unpublished data from Jen Burke (University of Washington, Seattle WA), USFS 1999, PWI 1998) and size classes of incoming sediment estimated from SSURGO database (NRCS 2004) and landslide surveys (unpublished data from Earth Systems Institute, Seattle WA) were used to predict the amount of fine sediment deposited, and an index of bed scour for unmodified and current conditions for each reach. The reservoirs were treated as sediment and flow sinks; sediment and 2.3-year flood flows were reset to base level for stream reaches immediately downstream of the dams. Output metrics from the routing model include fine and coarse sediment (by source) entering each reach laterally and from upstream; % fine sediment deposited; and an index of bed scour.

FishEye – FishEye is a logical model that combines habitat preferences (stream gradient, bankfull width, sediment deposition, bed scour, and hydrologic regime) by species based on published fish-habitat relationships (Beechie et al. 2006; Burnett 2003; WDNR 1991; Montgomery et Al. 1999; Salo 1991; WDFW 2000; WFPB 2000) (Appendix J). FishEye output metrics include species-specific natural habitat suitability ratings that include only the factors that are generally not modified by human actions (gradient, stream width, and hydrologic zone) and species-specific observed habitat suitability ratings for both current and future conditions that also include habitat factors impacted by management (riparian, sediment, and bed scour).

Remotely sensed suitability and capacity – A logical model that combines bankfull width, stream gradient, and seral stage of riparian areas using data from field studies describing how spawners respond to these habitat conditions (Beamer et al. 2000; Lunetta et al. 1997; Beth Sanderson,

NW Fisheries Science Center, 2725 Montlake Blvd E, Seattle, WA 98112 – Personal Communication)(Appendix I). Model output metrics include habitat suitability ratings (good/fair/poor) and spawner capacity estimates for chinook salmon.

Ecosystem Diagnosis and Treatment (EDT) – EDT is a proprietary, habitat suitability model employed in most watershed-scale habitat recovery-planning projects in the region. (Mobrand Biometrics Inc., 2004). Model output metrics used to evaluate future landscapes include watershed-scale productivity, capacity, and equilibrium abundance estimates for chinook salmon.

Sediment and survival – Sediment and survival models are statistical relationships between modeled fine sediment deposited in a reach and egg-to-fry survival, based on a compilation of published studies (Appendix K). Published sources included Bennett et al 2003; Hall 1986; Reiser and White 1988; Tappel and Bjornn 1983 for chinook salmon; Cederholm and Lestelle 1974; Tappel and Bjornn 1983 for steelhead; Cederholm and Salo 1979; Hall 1986; Hall and Lantz 1969; Reiser and White 1988; Tagart 1984 for coho salmon. Model output metrics include egg-to-fry survival estimates and confidence intervals for chinook, steelhead, and coho salmon.

Synthesizing the output of the decision support system

In the fourth step of the decision support system, we synthesize modeled predictions of future physical habitat conditions and of potential biological response to each of the 6 watershed management strategies (Table 5).

Continuous data (e.g., flood discharge, % fine sediment deposited, survival, spawner capacity) were evaluated as percent change from current conditions. For certain results, we also calculated total improvement (as sums of reach-specific values) and average values weighted by reach length. Other data were represented as indices (good, fair, poor) of conditions and so were categorical. For these data (e.g., riparian function scores, FishEye habitat suitability scores), we calculated change from current conditions in terms of km. We quantified km where scores improved at all as well as where conditions improved to the best possible score, and also where conditions were degraded to those worse than current conditions (primarily effects of strategies incorporating effects of future land use trends). For comparison to categorical data metrics, we calculated km where continuous data metrics improved or were degraded.

We summarized results for individual reaches into a series of watershed-scale evaluation metrics. For sediment, hydrology, and riparian results, we summarized metrics over all reaches in the watershed. For habitat suitability, spawner capacity, and egg-to-fry survival, we summarized metrics over reaches currently accessible to winter steelhead (the most far-ranging species). Metrics fall into several general categories: 1) km improved; 2) km newly accessible; and 3) EDT outputs. We calculated km improved as the length (in km) of all reaches in a spatial extent (i.e., entire watershed or fish-accessible) where conditions improved due to the effect of an action. Habitat suitability improvements included increases in both quantity and quality of habitat. Newly accessible habitat was summarized for each species as km opened by barrier improvements or floodplain restoration. For strategies with more than one modeled future landscape, outcome metrics were averaged. Although the potential for salmon reintroduction above the dams was not modeled explicitly, we quantified potential future habitat conditions over the area that would become accessible to salmon under such a scenario to provide estimates of potential habitat in those areas (e.g., sediment inputs). Limitations on available data prevented us from applying all models to areas above the dams.

Results

Each watershed management strategy resulted in a unique distribution of habitat changes, which could be traced to the spatial distribution of actions (Figure 11 through Figure 23). Because changes in sediment and hydrology were routed downstream, habitat changes could also be detected in downstream subwatersheds. These habitat changes were captured in a suite of habitat outcome metrics (Table 5). Biological response to these habitat changes was predicted using the biological response models described above and captured in a suite of biological outcome metrics (Table 5). Selection of the best strategy with respect to increases or improvements in suitable habitat was relatively constant across species except for chum salmon.

No one watershed management strategy performed best with respect to all of the habitat or biological response metrics (Table 5). The strategy emphasizing actions in the upper watershed, the federal strategy, performed best with respect to reductions in flood discharge (Table 5, Figure 14) and some types of sediment input (Table 5, Figure 14). However, the federal strategy ignored downstream habitats which may have higher potential suitability and which are currently accessible to fish (Table 5, Figure 14). The EDT strategy, which spent the most money on riparian restoration and protection, performed best with respect to some riparian functions, shade and large-woody debris recruitment, (Table 5, Figure 13) and provided the most dramatic reductions in lateral hydrologic flow volumes (through the riparian modifier on lateral flow volume). It focused almost completely on mainstem reaches in the lower watershed and the resulting future landscapes showed little improvement with respect to increases in accessible habitat or reductions in sediment delivery in the upper watershed. The barriers strategy (Figure 11), which opened up only 9 barriers, performed extremely well with respect to improvements in suitability, accessibility, and capacity for multiple species. This strong performance was due to instant new habitat in two lower subwatersheds; however, the rest of the Lewis River watershed and large-scale habitat processes such as sediment delivery, hydrologic function, and riparian condition were unchanged. The barriers and riparian strategy balanced the strengths of opening up some new habitat in the lower watershed with riparian improvements throughout the watershed. The landscape and expert strategies, which averaged several future landscapes, had the widest spectrum of restoration and preservation actions (Table 5, Figure 15 through Figure 23). These strategies tended to balance performance on habitat and biological metrics and rarely had the best or worst performance on any one metric (Table 5).

The largest gains, across all 6 watershed management strategies, for sediment included 56-58 km of stream with a reduction in locally-derived surface sediments (EDT, landscape, and expert strategies), 90 km of stream with reduced sediment inputs from mass wasting (federal strategy), and 717 km of stream with reduced road-derived sediment (federal strategy). The largest length of stream with a reduction in flood magnitude was 352 km (federal strategy). The longest gain in riparian conditions was about 27 km of newly improved habitat (EDT strategy). Maximum km of new or improved habitat suitability (FishEye) was about 38 km for all species (barriers strategy) except chum, which had a slightly larger increase in suitability (17 km) with the EDT strategy. The maximum length of stream with an increase in remotely sensed spawner capacity estimates was 38 km (barriers strategy). The maximum length of stream improved with respect to egg-to-fry survival was 97 km for steelhead, chinook, and coho salmon. The maximum increase in accessible stream distance within the historical species range was only 10.7 km for chum salmon but approximately 38 km for the other modeled species (barriers strategy). The maximum fall chinook salmon capacity predicted by EDT was 25,102 fish (EDT strategy) for the basin;

however, this was within 700 fish of the minimum fall chinook capacity across all 6 watershed management strategies (Table 5). Because of the number and, in some cases, complexity of the models used in this analysis, confidence intervals for these estimates are not yet available.

The largest gains in riparian function were achieved using the EDT strategy (Table 5). These gains were not consistent across all 3 riparian functions. The EDT strategy outperformed the other strategies with respect to large-woody debris recruitment and shade function but the differences in recruitment of pool-forming conifers was less dramatic between strategies. Likewise, the federal strategy did much better at reducing road sediment and mass wasting sediment but, because all riparian areas on federal lands are already protected, no riparian restoration or protection actions were added and the federal strategy showed no improvement in surface-derived sediment. The EDT strategy, emphasizing riparian protection and restoration, showed the largest lengths of stream with improved surface-derived sediment (Figure 13) but these were quite similar to improvements observed with the landscape and expert strategies.

Discussion

Our decision support framework provides the essential predictions necessary for identifying the best watershed management strategy. No one strategy will maximize all possible outcomes. But, by examining multiple metrics representing large-scale landscape processes, local habitat conditions, and predicted fish response, the best strategy or combination of strategies for meeting a particular set of goals can be selected. We provide managers with tools for examining the more certain habitat impacts at the same time as the less certain biological response predictions. Each model has inherent inaccuracies, imprecision, and biases. Because of these model limitations, experts, modelers, and decision-makers have demanded a reduced reliance on individual models (Burgman et al. 2005). By integrating multiple models, we provide robust predictions on which to make decisions. The final strategy selection will require subjective decision-making based on local habitat knowledge, current population status of all affected species, insights about local model accuracy, social values, and risk tolerances.

These analyses provide technical guidance for managers working in the Lewis River watershed. Managers may choose to combine the strategies modeled here and to develop a customized strategy given interest in a particular species or area. The results presented here can provide quantitative insights for designing customized strategies. Our results also provide managers with crude estimates of potential fish response given 2 million dollars of restoration projects.

Our approach also provides guidance for other watersheds in how we structured of the problem. Estimates of potential future conditions given particular spending plans will always be useful planning tools. By explicitly comparing the predicted biological response to various spending plans, managers can choose the spending plan that maximizes their goals whether the goals are to increase juvenile survival of a particular species or to balance increases in new habitat for multiple species.

As in any modeling effort, assumptions are built into the final outcomes. We have tried to make these assumptions transparent and future research will include sensitivity analyses of key parameters. Any implementation of these modeled results in the Lewis River or other watersheds should consider the potential impacts of model assumption. Effects of some restoration actions are better captured by any one of our evaluation models than by the others. In-stream restoration, for example, can only be modeled by the remotely-sensed capacity model and the EDT model. In

predicting the impacts of any particular type of restoration action or in comparing effects of alternative restoration actions, the ability of the evaluation models to detect those actions should be considered.

The value of the decision support system is in the identification of realistic alternatives, the estimation of potential outcomes, and the organization of that information. By providing suites of predictions about the performance of multiple watershed management strategies, there is objective information on which to base critical management decisions. The process increases accountability in decision-making while allowing subjective information such as belief in outcome from particular models or willingness to take certain kinds of risks. Users of this type of decision support system can make explicit trade-offs between spatial allocation of funds or allocation between actions that might benefit particular species or habitat types. These trade-offs are transparent to those impacted by the decision or tasked with implementing the watershed management strategy. The use of multiple models increases the robustness of the decision-making process and reduces reliance on any one model. Tools for making robust and transparent trade-offs will be essential as pressure to balance the competing habitat needs of multiple species increases.

Figures and Tables

Table 1. Geographic information system (GIS) datalayers used in the DSS. Most datalayers were modified slightly from their original source for this analysis. The source column includes an acronym for the agency providing the data and the year of the data layer used in our analysis. Data processing notes are included in the description column. Full data references are included in the literature cited.

Data Source	Source	Description	Resolution
Sediment			
Soils on U.S. Forest Service land	USFS (1999)	U.S. Forest Service (Gifford Pinchot National Forest) forest soils and soil map units.	1:15,840
Soils on state, county, and private lands	NRCS (2003-2004)	USDA Natural Resources Conservation Service Soil Survey Geographic Database (SSURGO) for Cowlitz, Clark, and Skamania counties.	1:250,000
Hydrology			
Stream hydrography (routed)	SSHIAP (2004)	Salmon and Steelhead Habitat Inventory and Assessment Program (SSHIAP) unpublished spatial data on hydrology and stream conditions (1:24,000) for Watershed Inventory Area (WRIA) 27.	1:24,000
Stream hydrography (drainage enforced, routed)	Miller (2003)	Routed, cleaned and attributed stream hydrography generated to match SSHIAP hydrography following methods by Miller (2003). Generated to facilitate sediment routing and estimation of channel characteristics.	1:24,000
6th Field Hydrologic Unit boundaries (HUCs)	BLM (2002)	Regional Ecosystem Office (REO) Hydrologic Unit Boundaries for Oregon, Washington, and California. Portland, Oregon.	1:24,000
7th Field Hydrologic Unit boundaries (HUCs)	Lewis Co (2000)	Lewis County GIS (2001) data on 7th field hydrologic boundaries for the Lewis watershed.	unknown
Topography and Geology			
Surficial Geology	WDNR (2003)	Washington State Department of Natural Resources (WDNR) classification of geologic map units according to major lithology (WDNR 2003).	1:100,000
Slope stability	WDNR (2000)	WDNR predictive data layer of shallow-rapid slope stability from calibrated GIS-based models. Updated for the Lewis watershed using methods by Shaw and Vageois (1999).	1:24,000
Elevation	USGS (2003)	USGS 10 m drainage enforced Digital Elevation Model (DEM). Multiple DEMs mosaicked, and used to generate hydrographic stream layer, to associate streams with topographic features, and to generate lateral hillslope watersheds for stream segments.	1:24,000
Hillslope	USGS (2003)	Hillslope gradient calculated for every 10 m gridcell in the mosaicked 10 m drainage enforced DEM, using ARC/INFO.	1:24,000
Barriers			
SSHIAP barriers	WDFW (2004)	Salmon and Steelhead Habitat Inventory and Assessment Program (SSHIAP) unpublished data on fish passage barriers (1:24,000) for Watershed Inventory Area (WRIA) 27.	1:24,000
Dams	BPA (2001)	Bonneville Power Administration (BPA) dams and possible hydroelectric development sites (BPA 2001). Original source database converted to a spatial data layer.	1:100,000
Political			
Regional ownership	ICBEMP (1995)	Interior Columbia Basin Ecosystem Management (ICBEMP) regional land ownership.	1:100,000
Parcel ownership	Clark Co	Land ownership, parcel boundaries, and land use for Clark	1:24,000

	(2004)	County.	
Parcel ownership	WDNR (2005)	Land ownership, parcel boundaries, and land use statistics for Clark, Cowlitz, and Skamania counties.	1:24,000
County ownership	CommEn Space	Washington Protected Lands Database (PLDB) that includes spatial location and conservation status for private and public lands. (PLDB 200?) http://protectedlands.org.	-
Urban growth	Clark Co (2004)	Urban growth boundary for Clark County.	-
Land use	Clark Co (2004)	Comprehensive plan and land use/zoning for Clark County.	-
Vegetation			
Land cover and forest cover	IVMP (2001)	Interagency Vegetation Mapping Project, Western Cascades (version 2.0) and Western Lowlands (version 1.0) Spatial Data, 1996 (BLM 2001).	30 meter
National Land Cover Data	USGS (1999)	USGS classification of land cover data from LANDSAT TM satellite imagery (level 2). Generated by USGS using Anderson et al. (1976) protocols.	30 meter
Fish Distribution			
Fish distribution	WDFW (2004)	Washington Department of Fish and Wildlife (WDFW) Salmon and Steelhead Habitat Inventory and Assessment Program (SSHIAP) data on fish distribution for Watershed Inventory Area (WRIA) 27.	1:24,000
Transportation			
Roads	GP (1995)	Forest roads and associated attributes.	1:24,000
Roads	WDNR (2005)	WDNR transportation data layer of roads, railroad, and other land and water transportation routes within Clark, Cowlitz, and Skamania counties.	1:24,000

Table 2. (a) Linear model to estimate the log-transformed bankfull width for streams in the Lewis River watershed, with the exception of small watershed (< 1.43 km^2) and those impacted by volcanic activity (p<0.0001, AIC=906.1, AIC null model = 1982.8). (b) Alternate model for small watersheds not impacted by volcanic activity (p<0.0001, AIC=1714.0, AIC null model = 2548.3). (c) Alternate model to predict bankfull width in the volcano-impacted reaches on the north side of the watershed (e.g., Muddy Creek, Clear Creek) (p<0.0001, AIC=373.1, AIC null model = 405.2). In all cases, drainage area is watershed area above the reach in km^2, and precipitation is the cumulative annual precipitation in mm. Details are provided in Appendix G.

(a)

Variable	Coefficient	Standard error	t-Statistic	p-Value
intercept	3.43	1.77	1.94	0.053
log (drainage area)	-5.20	1.23	-4.24	< 0.0001
log (drainage area)^2	0.94	0.20	4.83	< 0.0001
log (precipitation)	-0.23	0.23	-1.01	0.315
log (drainage area) * log (precipitation)	0.68	0.16	4.35	< 0.0001
log (drainage area)^2 * log (precipitation)	-0.12	0.025	-4.63	< 0.0001

(b)

Variable	Coefficient	Standard error	t-Statistic	p-Value
intercept	1.65	0.062	26.54	< 0.0001
log (drainage area)	0.28	0.035	7.79	< 0.0001
log (drainage area)^2	0.018	0.0046	3.98	< 0.0001

(c)

Variable	Coefficient	Standard error	t-Statistic	p-Value
intercept	-12.30	6.03	2.04	0.043
log (drainage area)	0.22	0.049	4.35	< 0.0001
log (precipitation)	-1.20	0.76	-1.59	0.113

Table 3. Restoration action and economic models. Possible restoration and preservation actions are identified in the first column. The landscape impact column describes how the action was implemented on the landscape in our modeling framework (Appendix D). A description of how each modeled action was translated into EDT input data is found in Appendix L. The economic model column describes the cost estimated for each action type.

Restoration or preservation action	Economic model	Modeled landscape impact
Culvert removal	C = 178,430*ln(1.2W+0.61)-34,773 based on data from Evergreen Funding Consultants (2003).	Upstream reaches reclassified as passable, provided that they were historically accessible to fish.
Riparian protection	Forest lands: Cost of lost riparian timber production = $10,000 per acre. Non-forest Lands: Cost of acquisition (C/acre) depends on parcel size and current land-use designation: forested (40-80 acre plot) = $7,080; forested (>80 acre plot) = $2,856; open space = $10,730; agriculture (min 20 acre plot) = $6820; rural (< 5 acre plot) = $16,997; rural (5-10 acre plot) = $14,456; rural (10-20 acre plot) = $11,064; rural (min 20 acre plot) = $7,966; urban residential = $40,344; urban commercial = $39,199. Note: Riparian areas were protected to 60 m however the costs were only calculated for the fraction of the riparian area not currently protected by county, state, or federal riparian ordinances.	Riparian functions and seral stage ↑ by one level (where possible to improve), and riparian land cover was re-classified to 20-yr forested. This reduced the amount of sediment and hydrologic runoff entering the reach.
Riparian planting	Riparian planting only occurred on areas where costs were not prohibitive. These included reaches for which >= 35% of the area within 20 m of the channel was < 5% hillslope and >= 50% of the area within 20 m of the channel was not in bare ground, shrubs, or short grass. The cost for riparian planting was estimated as C/acre = $15,000 (slope <0.05).	Riparian functions and seral stage ↑ to the best possible level, and riparian land cover was re-classified to 20-yr forested. This reduced the amount of sediment and hydrologic runoff entering the reach.
In-stream restoration	C/km = $78,593	Improved spawner capacity in reach by adjusting input variables. Small streams (BFW ≤25 m): redds/km ↑ to 90th percentile of estimated current values. Large streams (BFW >25 m): spawnable area ↑ by 32%.
Floodplain restoration	C/ stream km = $155,507	Increased length of reach by 39.4% to represent inclusion of historical side channels, as determined from aerial photographs (Appendix M). Habitat conditions were inherited from existing reach (may have been

		modified by other actions). An outline of the floodplain for the Lewis River watershed (WDFW 2003) was used to identify segments appropriate for side channel restoration unless specifically identified in the landscape and expert strategies. All mainstem North Fork, East Fork, and Upper North Fork segments within the floodplain boundaries were considered, as well as tributaries that were within the extent of the floodplain.
Road decommis-sioning	C/road km = $12,427	Reduced length of existing roads by 95% in areas draining to reach; thereby reducing sediment input.
Road repair	C/road km = $6,214	Reduced length of existing roads by 50% in areas draining to reach; thereby reducing sediment input.

C = Project cost in U.S. dollars, W = Channel width in meters.

Table 4. GIS-based models to evaluate the current landscape and generate a watershed management strategy. The GIS-based riparian model was also used to evaluate future landscapes. Abbreviations: g/f/p = good/fair/poor ratings; Δ = change.

Model	Model description	Output metrics
GIS-based Riparian condition (Appendix H)	Logical model that combines bankfull width, elevation (from DEM), stream gradient, and estimates of riparian vegetation cover (total cover and % coniferous vs. deciduous) and tree size (dbh) to predict qualitative riparian conditions within 60 m of each bank (BLM 2001; FEMAT 1993; Lunetta et al. 1997; Montgomery et al. 2003; WFPB Assessment Method Riparian Module 1997). Shade and large woody debris models were modified from WFPB method, and the pool-forming conifer model was based on Montgomery et al. 2003, Beechie et al. 2000; Buffington et al 2002.	Shade (g/f/p); Pool-forming conifer potential (g/f/p); Large woody debris recruitment (g/f/p); Seral stage (late, mid, early, mixed, deciduous, nonforested)
GIS-based sediment	GIS-based assessment of relative differences in estimated historical and current sediment budgets. Forested area budgets based on roads, mass wasting, area in clearcuts, hillslopes, and erosion rate studies and USFS modified WEPP (Elliot et al. 1995). Agricultural area budgets based on the modified universal soil loss equation (RUSLE) using soil erosivity, slope, and land use and land cover (Beechie et al. 2004; Flanigan and Livingstone 1995).	6th field HU summaries of annual yield (kg/yr); % Δ between estimated historical and current conditions
GIS-based hydrology	GIS_based assessment of relative differences in estimated historical and current runoff estimated from land cover, land slope, soil texture using the modified WEPP in forested areas (Elliot et al. 1995) and WEPP in agricultural areas; and coarser scale impact ratings based on forested areas: % immature vegetation and road density; lowland areas: % impervious areas (Beamer et al. 2000; Booth and Jackson 1997; Dinicola 1989; Lunetta et al. 1997).	6th field HU summaries of % impaired due to impervious areas; % Δ between estimated historical and current conditions

Table 5. Results for selected metrics used to evaluate future impacts of each of the 6 watershed management strategies. Evaluation metrics are summarized over all reaches in the watershed for sediment, hydrology, and riparian metrics, and over reaches currently accessible to winter steelhead for habitat suitability, spawner capacity, and egg-to-fry survival; newly accessible habitat is summarized for each species. The maximum potential change column describes the difference between estimates for current and historical conditions, the maximum improvement in habitat condition or biological response that could be expected with infinite resources. Habitat suitability increases include increases in both quantity and quality of habitat.

Evaluation Metric	Barriers	Bar./Rip.	EDT	Federal	Landscape	Expert
Sediment						
Surface-derived						
km where laterally-derived[1] surface sediment ↓[f]	0.0	4.7	27.7	0.0	11.6	11.5
km where locally-derived[2] surface sediment ↓[f]	0.0	14.6	58.2	0.0	56.1	56.7
total coarse surface sed. entering reach (kg/yr)[a]	6,460,211	6,459,514	6,456,874	6,460,211	6,431,125	6,444,927
total fine surface sediment entering reach (kg/yr)[a]	13,483,457	13,481,908	13,470,295	13,483,457	13,427,850	13,461,880
% Δ in fine surface sediment entering reach[c]	0.00	-0.01	-0.10	0.00	-0.41	-0.16
Mass wasting-derived						
km where laterally-derived[1] MW sediment ↓[f]	0.0	0.79	0.0	9.3	3.0	2.2
km where locally-derived[2] MW sediment ↓[f]	0.0	5.8	7.6	90.3	42.0	29.5
total coarse MW sediment entering reach (kg/yr)[a]	249,304,560	249,274,656	249,301,086	247,761,947	248,670,368	248,839,015
total fine MW sediment entering reach (kg/yr)[a]	27,660,785	27,657,467	27,660,400	27,489,629	27,590,420	27,609,132
% Δ in fine MW sediment entering reach[b]	0.00	-0.01	0.00	-0.62	-0.25	-0.19
Road-derived						
km where laterally-derived[1] road sediment ↓[f]	0.0	0.0	70.3	256.9	142.6	98.9
km where locally-derived[2] road sediment ↓[f]	0.0	0.0	105.9	716.7	457.8	239.5
total fine road sediment entering reach (kg/yr)[a]	44,563,365	44,563,365	44,475,772	26,646,625	36,582,219	41,184,630
% Δ in fine road sediment entering reach[b]	0.00	0.00	-0.20	-40.21	-17.91	-7.58
Fine Sediment						
km where % of fine sediment entering reach ↓[f]	0.0	7.0	108.0	710.5	442.9	229.6
total locally-derived[2] fine sediment (kg/yr)[a]	12,620	12,621	12,614	11,926	12,356	12,489
km where % fine sediment deposited in reach ↓[f]	0.0	5.9	101.7	705.2	424.8	215.2
% Δ in % fine sediment deposited in reach[b]	0.00	0.00	-0.01	-2.21	-0.88	-0.45
total km where fines deposited <10%[d]	907.4	907.4	912.1	932.2	917.2	915.2
km where fines deposited is newly <10%[e]	0.0	0.0	4.7	24.8	9.8	7.8
Hydrology						
total laterally-derived hydrologic runoff (m/yr)[a]	87,106,047	87,090,825	86,844,517	87,106,047	86,702,319	86,904,968
% Δ in laterally-derived hydrologic runoff[b]	0.00	-0.01	-0.31	0.00	-0.50	-0.19
km where hydrologic runoff entering reach ↓[f]	0.0	4.7	29.0	0.0	11.7	11.9
weighted mean 2.33-yr flood discharge[c]	21.22	21.22	21.24	21.21	21.32	21.30
% Δ in 2.33-yr flood discharge[b]	0.00	0.00	0.00	-0.04	-0.03	-0.03
km where 2.33-yr flood discharge ↓[f]	0.0	5.0	6.2	352.2	199.1	105.2
weighted mean bed scour index[c]	0.09	0.09	0.09	0.09	0.09	0.09

% Δ in bed scour index [b]	0.00	0.00	-0.02	-1.39	-0.61	-0.32
total km where bed scour index <0.0587 [d]	2,546	2,546	2,547	2,596	2,569	2,557
km where bed scour index is newly <0.0587 [e]	0.0	0.0	0.7	49.7	23.3	10.8
km where the index of bed scour ↓ [f]	0.0	8.6	97.4	750.5	470.2	231.3
Riparian						
total km where shade score is "good" [d]	4,146.1	4,146.1	4,165.5	4,146.1	4,153.4	4,153.8
km where shade score is newly "good" [e]	0.0	0.0	19.3	0.0	7.2	7.7
km where shade score has ↑ [f]	0.0	0.0	19.3	0.0	7.3	8.5
total km where PFC score is "good" [d]	132.8	132.8	138.2	132.8	140.4	137.3
km where PFC score is newly "good" [e]	0.0	0.0	5.5	0.0	7.6	4.5
km where PFC score has ↑ [f]	0.0	0.0	5.5	0.0	8.3	5.4
total km LWD score is "good" [d]	2,054.8	2,054.8	2,082.2	2,054.8	2,064.0	2,064.0
km where LWD score is newly "good" [e]	0.0	0.0	27.2	0.0	9.2	9.5
km where LWD score has ↑ [f]	0.0	0.0	27.2	0.0	9.6	10.3
total km where all 3 riparian scores are "good" [d]	1,920.3	1,920.3	1,947.6	1,920.3	1,929.0	1,930.3
km where all 3 riparian scores are newly "good" [e]	0.0	0.0	27.2	0.0	8.7	10.0
km where all 3 riparian scores ↑ [f]	0.0	0.0	27.2	0.0	9.1	10.7
total km where seral stage is "late" [d]	1,122.1	1,122.1	1,122.2	1,122.1	1,123.2	1,122.2
km where seral stage is newly "late" [e]	0.0	0.0	0.1	0.0	1.1	0.6
km where seral stage ↑ [f]	0.0	0.9	27.0	0.0	7.2	7.7
Habitat Suitability						
km predicted to be "good" for chum [d]	24.6	24.6	39.5	24.6	25.0	26.3
km where suitability ↑ for chum [f]	0.0	0.0	16.9	0.0	1.3	2.7
km predicted to be "good" for spring chinook [d]	31.1	31.0	34.5	29.1	29.9	31.0
km where suitability ↑ for spring chinook [f]	0.0	0.0	12.1	0.0	1.1	2.6
km predicted to be "good" for fall chinook [d]	17.7	17.7	28.0	17.7	18.1	18.3
km where suitability ↑ for fall chinook [f]	0.0	0.0	18.8	0.0	1.2	2.5
km predicted to be "good" quality for win. stlhd. [d]	58.1	58.0	63.3	55.7	56.7	57.4
km where suitability ↑ for winter steelhead [f]	0.0	0.0	12.1	0.0	1.2	2.6
km predicted to be "good" quality for sum. stlhd. [d]	48.8	48.7	54.1	46.4	47.4	48.4
km where suitability ↑ for summer steelhead [f]	0.0	0.0	12.1	0.0	1.1	2.6
km "good" quality for chinook spawning [d†]	116.1	116.1	116.3	116.1	117.3	117.0
km newly "good" for chinook spawning [e†]	0.0	0.0	0.3	0.0	1.3	0.9
Spawner Capacity (Chinook)						
km where capacity ↑ (reach quality) [f]	0.0	0.0	14.8	0.0	9.2	5.3
km where capacity ↑ (reach qual. & new habitat) [g]	38.0	22.3	15.2	0.0	13.7	6.6
total spawner capacity (mean) [a]	96,642	95,894	96,229	95,271	96,491	96,300
total spawner capacity (10th percentile) [a]	26,587	26,492	26,737	26,413	26,779	26,739

total spawner capacity (90th percentile) [a]	403,999	402,107	404,709	400,532	405,960	405,320
weighted mean spawner capacity per reach [c]	223.9	229.2	240.0	237.6	242.3	243.4
% Δ in spawner capacity [b]	1.4	0.7	1.0	0.0	1.3	1.1
Egg-to-Fry Survival						
km where chinook/steelhead survival ↑ [f]	0.0	0.0	96.6	0.0	73.0	58.0
km where coho survival ↑ [f]	0.0	0.0	96.5	0.0	70.6	56.1
weighted mean chinook/steelhead survival [c]	0.12	0.12	0.13	0.13	0.13	0.13
weighted mean Chin./sthd. survival CI (5 – 95%) [c]	0.10 – 0.16	0.10 – 0.16	0.10 – 0.17	0.10 – 0.17	0.10 – 0.17	0.10 – 0.17
weighted mean coho survival [c]	0.27	0.28	0.29	0.29	0.29	0.29
weighted mean coho survival CI (5 – 95%) [c]	0.22 – 0.32	0.23 – 0.33	0.24 – 0.35	0.24 – 0.34	0.24 – 0.34	0.24 – 0.34
%Δ in weighted mean Chin./sthd. survival [b]	-5.9	-3.4	2.0	0.0	0.2	0.4
%Δ in weighted mean coho survival [b]	-5.8	-3.3	1.6	0.0	0.2	0.4
EDT Outputs (current w/out harvest; fall chinook)						
Capacity [a]	24370	24370	25102	24370	24402	24489
Equilibrium Abundance [a]	22367	22367	23305	22367	22406	22523
Productivity [a]	26.89	26.89	30.83	26.89	26.99	27.62
Accessibility						
km accessible to chum [d]	192.0	186.5	181.7	181.3	182.5	182.3
km newly accessible to chum [f]	10.7	5.2	0.4	0.0	1.2	1.0
km accessible to coho [d]	527.0	511.3	489.4	489.0	493.6	490.3
km newly accessible to coho [f]	38.0	22.3	0.4	0.0	4.5	1.3
km accessible to spring & fall chinook [d]	527.0	511.3	489.4	489.0	493.6	490.3
km newly accessible to spring & fall chinook [f]	38.0	22.3	0.4	0.0	4.5	1.3
km accessible to winter & summer steelhead [d]	580.5	564.8	542.4	542.0	547.1	543.3
km newly accessible to both steelhead [f]	38.5	22.8	0.4	0.0	5.0	1.3

Bar./Rip. = Barriers & Riparian management strategy; MW = mass wasting; PFC = pool-forming conifers; LWD = large woody debris; CI = confidence interval; Δ = change; ↑ = an increase in the value over current conditions; ↓ = a decrease in the value. Fine sediment = 0.25 to 1.0 mm; coarse sediment = ≥ 4.8 mm.
[1]Laterally-derived sediment = sediment entering a reach from adjacent hillslopes;
[2]locally-derived sediment = sediment entering a reach from adjacent hillslopes and from upstream reaches.
[†]Modeled using the remotely-sensed spawner suitability model (Appendix I); other suitability indices modeled using the FishEye model (Appendix J).
Lowercase superscripts indicate equations used to calculate each metric: *a*: Total = ∑(value); *b*: %Change = ∑(value, strategy) - ∑(value, "current" conditions) / ∑(value, "current" conditions) x 100; *c*: Weighted Mean = ∑(value * reach length)/∑(reach length); *d*: Km Good = ∑(reach length) where value = "good;" *e*: Km Newly Good = (equation *d*, strategy) - (equation *d*, "current" conditions); *f*: Km Improved = ∑(reach length) for reaches where new value > current conditions value; and *g*: a combination of equation *f* for quality improvements and equation *d* for improvements in quantity. For categorical metrics, equation *f* requires one level of improvement in score to be counted whereas for continuous metrics, the required level is 0.1% better than current conditions. For accessibility, equation *d* is used where "accessible" substitutes for "good."

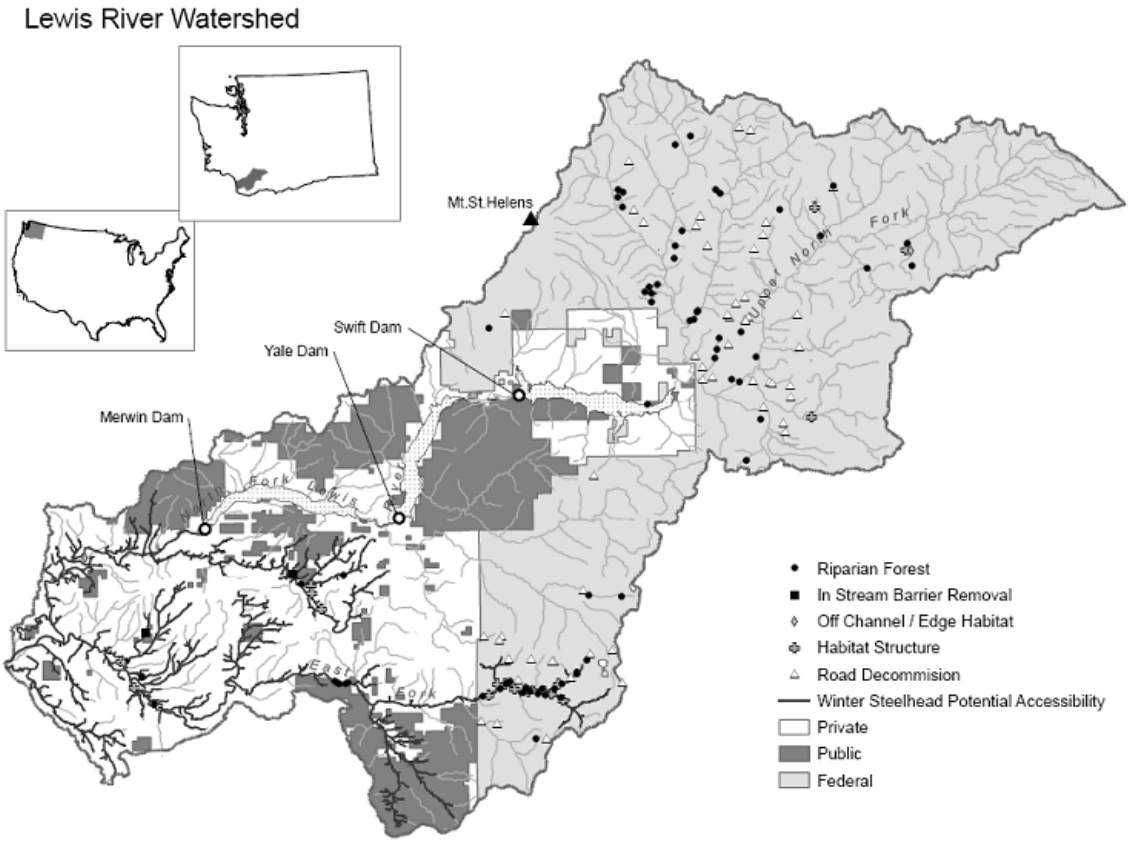

Figure 1: The Lewis River watershed and its location in SW Washington State, USA. The estimated linear extent of streams and rivers accessible to winter steelhead is identified with a thick line. Key disturbance elements include three large dams, and Mt.St. Helens, an active volcano. Landownership as private, public non-Federal, and Federal are denoted with shading. Restoration actions completed between 1998 and 2003 and, therefore, included as part of the modeled current conditions are identified with symbols.

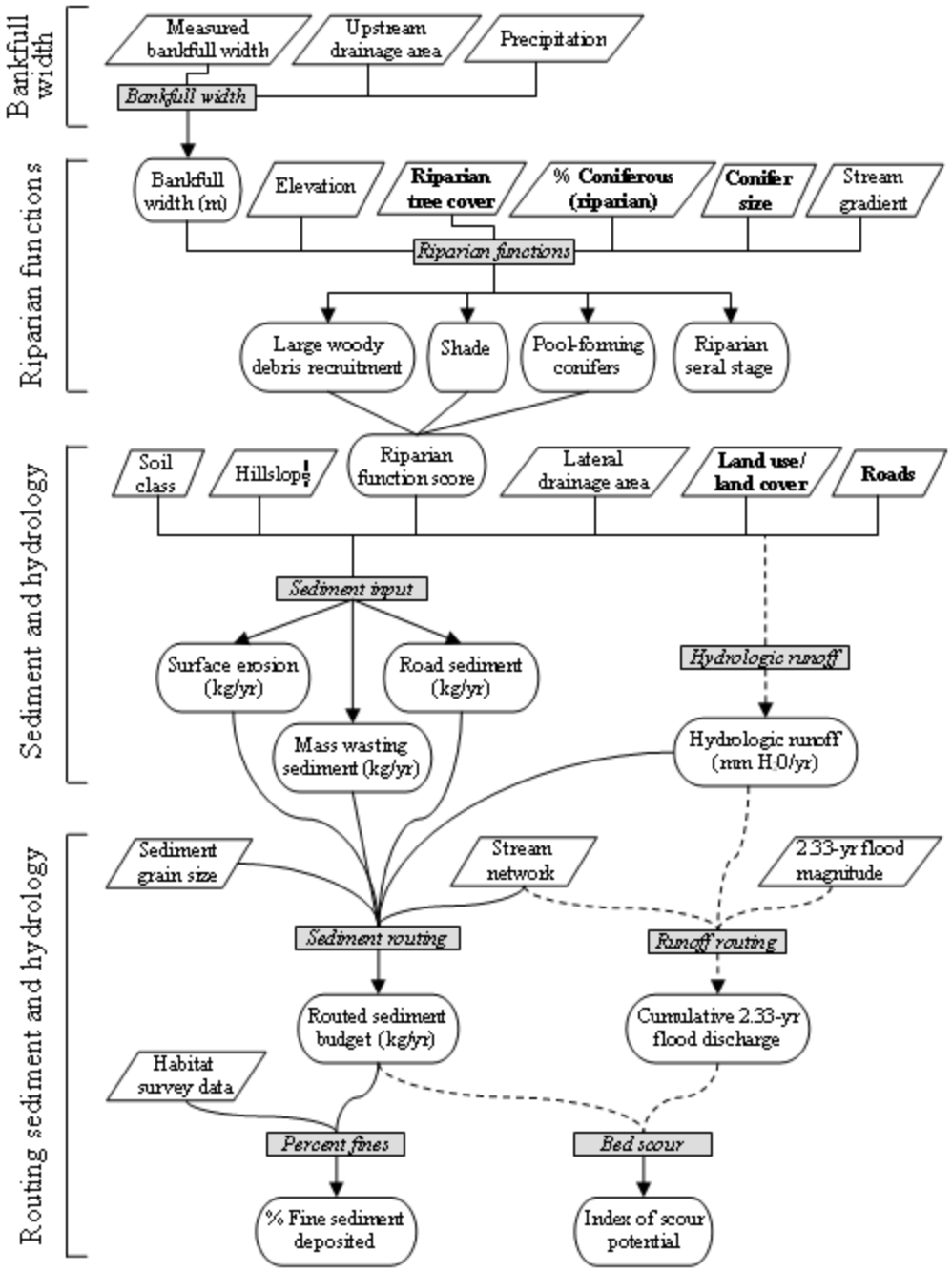

Figure 2. Interaction of data (trapezoids), watershed process models (shaded rectangles), and predicted responses (ovals) in the Decision Support System. Data that can be modified by restoration actions are in bold. All models act on individual stream reaches, and can be summarized at multiple spatial scales (e.g., all reaches, or reaches currently or historically accessible).

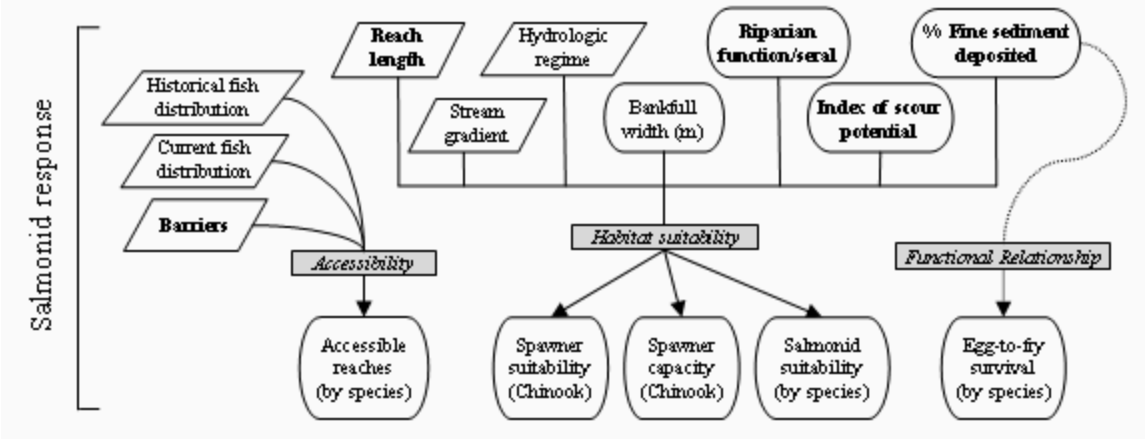

Figure 3. Interaction of data (trapezoids), salmonid response models (shaded rectangles), and predicted responses (ovals) in the Decision Support System. Some inputs to models are predicted by watershed process models (Figure 2). Data that can be modified by restoration actions are in bold. All models act on individual stream reaches, and can be summarized at multiple spatial scales (e.g., all reaches, or reaches currently or historically accessible).

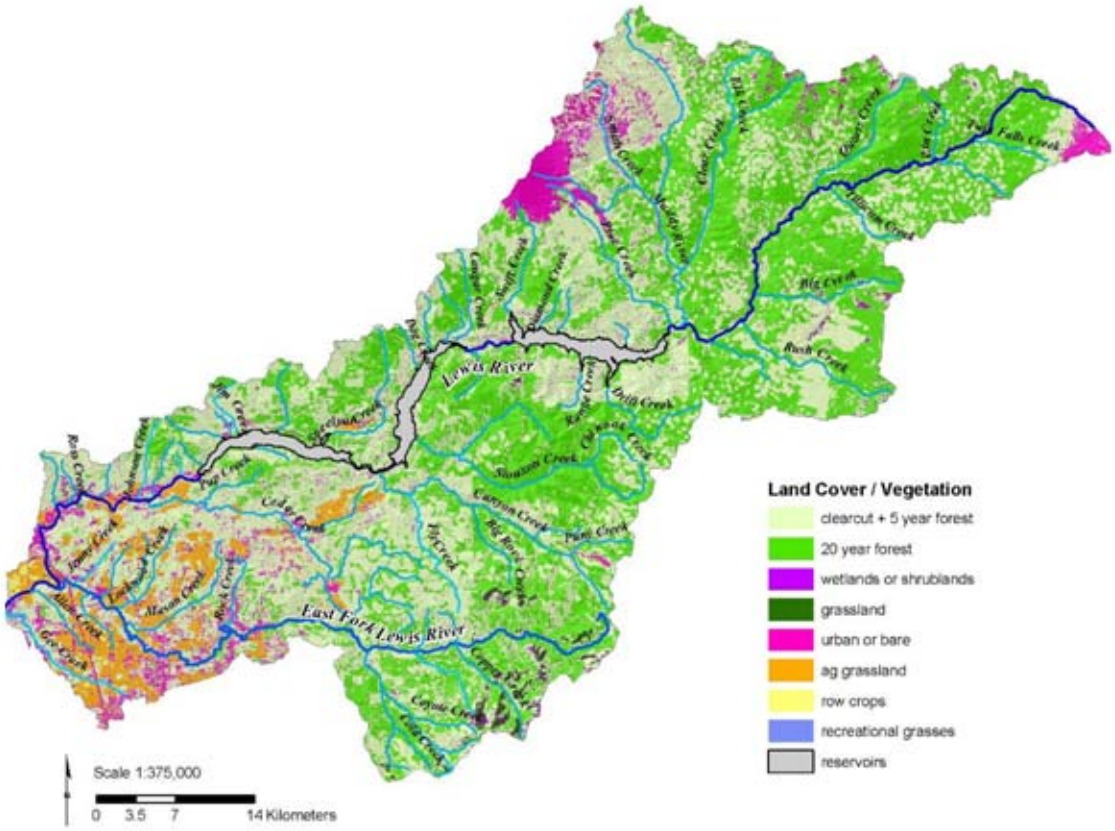

Figure 4. Current land cover and vegetation used in the DSS. Land cover information is from IVMP (BLM 2001). Classes listed here are groups which closely match categories required for the WEPP model, described more fully in Appendix E. Historical upland land cover conditions in the DSS were represented by converting all conditions (with the exception of wetlands, shrublands, grasslands) to 20 year forest.

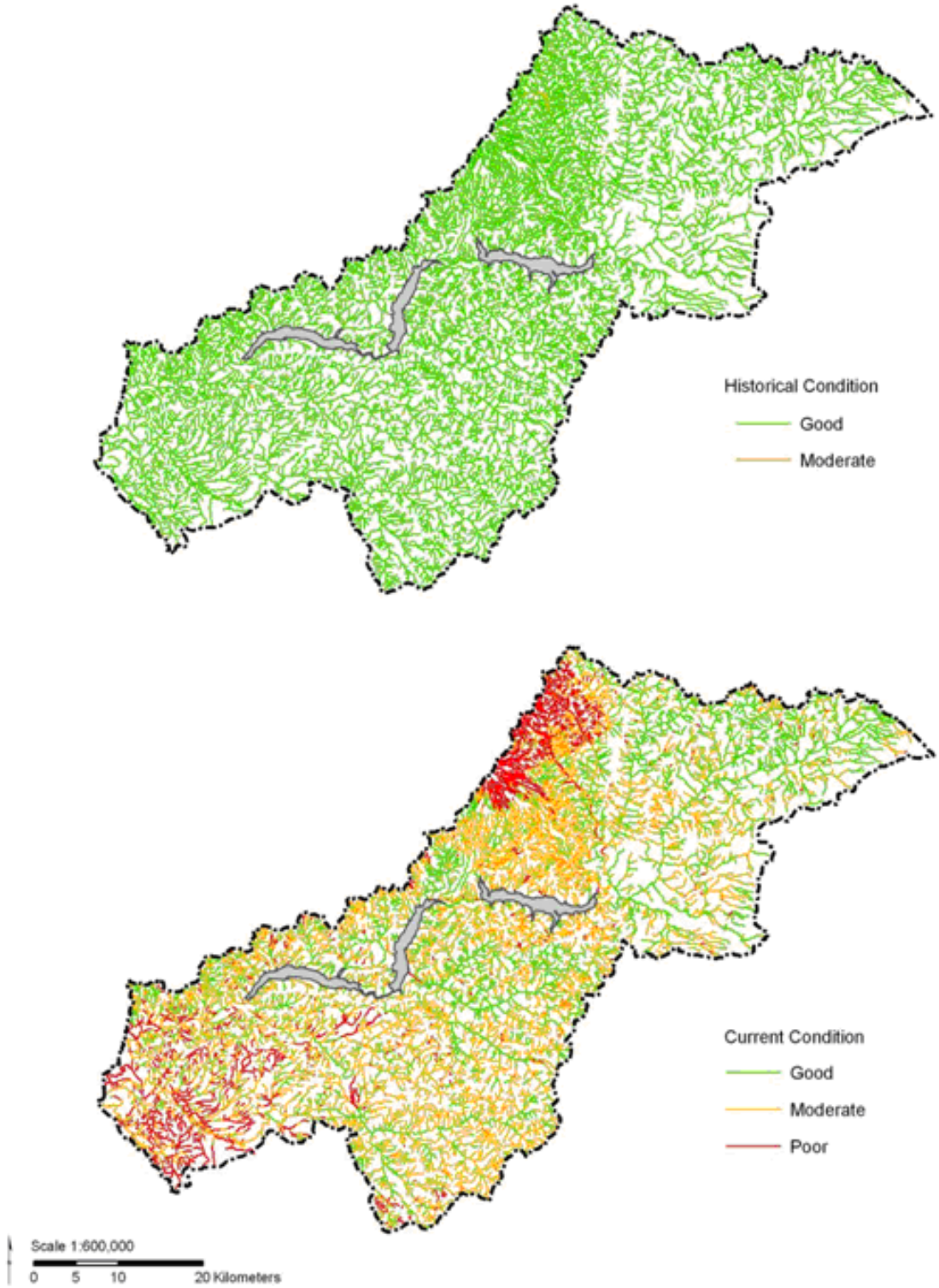

Figure 5. Riparian function rankings for stream segments for base current and historical conditions, as determined by the riparian function model (Appendix H).

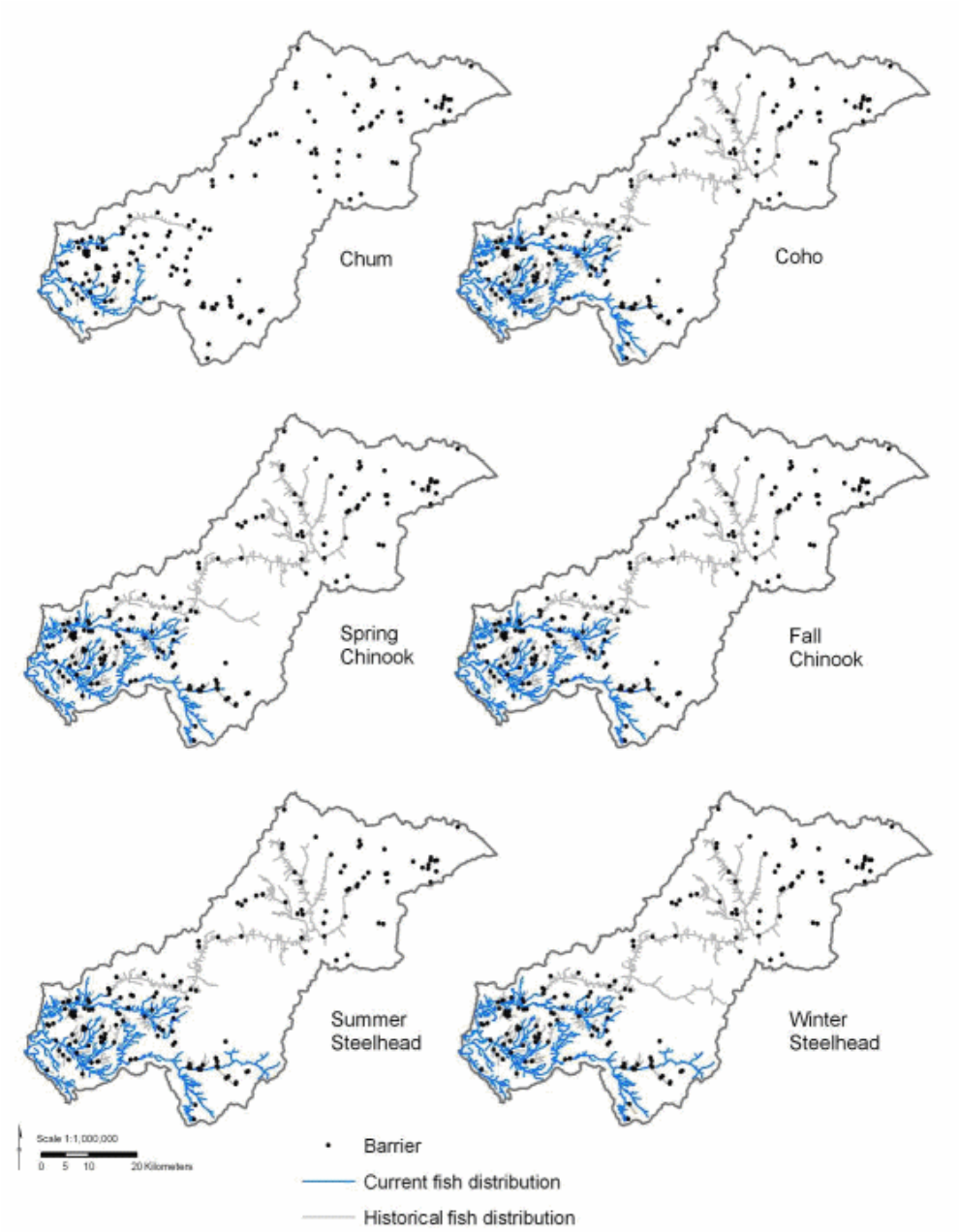

Figure 6. Historical and current fish distribution by species. These distributions, current, potential, and historical (gray), were the base distributions used in the DSS to determine increases in accessibility for barrier removal restoration actions. The barriers database and fish distribution calculations are described in Appendix A.

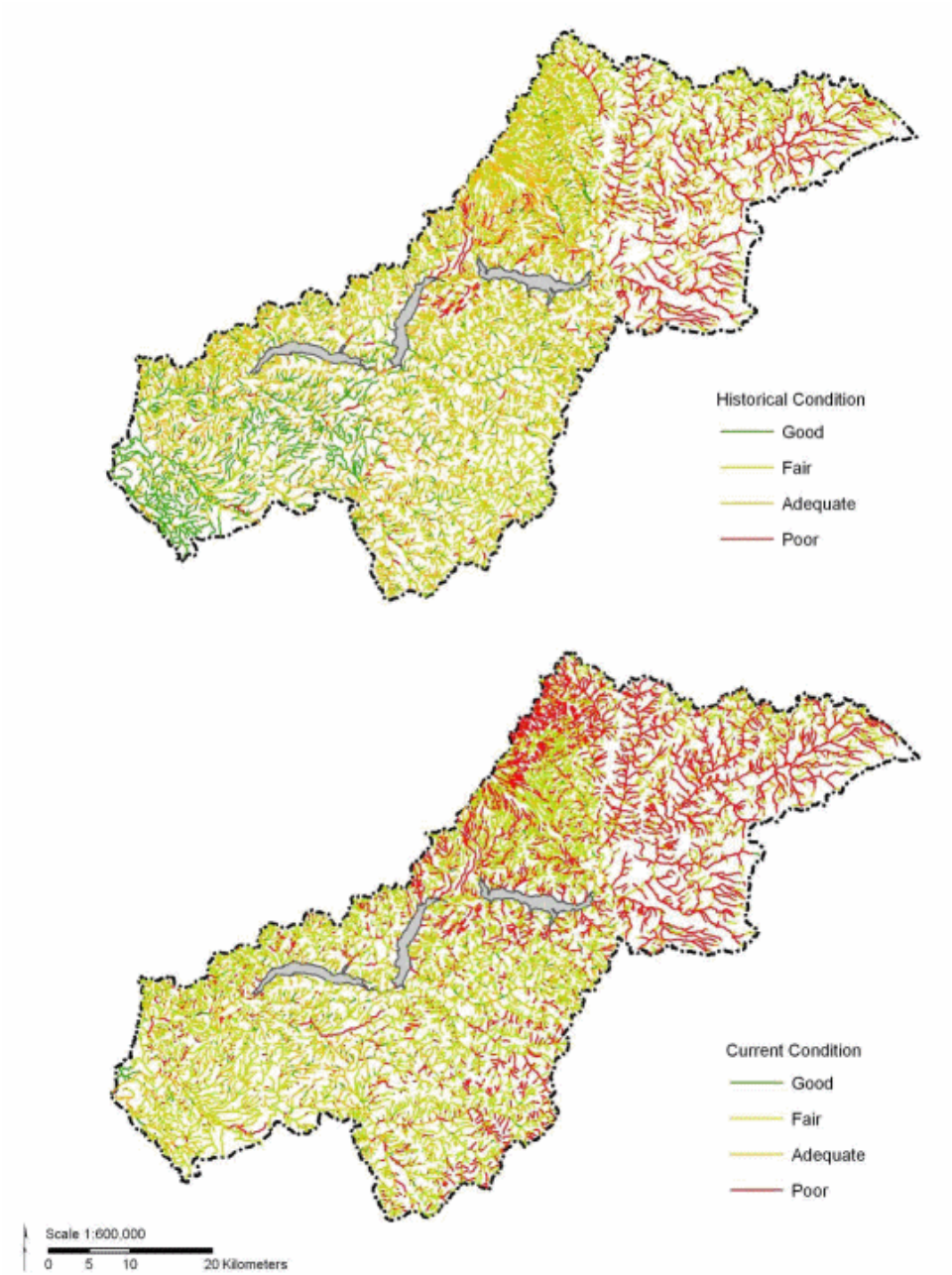

Figure 7. FishEye habitat potential rankings for stream segments for base current and historical conditions, as determined by the FishEye (Appendix J).

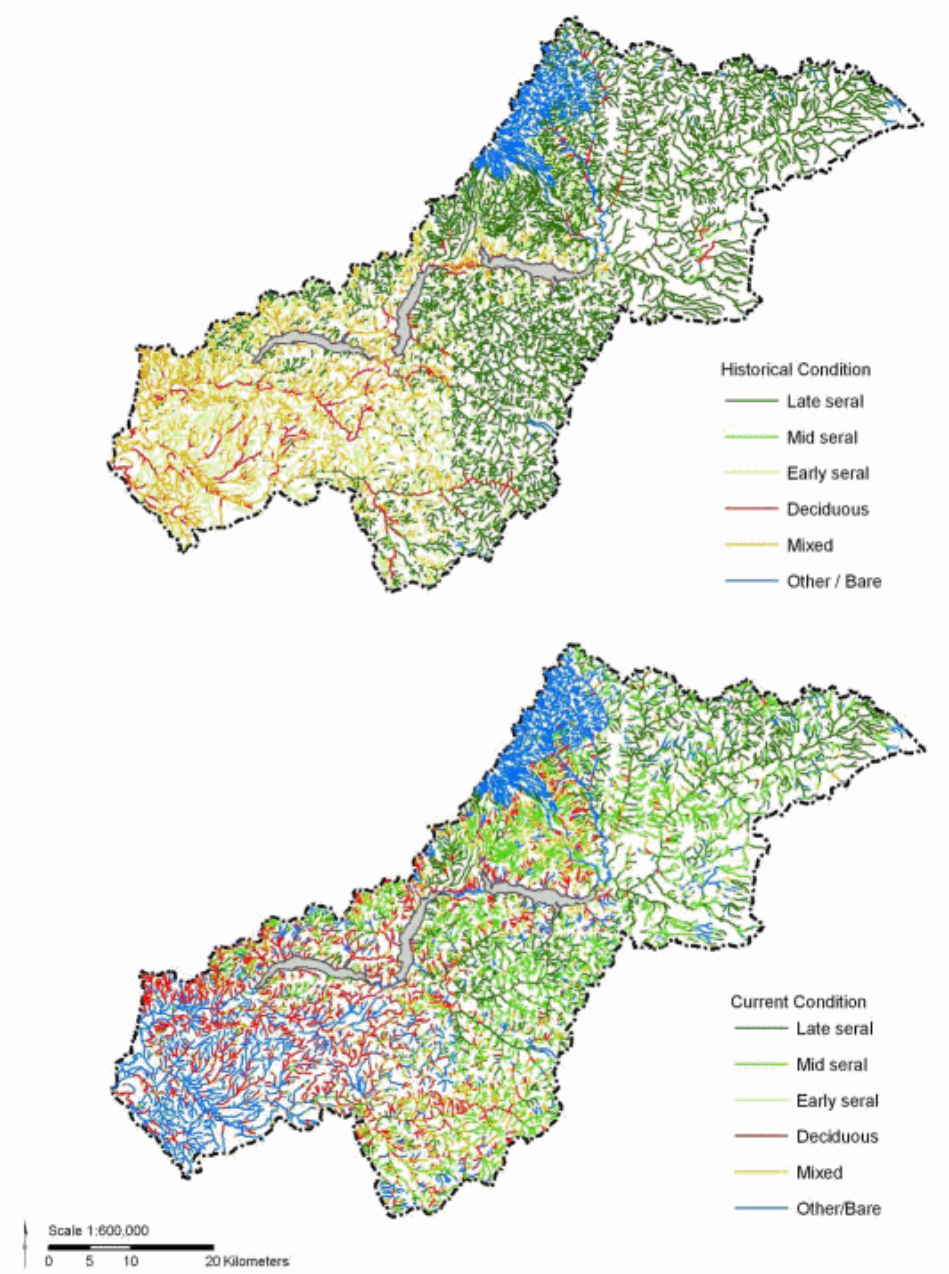

Figure 8. Seral stage of current vegetation in the Lewis River, derived from BLM (2001). Seral stage is used as a primary input for the spawner suitability and potential capacity model (Appendix I).

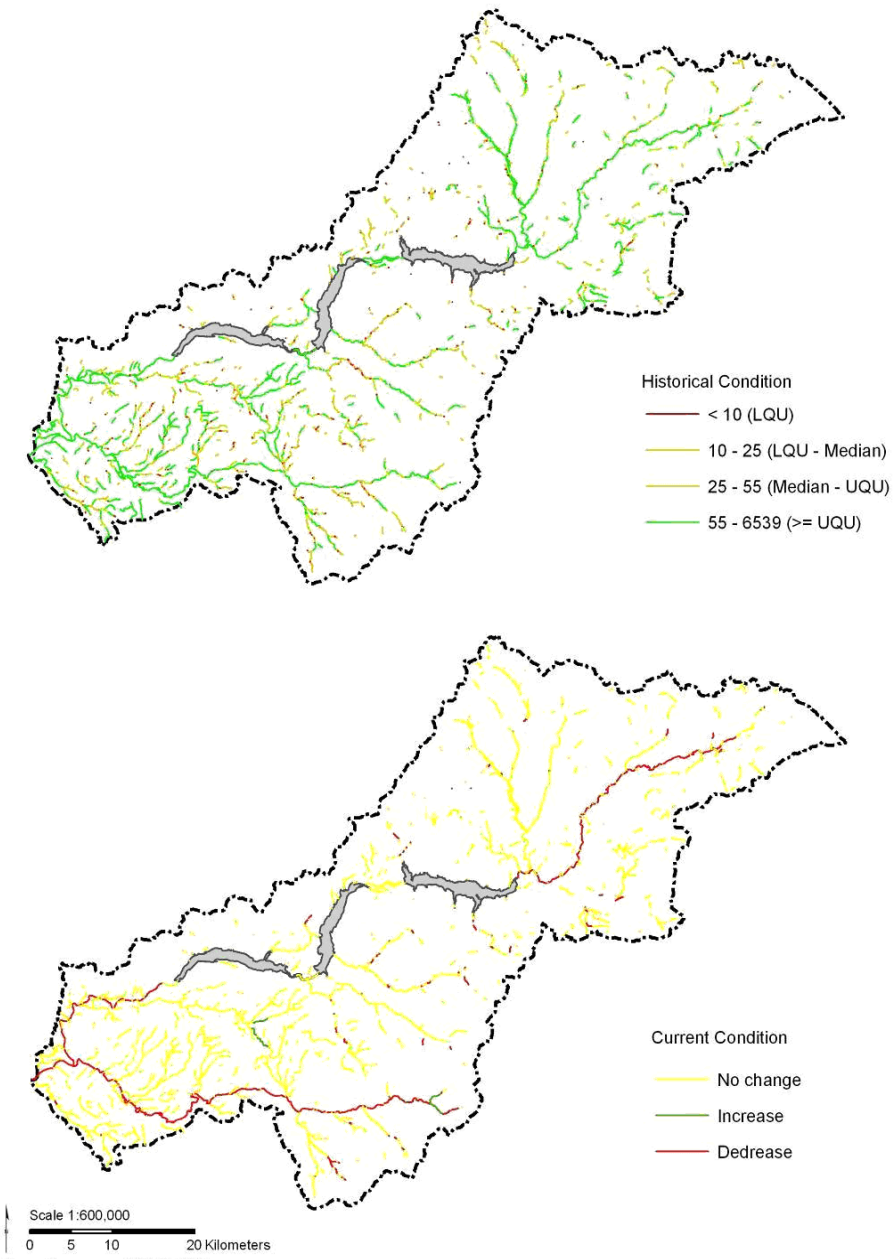

Figure 9. Mean spawning capacity (number of fish), based on the spawner suitability model (Appendix I). Historical conditions (top) were derived by applying the model using historical seral stage estimates (Figure 8).

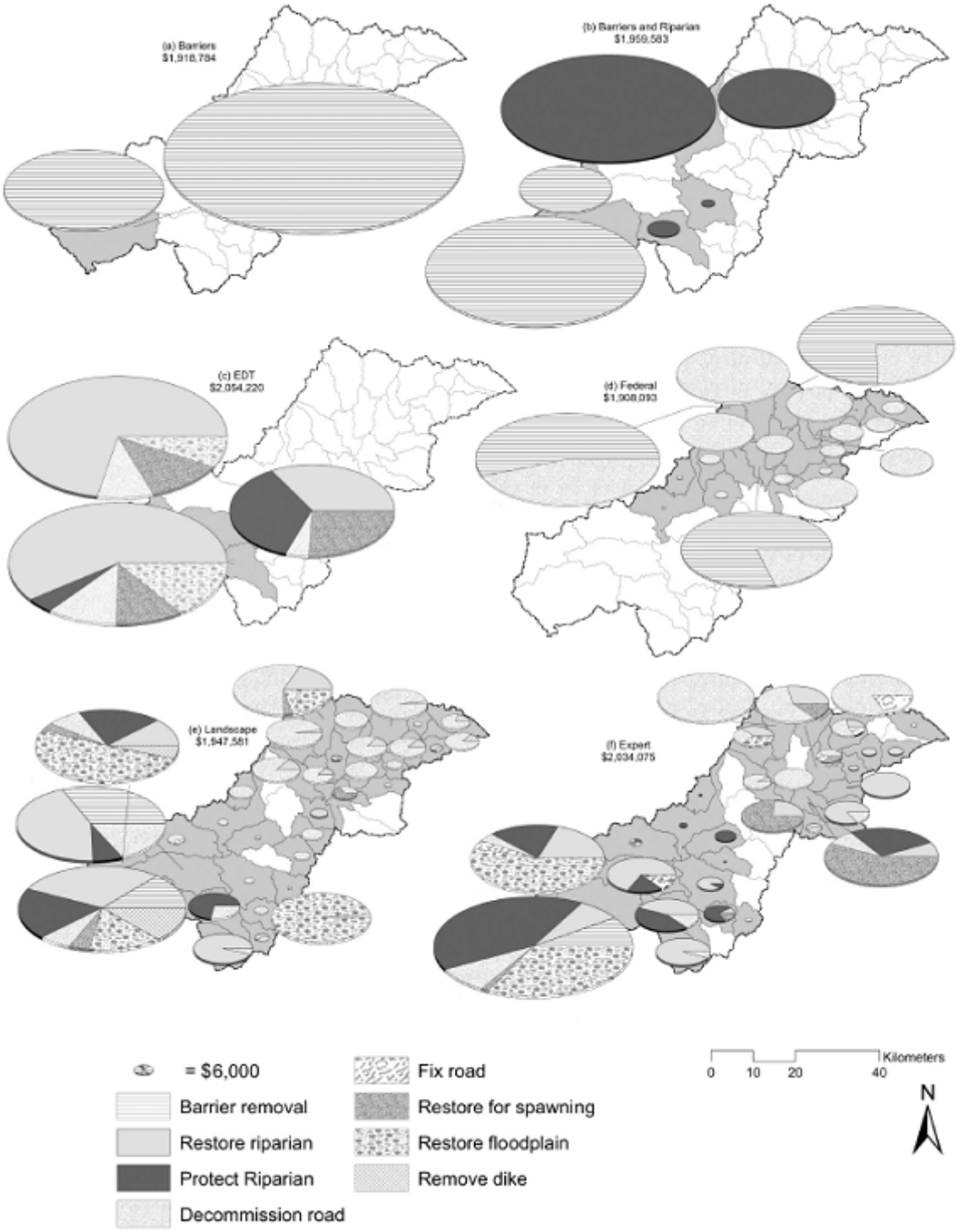

Figure 10. Pie charts (a) through (f) describe the spatial allocation of funds for each of the 6 watershed management strategies. The size of a pie chart represents the total funds allocated per subwatershed. The slices of pie describe how funds were allocated among possible restoration and protection activities. A $6000 pie chart is shown in the legend for scale. Budgets for the landscape and expert strategies were averaged for display purposes.

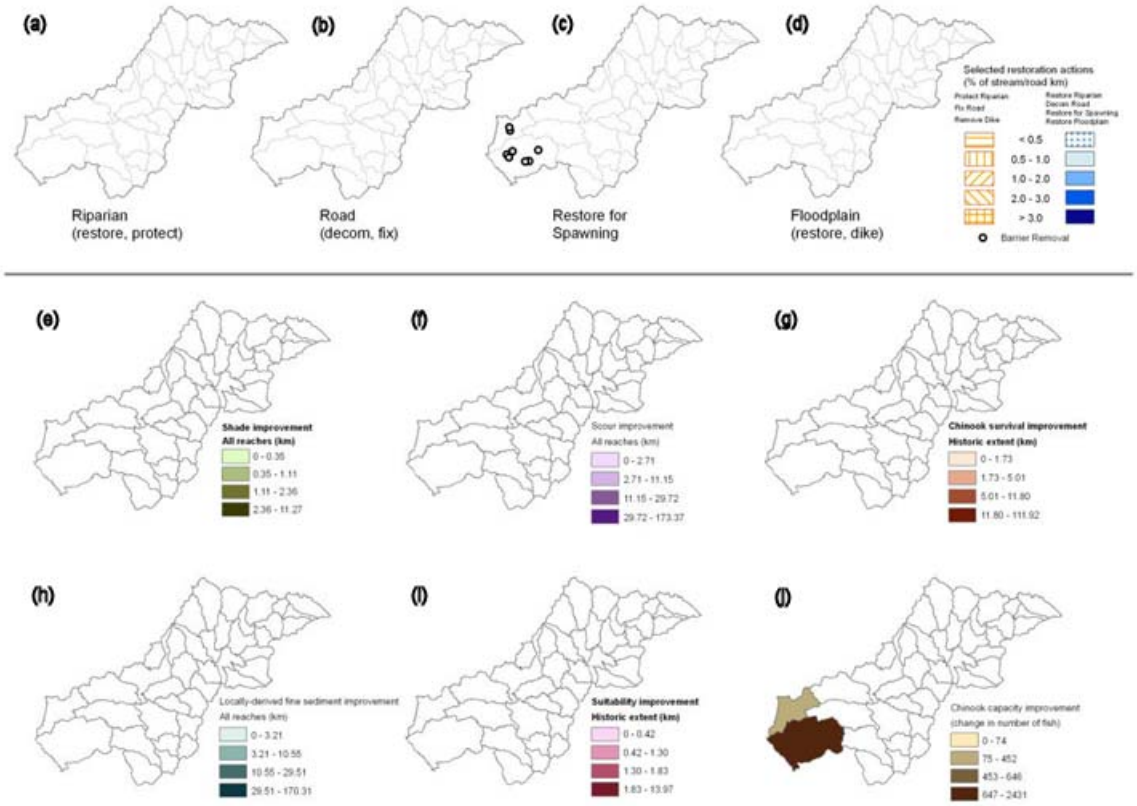

Figure 11. Detailed description of the barriers watershed management strategy. Subwatershed targeted for (a) riparian restoration or protection, (b) road decommissioning or repair, (c) instream restoration to improve spawning habitat, barrier removal, or (d) floodplain restoration are identified in the top row of maps. Panels (e) through (j) describe the impact of those actions in terms of (e) km of stream with increased shade, (f) km of stream with reduced scour, (g) km of stream with improved chinook salmon survival, (h) km of stream with reduced inputs of fine sediment, (i) km of stream with increased chinook salmon suitability, and (j) increased chinook capacity as estimated with the remotely-sensed capacity model (Appendix I). Note that maps are summarized over different spatial extents as denoted in the legend for each map.

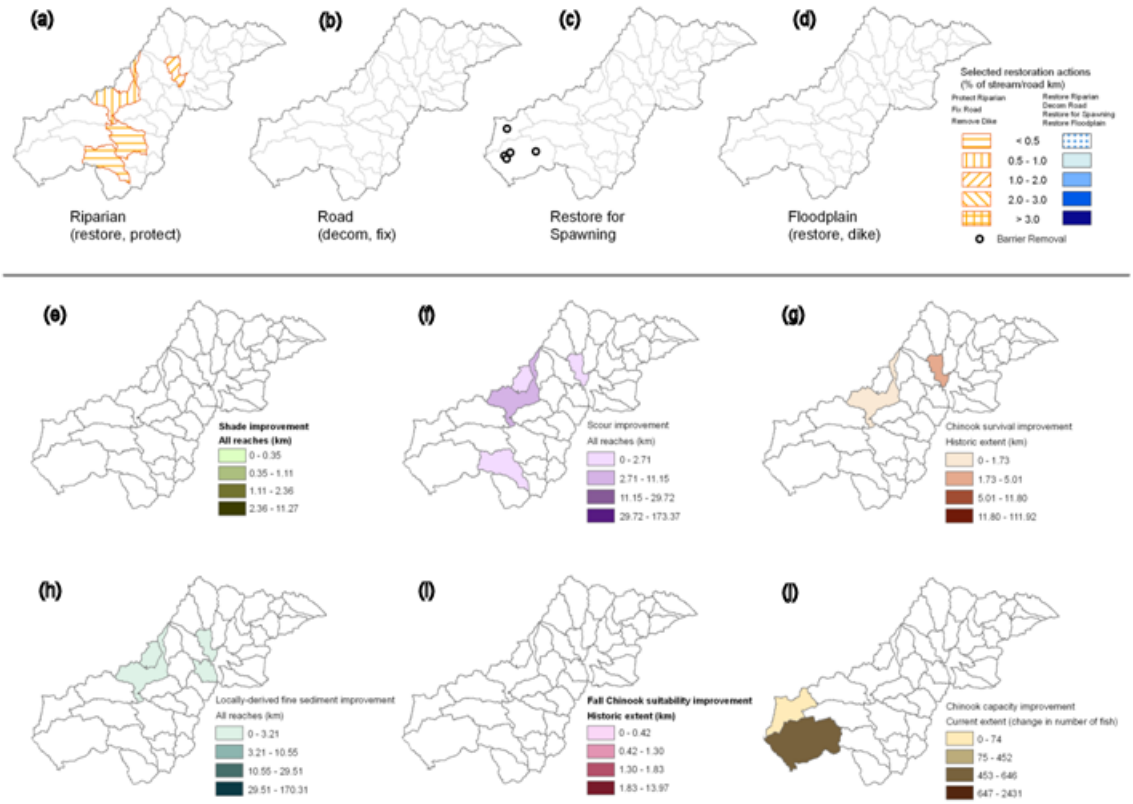

Figure 12. Detailed description of the barriers and riparian watershed management strategy. Subwatershed targeted for (a) riparian restoration or protection, (b) road decommissioning or repair, (c) instream restoration to improve spawning habitat, barrier removal, or (d) floodplain restoration are identified in the top row of maps. Panels (e) through (j) describe the impact of those actions in terms of (e) km of stream with increased shade, (f) km of stream with reduced scour, (g) km of stream with improved chinook salmon survival, (h) km of stream with reduced inputs of fine sediment, (i) km of stream with increased chinook salmon suitability, and (j) increased chinook capacity as estimated with the remotely-sensed capacity model (Appendix I). Note that maps are summarized over different spatial extents as denoted in the legend for each map.

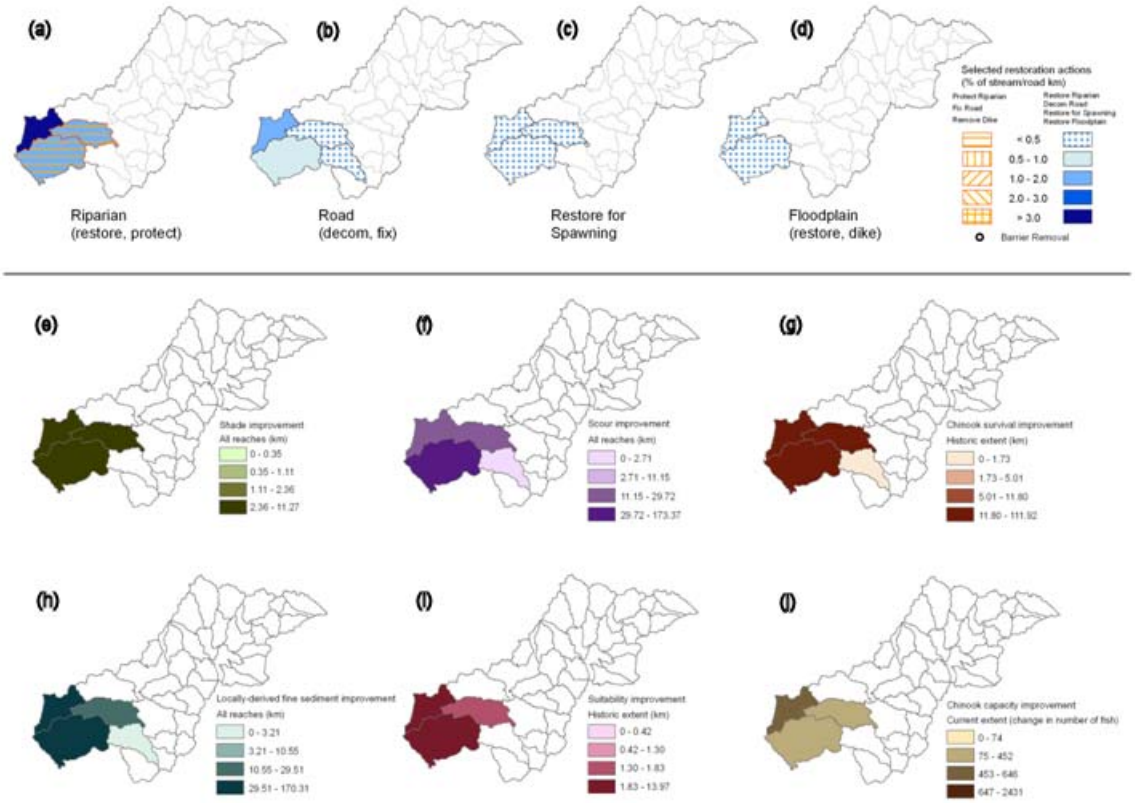

Figure 13. Detailed description of the EDT watershed management strategy. Subwatershed targeted for (a) riparian restoration or protection, (b) road decommissioning or repair, (c) instream restoration to improve spawning habitat, barrier removal, or (d) floodplain restoration are identified in the top row of maps. Panels (e) through (j) describe the impact of those actions in terms of (e) km of stream with increased shade, (f) km of stream with reduced scour, (g) km of stream with improved chinook salmon survival, (h) km of stream with reduced inputs of fine sediment, (i) km of stream with increased chinook salmon suitability, and (j) increased chinook capacity as estimated with the remotely-sensed capacity model (Appendix I). Note that maps are summarized over different spatial extents as denoted in the legend for each map.

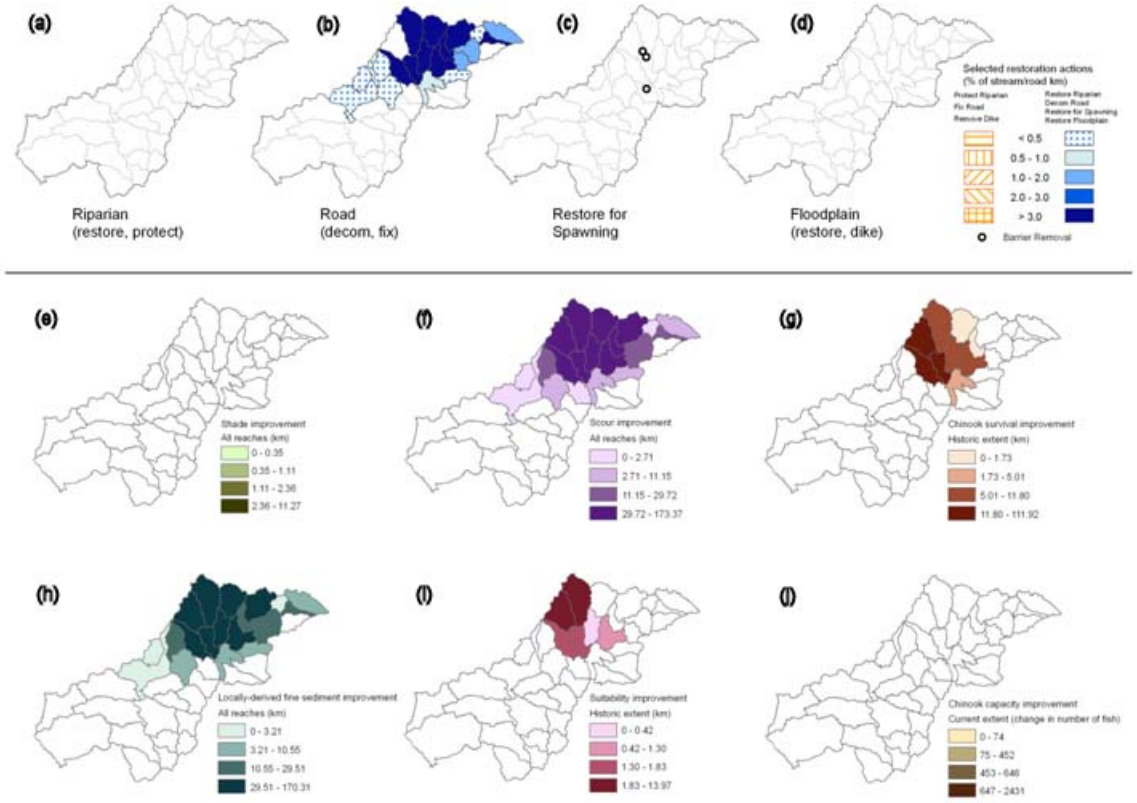

Figure 14. Detailed description of the federal watershed management strategy. Subwatershed targeted for (a) riparian restoration or protection, (b) road decommissioning or repair, (c) instream restoration to improve spawning habitat, barrier removal, or (d) floodplain restoration are identified in the top row of maps. Panels (e) through (j) describe the impact of those actions in terms of (e) km of stream with increased shade, (f) km of stream with reduced scour, (g) km of stream with improved chinook salmon survival, (h) km of stream with reduced inputs of fine sediment, (i) km of stream with increased chinook salmon suitability, and (j) increased chinook capacity as estimated with the remotely-sensed capacity model (Appendix I). Note that maps are summarized over different spatial extents as denoted in the legend for each map.

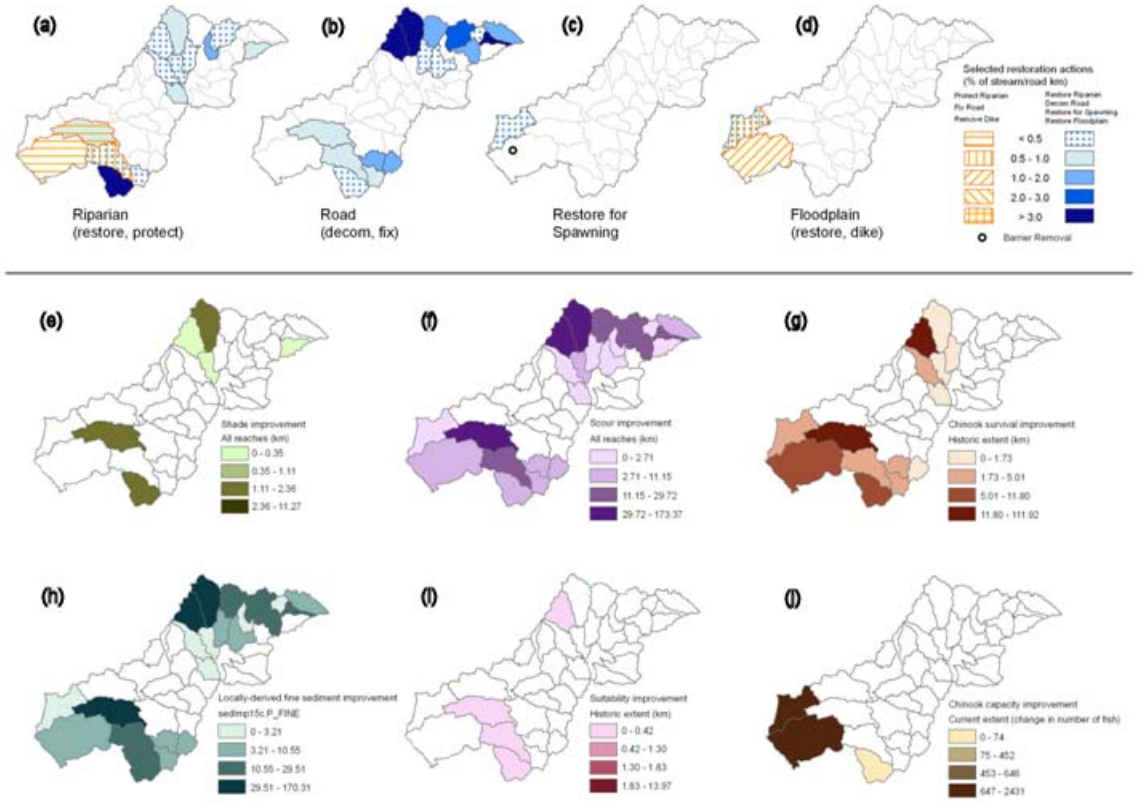

Figure 15. Detailed description of the landscape (1) watershed management strategy. Subwatershed targeted for (a) riparian restoration or protection, (b) road decommissioning or repair, (c) instream restoration to improve spawning habitat, barrier removal, or (d) floodplain restoration are identified in the top row of maps. Panels (e) through (j) describe the impact of those actions in terms of (e) km of stream with increased shade, (f) km of stream with reduced scour, (g) km of stream with improved chinook salmon survival, (h) km of stream with reduced inputs of fine sediment, (i) km of stream with increased chinook salmon suitability, and (j) increased chinook capacity as estimated with the remotely-sensed capacity model (Appendix I). Note that maps are summarized over different spatial extents as denoted in the legend for each map. For comparisons across watershed management strategies, the results of all 5 landscape strategies were averaged.

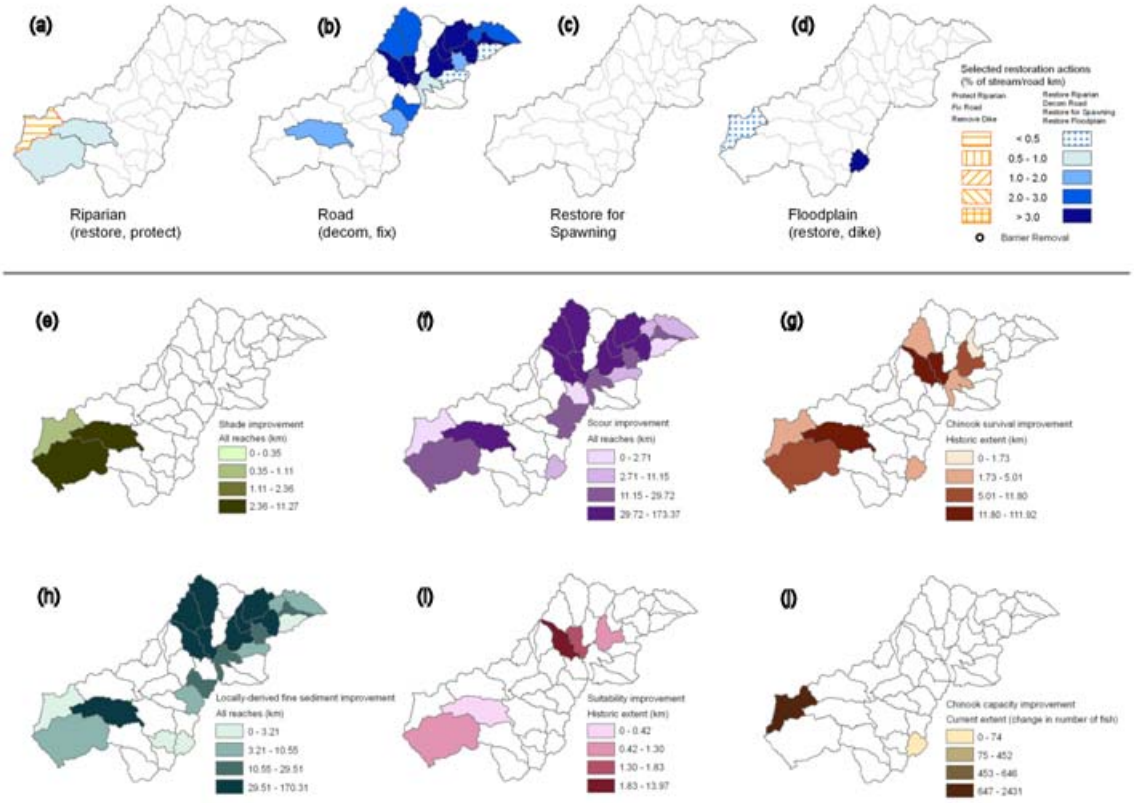

Figure 16. Detailed description of the landscape (2) watershed management strategy. Subwatershed targeted for (a) riparian restoration or protection, (b) road decommissioning or repair, (c) instream restoration to improve spawning habitat, barrier removal, or (d) floodplain restoration are identified in the top row of maps. Panels (e) through (j) describe the impact of those actions in terms of (e) km of stream with increased shade, (f) km of stream with reduced scour, (g) km of stream with improved chinook salmon survival, (h) km of stream with reduced inputs of fine sediment, (i) km of stream with increased chinook salmon suitability, and (j) increased chinook capacity as estimated with the remotely-sensed capacity model (Appendix I). Note that maps are summarized over different spatial extents as denoted in the legend for each map. For comparisons across watershed management strategies, the results of all 5 landscape strategies were averaged.

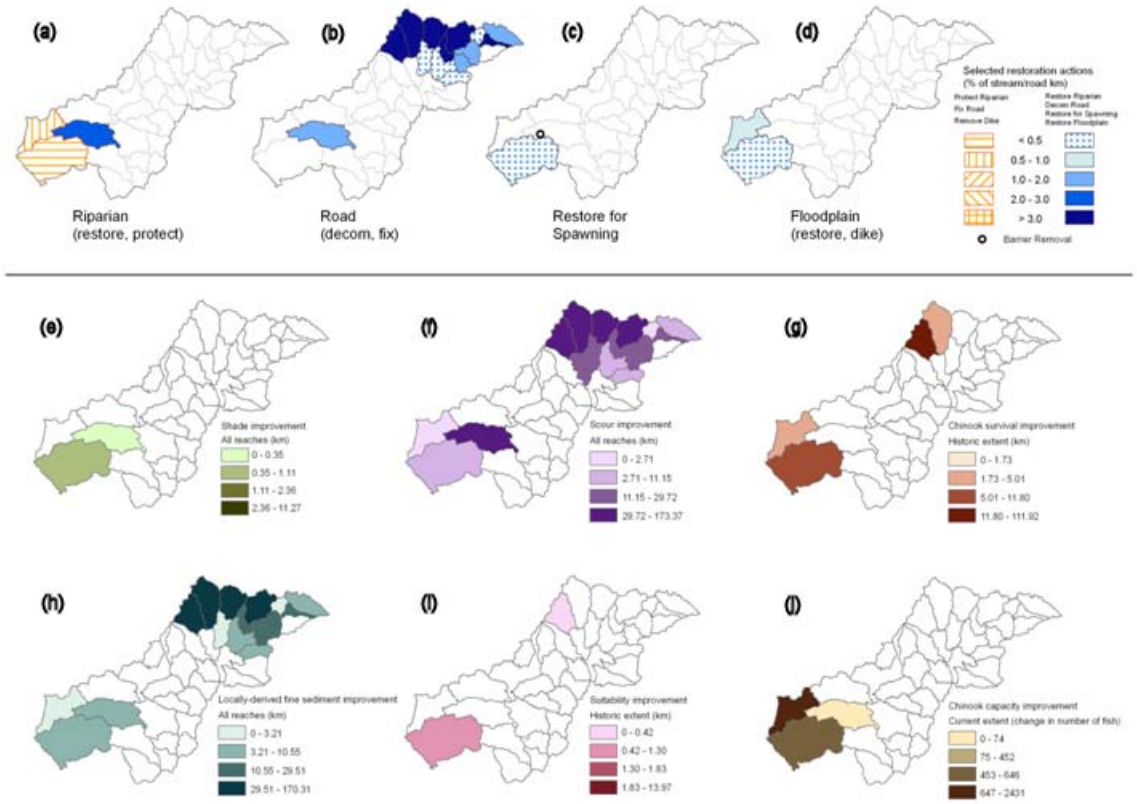

Figure 17. Detailed description of the landscape (3) watershed management strategy. Subwatershed targeted for (a) riparian restoration or protection, (b) road decommissioning or repair, (c) instream restoration to improve spawning habitat, barrier removal, or (d) floodplain restoration are identified in the top row of maps. Panels (e) through (j) describe the impact of those actions in terms of (e) km of stream with increased shade, (f) km of stream with reduced scour, (g) km of stream with improved chinook salmon survival, (h) km of stream with reduced inputs of fine sediment, (i) km of stream with increased chinook salmon suitability, and (j) increased chinook capacity as estimated with the remotely-sensed capacity model (Appendix I). Note that maps are summarized over different spatial extents as denoted in the legend for each map. For comparisons across watershed management strategies, the results of all 5 landscape strategies were averaged.

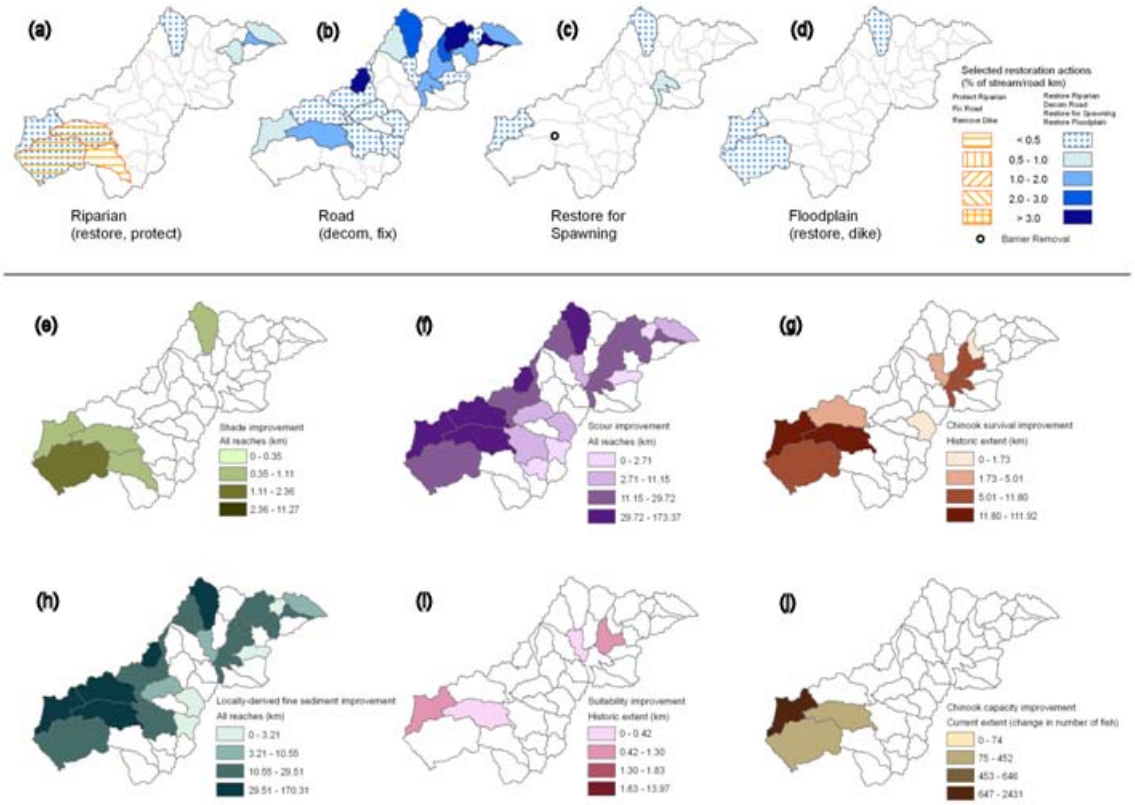

Figure 18. Detailed description of the landscape (4) watershed management strategy. Subwatershed targeted for (a) riparian restoration or protection, (b) road decommissioning or repair, (c) instream restoration to improve spawning habitat, barrier removal, or (d) floodplain restoration are identified in the top row of maps. Panels (e) through (j) describe the impact of those actions in terms of (e) km of stream with increased shade, (f) km of stream with reduced scour, (g) km of stream with improved chinook salmon survival, (h) km of stream with reduced inputs of fine sediment, (i) km of stream with increased chinook salmon suitability, and (j) increased chinook capacity as estimated with the remotely-sensed capacity model (Appendix I). Note that maps are summarized over different spatial extents as denoted in the legend for each map. For comparisons across watershed management strategies, the results of all 5 landscape strategies were averaged.

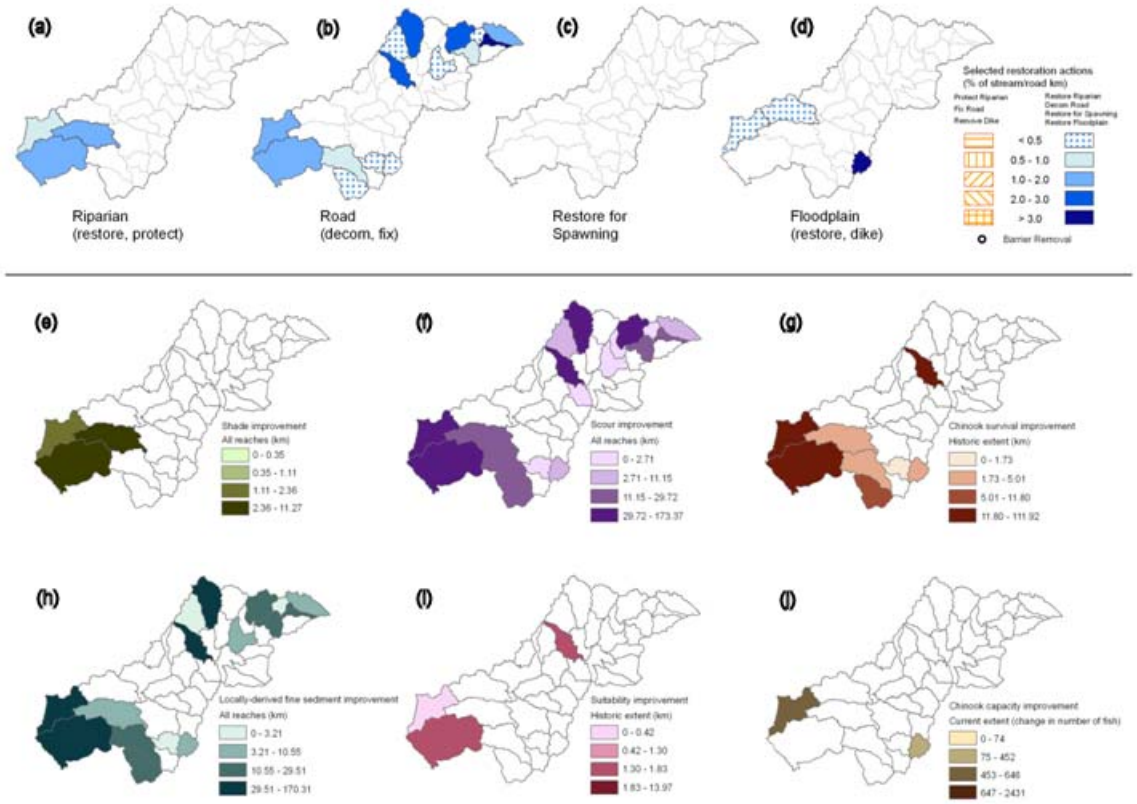

Figure 19. Detailed description of the landscape (5) watershed management strategy. Subwatershed targeted for (a) riparian restoration or protection, (b) road decommissioning or repair, (c) instream restoration to improve spawning habitat, barrier removal, or (d) floodplain restoration are identified in the top row of maps. Panels (e) through (j) describe the impact of those actions in terms of (e) km of stream with increased shade, (f) km of stream with reduced scour, (g) km of stream with improved chinook salmon survival, (h) km of stream with reduced inputs of fine sediment, (i) km of stream with increased chinook salmon suitability, and (j) increased chinook capacity as estimated with the remotely-sensed capacity model (Appendix I). Note that maps are summarized over different spatial extents as denoted in the legend for each map. For comparisons across watershed management strategies, the results of all 5 landscape strategies were averaged.

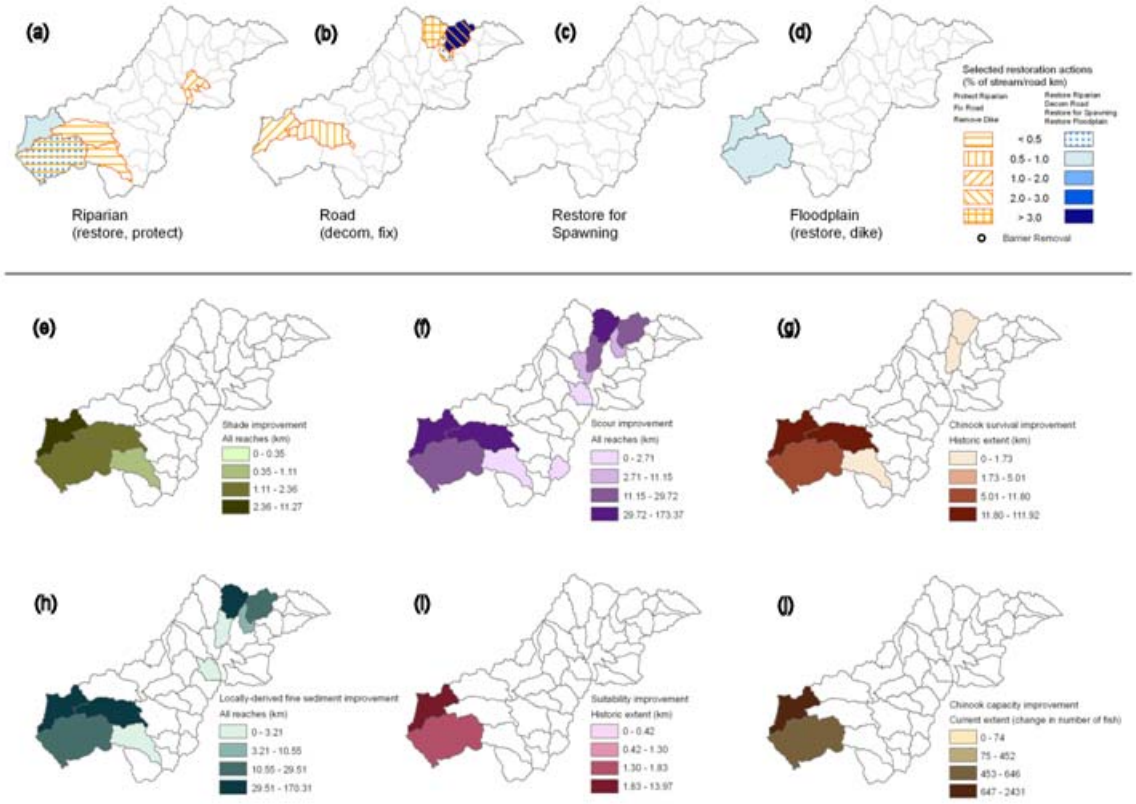

Figure 20. Detailed description of the expert (1) watershed management strategy. Subwatershed targeted for (a) riparian restoration or protection, (b) road decommissioning or repair, (c) instream restoration to improve spawning habitat, barrier removal, or (d) floodplain restoration are identified in the top row of maps. Panels (e) through (j) describe the impact of those actions in terms of (e) km of stream with increased shade, (f) km of stream with reduced scour, (g) km of stream with improved chinook salmon survival, (h) km of stream with reduced inputs of fine sediment, (i) km of stream with increased chinook salmon suitability, and (j) increased chinook capacity as estimated with the remotely-sensed capacity model (Appendix I). Note that maps are summarized over different spatial extents as denoted in the legend for each map. For comparisons across watershed management strategies, the results of all 4 expert strategies were averaged.

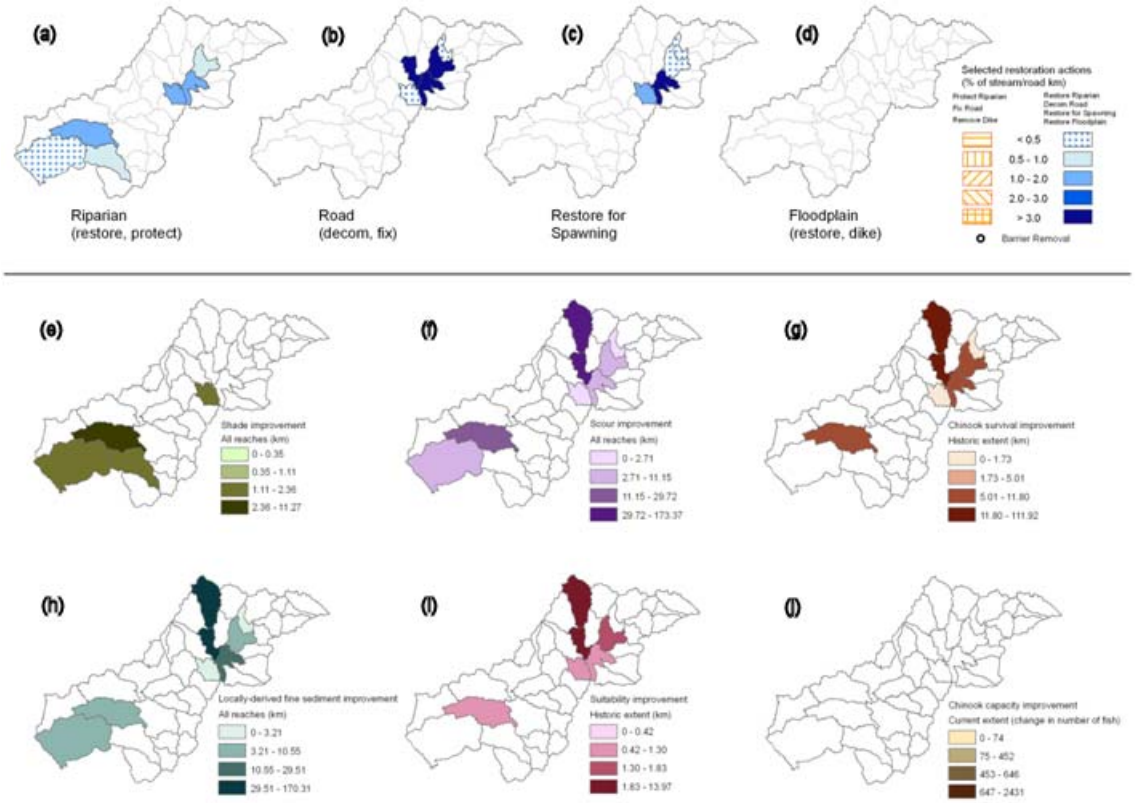

Figure 21. Detailed description of the expert (2) watershed management strategy. Subwatershed targeted for (a) riparian restoration or protection, (b) road decommissioning or repair, (c) instream restoration to improve spawning habitat, barrier removal, or (d) floodplain restoration are identified in the top row of maps. Panels (e) through (j) describe the impact of those actions in terms of (e) km of stream with increased shade, (f) km of stream with reduced scour, (g) km of stream with improved chinook salmon survival, (h) km of stream with reduced inputs of fine sediment, (i) km of stream with increased chinook salmon suitability, and (j) increased chinook capacity as estimated with the remotely-sensed capacity model (Appendix I). Note that maps are summarized over different spatial extents as denoted in the legend for each map. For comparisons across watershed management strategies, the results of all 4 expert strategies were averaged.

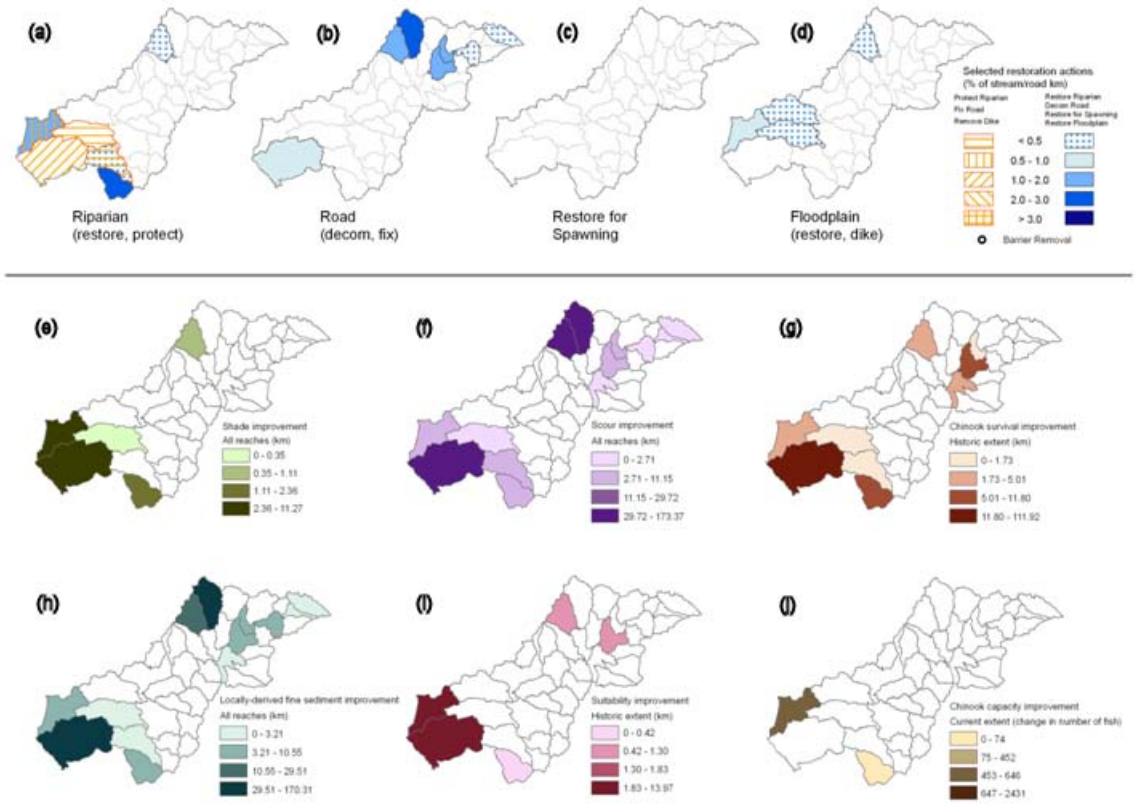

Figure 22. Detailed description of the expert (3) watershed management strategy. Subwatershed targeted for (a) riparian restoration or protection, (b) road decommissioning or repair, (c) instream restoration to improve spawning habitat, barrier removal, or (d) floodplain restoration are identified in the top row of maps. Panels (e) through (j) describe the impact of those actions in terms of (e) km of stream with increased shade, (f) km of stream with reduced scour, (g) km of stream with improved chinook salmon survival, (h) km of stream with reduced inputs of fine sediment, (i) km of stream with increased chinook salmon suitability, and (j) increased chinook capacity as estimated with the remotely-sensed capacity model (Appendix I). Note that maps are summarized over different spatial extents as denoted in the legend for each map. For comparisons across watershed management strategies, the results of all 4 expert strategies were averaged.

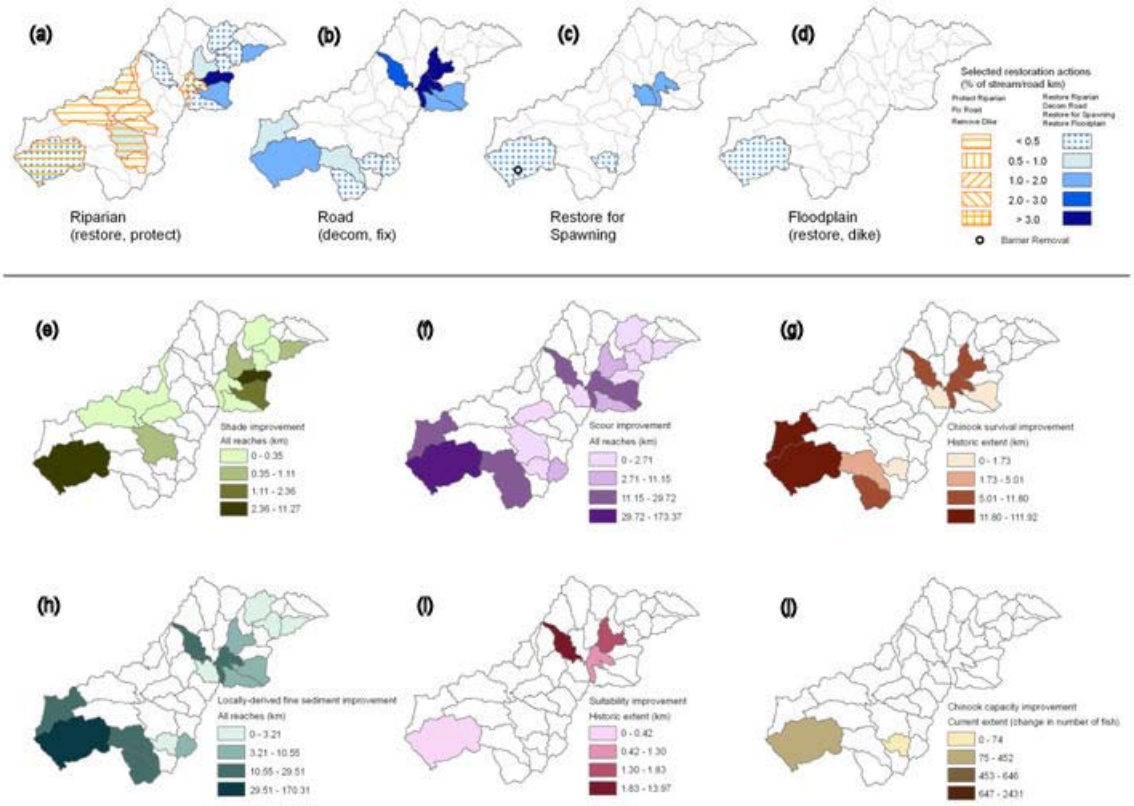

Figure 23. Detailed description of the expert (4) watershed management strategy. Subwatershed targeted for (a) riparian restoration or protection, (b) road decommissioning or repair, (c) instream restoration to improve spawning habitat, barrier removal, or (d) floodplain restoration are identified in the top row of maps. Panels (e) through (j) describe the impact of those actions in terms of (e) km of stream with increased shade, (f) km of stream with reduced scour, (g) km of stream with improved chinook salmon survival, (h) km of stream with reduced inputs of fine sediment, (i) km of stream with increased chinook salmon suitability, and (j) increased chinook capacity as estimated with the remotely-sensed capacity model (Appendix I). Note that maps are summarized over different spatial extents as denoted in the legend for each map. For comparisons across watershed management strategies, the results of all 4 expert strategies were averaged.

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Lewis River Case Study Final Report

A decision-support tool for assessing watershed-scale habitat recovery strategies for ESA-listed salmonids

Appendix A: DSS Base Data

May 2007

Introduction

In this appendix, we describe methods and sources used to generate descriptions of current conditions in the Lewis River Decision Support System (DSS). We describe how we characterized the landscape and how each data source was used in the analysis.

Stream Network Features

Our first step was to delineate basic stream network features: a routed stream network; stream segment lengths; gradients; and lateral drainage areas. We used NetStream (Miller 2003) to generate a stream network based on a 10m DEM and to segment the stream network. We used spatial data from Salmon and Steelhead Habitat Inventory and Assessment Program (SSHIAP) (WDFW 2004) to determine stream segment lengths and tributary junctions for a routed spatial network. Using NetStream, we also generated indicators of stream connectivity (upstream and downstream segments identified) for each stream segment. Stream channel gradients were estimated by fitting a polynomial to stream reach elevations and slopes and estimating slopes for each channel reach pixel; gradient was calculated by averaging over the pixel values for the entire length of each reach (Miller 2003). We attributed each reach with bankfull width estimates modeled from precipitation, stream gradient, and field measurements throughout the Lewis watershed (Appendix G). Next, we used topographic (DEM) and hydrologic features to delineate the lateral drainage areas (drainage wings) for each stream reach (Figure A-1). Drainage wings are the areas that drain to each stream reach. They represent the smallest landscape summary unit and are essential to the sediment routing model (Appendix F) and the surface sediment model (Appendix E).

Vegetation and Land Cover

Land cover information from previously classified satellite imagery was used to represent current vegetation conditions (Lunetta et al. 1997; USGS 1999; BLM 2001; USFS 1995). Three land cover data sets were incorporated into a current template of land use and vegetation. Multiple land cover data sets were incorporated into the base spatial template, because no single data set contained all the necessary modifiers or the correct attribute resolution (Table A-1).

Vegetation type and land cover are used to characterize upslope and riparian areas in the watershed. Current vegetation and land cover are mainly used in the riparian model (Appendix H) and for selecting areas for riparian restoration and preservation. Riparian vegetation condition also modifies sediment and hydrological inputs to stream segments. Land cover types were determined by creating dominant categories that were compatible with the U.S. Forest Service WEPP model (Table E-1, Table E-2). Vegetation categories used for forested areas are 20-year forest, 5-year forest, shrub, grass, and bare ground (Table A-1).

We replaced areas with clearcuts and light fires with 5-year forest because the clearcut age was not evident from the IVMP classified vegetation. The clearcuts are of varying ages (all <10 years old), but the imagery does not readily provide a way to determine the age differences. Clearcut estimates in WEPP are suitable for recent clearcuts, and the

imagery was not recent enough to meet the WEPP clearcut description. The primary source of imagery represents vegetation conditions in 1996. Moreover, when we calibrated our sediment yield estimates with existing sediment yield data for the watershed, the 5-year forest surface erosion estimates more closely matched the field estimates (PWI 1998, PacifiCorp and Cowlitz PUD 2002).

Urban land use comprised less than 1% total area for any 7th field hydrologic unit within the watershed, so it was not originally given special treatment when considering surface erosion. Hydrologic units containing urban pixels in either the USGS or BLM original classified imagery were used to select streams segments with a meta-polygon code of ≥7000. The meta-polygon code for these streams was manually updated to differentiate land cover as urban vs. rock or bare ground.

Table A-1. Classification grouping scheme for vegetation and land use information used in the stratified landscape component of the DSS runoff and sediment yield models. Full references for data sets can be found at the end of this document. LCFRB = Lower Columbia Fish Recovery Board Data – various sources, primarily classified imagery from Lunetta et. al., 1997 (data from 1992 classified Landsat). IVMP = Bureau of Land Management Interagency Vegetation Mapping Project – data from 1996 imagery. USGS = U.S. Geological Survey.

Category/ Source	Vegetation Type	Description
20-year forest, 100% cover , mid-to-late seral stage		
LCFRB	Late Seral Stage	Coniferous crown cover >70%. More than 10% crown cover in trees ≥21 inches diameter at breast height (dbh).
	Mid Seral Stage	Coniferous crown cover >70%. Less than 10% crown cover in trees ≥ 21 inches dbh.
IVMP	Conifer cover ≥70%	
5-year forest, 85% cover, clearcut regrowth, new clearcuts, light burns, and early seral stage		
IVMP	≤70% coniferous vegetation OR ≥ 70% vegetation but < 30% coniferous vegetation	
	Total forest cover = 0-100%	Includes only forested areas
USFS	Recent wildfires	Fires that occurred within the Lewis River watershed between 1990 and 1994
LCFRB	Early Seral Stage	Coniferous crown cover ≥10% and <70%. Less than 75% of total crown cover in hardwoods tree/shrub cover.
	Other lands in forested areas	Areas in forested lands with <10% coniferous crown cover (can contain hardwood tree/shrubs, cleared forest lands, etc.).
Shrubland, 80% cover		
USGS	Shrubland	Areas dominated by shrubs; shrub canopy accounts for 25-100% of the cover
	Woody wetlands	Areas where forest or shrubland vegetation accounts for 25-100% of the cover and soil/substrate periodically saturated with or covered with water
	Herbaceous wetlands	Areas where perennial herbaceous vegetation accounts for 75-100% of cover and soil/substrate is periodically saturated with or covered with water.
Grasslands/prairie, 40% cover		
USGS	Grasslands/Herbaceous	Areas dominated by upland grasses and forbs
Severe fire 45% (15% true) cover Severe category is recommended for recent, high intensity burns		
USFS	Historical severe burns	General boundaries of historical burn areas from USFS fires history map, includes documented fires of 1902,

		1917, and 1924
Urban use, rocky outcrops, quarries, or bare ground		
USGS	Low Intensity Residential	Areas with a mixture of constructed materials and vegetation that most commonly include single-family housing units.
	High Intensity Residential	Areas with heavily built up urban centers such as apartment complexes and row houses.
	Commercial/Industrial Transportation	Areas with infrastructure (e.g. roads, railroads, etc.) and all highways and all developed areas not classified as High Intensity Residential
	Bare Rock/Sand/Clay	Areas of bedrock, desert, pavement, scarps, talus, slides, volcanic material, or glacial debris
	Quarries/Strip Mines/Gravel Pits	Areas of extractive mining activities with significant surface expression.
	Transitional	Areas of sparse vegetative cover, often because of land use activities.
IVMP	Urban	
	Barren	bare soil, rock, lava, sand, etc.
Agricultural use		
USGS	Pasture/hay, small grains, fallow	Areas of grasses, legumes, grass-legume mixtures, wheat, barley, oats or temporarily barren from tilling
	Orchards, vineyards, and row crops	Areas of crops, such as corn, soybeans, vegetables, or orchards/vineyards
	Urban/Recreational Grasses	Urban vegetation in parks, lawns, golf courses, airport grasses, and industrial site grasses.

Stratification of the Landscape

A primary objective of the DSS is to model changes to landscape features and measure their effect. Three landscape stratifications categories were used: soil texture, land cover (vegetation and land use), and topography (slope). These three stratified parameters were the primary descriptors of upslope landscape characteristics for stream reaches and drainage wing watersheds, determining, for example, surface sediment and runoff data used in the DSS. Vegetation, surface runoff and sediment input to streams are changed to model impacts of restoration and protection actions (Appendix D).

ESRI ArcInfo GRID and polygon spatial data structures were used to analyze and stratify the landscape into discrete units or patches. The grain size of the analysis was $30m^2$, as determined by the dataset with the lowest resolution (i.e., largest cell size) of the datasets used in spatial modeling. Each cell represents soil, vegetation, or slope at the $30m^2$ grain size. Vegetation, soil, and slope grids were overlaid to produce one grid, which identifies unique clusters or patches of co-located features with a grid code, which we call meta-polygons (Table A-2; Figure A-1). These cluster codes correspond to the WEPP surface and mass wasting sediment yield (SY), and the WEPP surface storm runoff (SSR), described in Appendix E. Data were converted to a polygon format, which resulted in a dataset with clusters of polygons, with homogeneous land cover and physical characteristics (hence the name “meta-polygons”).

Table A-2. Coding scheme for meta-polygons. Final cluster codes within the DSS database are an additive combination of vegetation code + slope code + soil code.

Vegetation		Slope		Soil	
Type	Code	(%)	Code	Type	Code
20 year forest	2000	0-10	100	silt loam	1
5 year forests	3000	10-20	200	clay loam	2
Shrubland	4000	20-30	300	sand loam	3
Grassland	5000	30-40	400	ash, pumice subsoil	4
Fire	6000	40-50	500		
Urban	7000	50-60	600		
Agriculture - alfalfa	8000				
row crops	9000				
recreational grasses	10000				
Bare rock	11000				

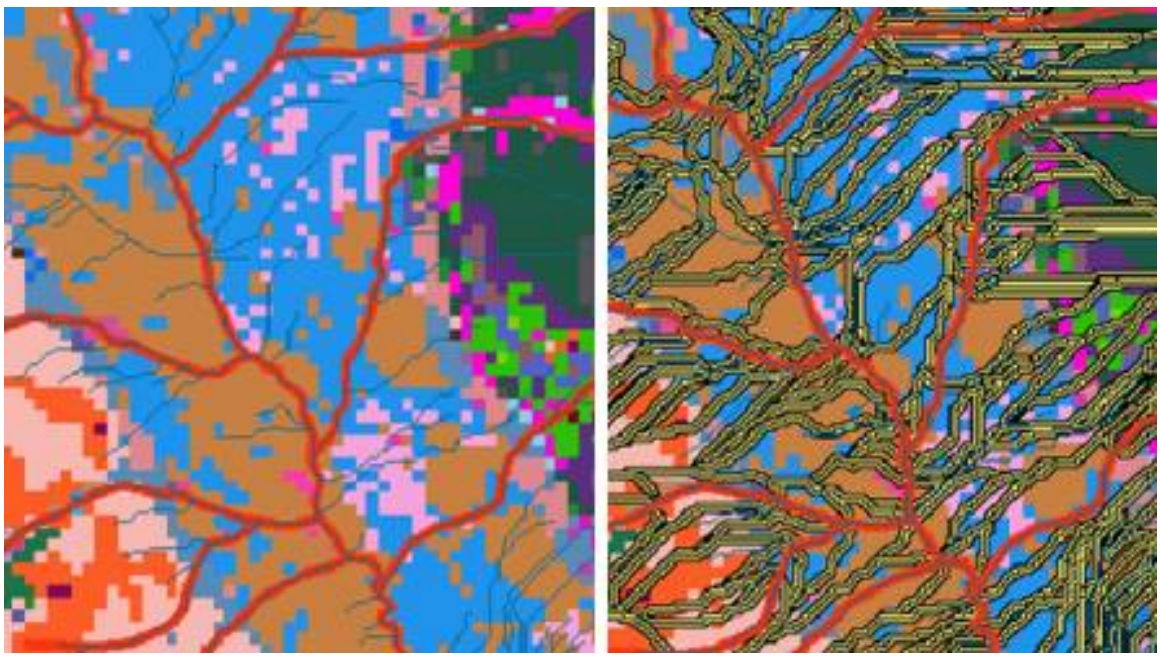

Figure A-1. The image on the left illustrates a section of the meta-polygon data for the Lewis River. Individual colors represent unique pixel cluster codes. Parts of this combined cluster code schema are identified in Table A-1. The image on the right illustrates the "wing code" or lateral drainage wings over which measurements for each reach are summarized. The number of paired drainage wings between tributary junctions is equal to the number of stream reaches.

The base land cover data were then intersected with the drainage wings for each stream segment to split meta-polygons that overlapped between multiple lateral drainages, and to identify the co-located drainage area (Figure A-1). Meta-polygons were further subdivided into riparian and upland zones by intersecting them with a 60m stream buffer per bank. This additional information was necessary for modeling the effects of riparian restoration or protection. Total area for each unique meta-polygon cluster code was then summarized for each lateral drainage zone, and transferred to a tabular database used in the DSS for modeling effects of conservation strategies. The summarized land cover codes, location (riparian or upland), and corresponding areas are stored as an SDE spatial feature dataset, with tabular attribute information and summaries. These data are

viewable within a GIS, but the DSS uses only the summarized tabular information for further processing. Only the total area of each unique class of meta-polygon was calculated for the each lateral drainage area. The grain size of the final analyses was dependent on the size of the drainage wing and the proportion of the homogeneous meta-polygon clusters within this wing.

Soil and Topography

Two main sources of data were used to characterize soils in Lewis subwatersheds. Soil information for the Gifford Pinchot National Forest (USFS 1999) was used for federal lands within the watershed. Soil Survey Geographic database (SSURGO) (NRCS 2000) and Department of Natural Resources 1:24,000 scale soil data (DNR 2000, DNR 2002) were used for the remaining non-federal sections of the watershed. The spatial soil information was generalized into a few classes of surface soil texture, and were edge-matched together to create one soil GIS GRID for the entire watershed, since the attributes for each were similar and had similarly described attributes and source scale. A variety of loam-dominated soil textures were present in the Lewis watershed, and we grouped the existing information so it best matched WEPP parameters (Table A-2).

Hillslope was calculated for each cell in a 10-meter Digital Elevation Model (DEM) of the watershed. GRID data were then resampled to match the minimum mapping unit of $30m^2$ cell size. Hillslope is calculated by the maximum rate of elevation change from each cell to adjacent cells. Percent slope was then grouped into six categories to correspond to WEPP interface parameters. Slope categories are split by 10% increments as depicted in Table A-2, and described in Appendix E. Slope categories were based on the general slope classification from the DEM (0%-10%, 11%-20%, 21%-30%, 31%-40%, 40%-50%, and 50%-80%).

Surface Sediment

The DSS sediment routing model (Appendix F) required information on the distribution of soil sediment size. Sediment size distributions for both surface and road sediment yields were obtained from the SSURGO databases and soil surveys for each county in Lewis basin (Clark, Cowlitz, and Skamania) (NRCS 2000; DNR 2000; DNR 2002).

The DSS variable used for estimating in-stream sediment size is derived from proportional estimates of soil size classes for each drainage wing and associated stream reach. Soil horizon estimates based on Washington DNR and SSURGO data were used to estimate the proportions of various sediment sizes (DNR 2000; NRCS 2000). Data were unavailable for a small section of the upper North Fork Lewis, so U.S. Forest Service soil information was used in this section (USFS 1995).

Each soil series type is differentiated by a code named the Map Unit Key (MUKEY) or MapUnit (USFS). Soil series with the same MUKEY are typically spatially disconnected polygons that may be distributed across the landscape. Specific information on the soil sizes within the soil horizon is available at the spatial scale of the MUKEY. The soil size comprising a soil series is called the *soil distribution*, and is determined by percent proportion of sample soils with grain size larger or smaller than a given sieve size (DNR 2000). Data from six to ten size categories were grouped into six size classes and percent of total was calculated for each sieve class by the MUKEY (Table A-3).

County-level SSURGO data was used as extensively as possible. Large sections of the Gifford Pinchot National Forest (GPNF) (upper North Fork Lewis) are outside the spatial extent of the SSURGO soils database. To obtain soil distribution information in these areas, USFS soil type MAP UNITS were calibrated to SSURGO soil series where spatial overlap occurred. The Skamania County SSURGO database overlapped the GPNF in portions of the Upper Lewis watershed, and the Clark County database overlapped a portion of the Gifford Pinchot in the Middle Lewis watershed and East Fork Lewis watershed. After calibrating the data, soil distribution values were extrapolated to the areas with similar soils in the upper North Fork Lewis.

Once the soil distribution summaries for road and surface sediment were complete for the entire watershed, we summarized soil distribution by lateral drainage area for each reach. Soil series polygons were intersected with the lateral drainage area polygons (drainage wings), so that each drainage wing contained multiple patches of each soil series and associated soil distributions. An output summary table was created for each of the six soil size categories. Each table contained the distribution of percent values that occur in each lateral drainage zone, weighted by area. The relative contribution of the particular soil size to the total area of the drainage wing zone was calculated by:

$$S = \sum \left(\frac{A_{mprc}}{A_z} \right) P$$

where

S = proportional sum for size class

A_{mprc} = area (m^2) of MUKEY fraction percent value within drainage zone

A_z = total area (m^2) of drainage zone

P = MUKEY fraction percent value

The resulting value *S* is an area-weighted soil size estimate for the zone.

The SSURGO database does not provide information to estimate sediment sizes from mass wasting. The sediment size distribution was estimated from mass wasting assessments from the Tilton and East Fork Lewis watershed (Appendix F, Miller, 2004 pers. comm.). The mass wasting size distributions were converted to volume per unit area and then combined with *S* to calculate total soil size estimate per unit area. The results were tested for consistency, and *S* for the size classes summed to 1 in most cases. The drainage zone and associated adjusted soil size fraction were imported into the DSS database, for use as a base input in the sediment routing procedure.

Table A-3. Sample table showing sediment size distribution classes by map unit key. Sediment size distributions (mm) for surface and road sediment yields were obtained from the SSURGO databases and soil surveys for each county in Lewis basin (NCRS GIS database). In the database, each soil series (MUKEY in the database) has a distribution based on percent of size greater or less than a given sieve size. Sediment sizes were distributed into 6 size classes that were then incorporated into the DSS Access database.

MUKEY	> 78mm	>4.8-78mm	1.0-4.8mm	<1.0-0.5mm	<0.5-0.25mm	<0.25mm
	Cobble	Coarse gravel	Very Coarse sand to gravel	Coarse sand	Med sand	Fine sand and less
	GT78_PRC	GT4.8_LT78	LT4.8_GT1	LT1_GT.5	LT.5_GT.25	LT.25_PRC
71952	0.0	28.0	2.5	3.6	5.0	60.9
71953	0.0	28.0	2.5	3.6	5.0	60.9
71954	2.7	22.7	10.0	12.4	11.7	40.5
71955	4.7	23.7	0.0	0.1	0.5	71.0
71956	0.0	0.0	0.1	0.4	1.0	98.5
71957	0.0	0.0	2.8	3.6	5.9	87.8
71958	0.0	0.0	2.8	3.6	5.9	87.8
71959	0.0	13.3	3.5	6.8	7.1	69.2
71960	0.0	13.3	3.5	6.8	7.1	69.2
71961	0.0	13.3	3.5	6.8	7.1	69.2
71962	0.0	13.3	3.5	6.8	7.1	69.2
71963	0.0	13.3	3.5	6.8	7.1	69.2
71964	0.0	13.3	4.2	6.1	6.4	69.9
71966	0.0	0.0	0.1	0.4	1.0	98.6
71967	1.7	38.3	2.2	3.0	4.2	50.6
71968	1.7	38.3	2.2	3.0	4.2	50.6
71969	1.7	38.3	2.2	3.0	4.2	50.6
71970	0.0	1.3	4.8	14.2	19.0	60.8

Roads

The forest road information was determined using the Washington Department of Natural Resources (DNR) GIS roads data and spatial data for roads in the Gifford Pinchot National Forest (DNR 2000; USFS 1995). The DNR roads were the primary source for the roads analysis. The USFS data was used to fill in gaps within the watershed, and to determine which DNR roads were decommissioned. Decommissioned roads were removed from the analysis. Road surface type was determined using multiple sources of information (BLM 1999; USFS 1995) and estimated from local knowledge where surface type was unavailable.

Road data were intersected with DNR and USFS soil data, and coded with the local soil type. The distance from each road segment to the nearest stream channel was calculated by further intersecting the roads with a GRID representing Euclidean distance from the stream channel.

Riparian Data

Estimates of riparian condition are used in DSS models (fish response, runoff and sediment models) that focus on reach-related measures of stream habitat. Remotely-sensed riparian data were combined to estimate shade, pool-forming conifers, and large-woody debris recruitment in the riparian model (Appendix H).

Land cover information from IVMP classified satellite imagery was used to estimate three measures of riparian conditions for each stream reach in the DSS (BLM 2001). Resolution of the IVMP data is $30m^2$. IVMP data includes estimates of tree size (as dbh) and percent vegetation cover for deciduous and coniferous forests in a series of GRID GIS data structures. It was the most appropriate data available to estimate vegetation cover in the absence of extensive field data, because of the time period and indicated accuracy of classification of this data set (BLM 2001).

The base riparian habitat for each segment was determined by summarizing the dominant canopy cover, stand type, and tree size within a 60 m zone from either side of the stream. We used a modified riparian buffer for streams with larger wetted widths and ponds and lakes, which were identified as those stream segments or lake segments with double-banked streams in the SSHIAP waterbodies spatial data set (WDFW 2004). In these cases, the 60 m riparian zone started at the edges of the wetted width boundary edge of the stream or lake, as defined by the edge of the double-banked stream polygon (WDFW 2004). Tree size estimates in the data were limited to conifers with >30% cover per cell. Due to the cell size of the source land cover information, riparian summaries include 3-4 land cover analysis GRID cells per thirty meters of stream, depending on the sinuosity of the segment.

Original GRID data were summarized into 10% bins for tree cover, and a series of bins for average conifer size. This information was summarized for each stream reach into the following fields that were used in DSS models: percent total tree cover; percent of trees that are coniferous; percent of trees that are deciduous; and average conifer size. We translated the GRID data into summarized percentages per stream reach as follows.

To calculate tree cover categories, we summed the median percent for each bin:

$$P_{cc} = \sum_{1}^{10} Mb\left(Vb/C\right)$$

$$P_{tt} = P_{cc} + P_{cd}$$

$$P_{tc} = P_{cc} / P_{tt}$$

Where:

P_{cc} = percent cover that is coniferous

P_{cd} = percent cover that is deciduous (calculated exactly as for coniferous)

P_{tt} = percent of cover that is in trees (a.k.a. total tree cover)

P_{tc} = percent of total tree cover that is coniferous

P_{td} = percent of total tree cover that is deciduous (calculated exactly as for coniferous)

M = median value in a bin (e.g., we used 15 as the median value for the 11-20% bin)

V = the GRID cell value in a given percent cover bin

C = count of the total cells associated with a stream reach

b1 = 0-10% bin; b2 = 11-20% bin; b3 = 21-30% bin … b10 = 91-100% bin

To calculate the average conifer size, we summed the median size for each bin:

$$S_c = \sum_{1}^{6} Mb\left(Vb/C\right)$$

Where:

S_c = average conifer size; diameter at breast height, in inches

V = the GRID cell value in a given size class bin

C = count of the total cells associated with a stream reach

b1 = 0-4.9” bin; b2 = 5-9.9” bin; b3 = 10-19.9” bin; b4 = 20-29.9” bin; b5 = 30-49.9” bin; and b6 = >50” bin. Bins identifying non-vegetated areas (cells with <70% vegetation) and cells where conifers were not >30% cover were not included in the average size calculation.

These data were used in the riparian function model (Appendix H) and were also used to estimate seral stage for the remotely-sensed spawner capacity model (Appendix I).

Barriers

Our base barriers information was from SSHIAP-WDFW WRIA 27 barriers spatial datasets (WDFW 2003, 2004) (Figure A-1). Because of the difficulty in differentiating the sensitivity of fish species to moderate barriers, we omitted partial blockages from the analysis and used only barriers that completely blocked fish migration, as indicated by the source information or historical accounts (Wade 2000). Barriers with questionable status were double checked with personnel from WDFW to assure data accuracy. We modified the status of a barrier on Brezee Creek based on input from the public works department of the City of La Center.

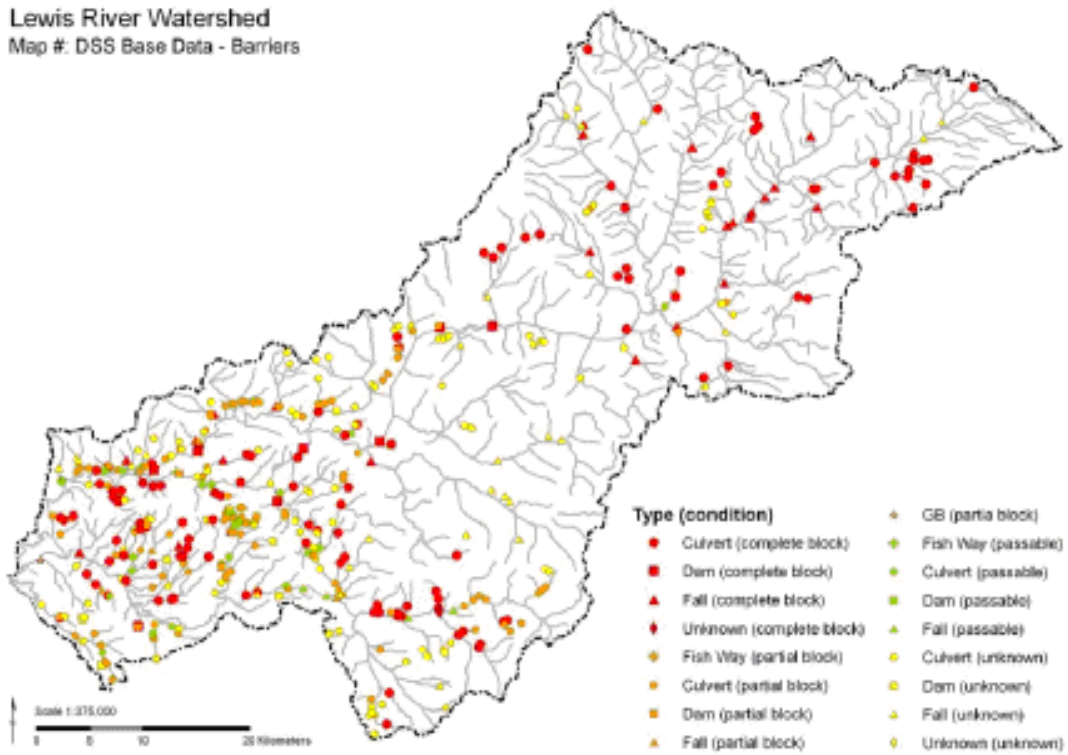

Figure A-1. Fish Barrier distribution in the Lewis River.

Fish Distribution

Distribution of salmon presence and potential spawning or rearing areas in the Lewis River watershed was delineated using SSHIAP documented fish distribution data (WDFW 2003) as our base data source (Figure 6 of the main report). Potential distribution (WDFW 2003) was defined by data on migration barriers, stream gradient, bankfull width, and other distribution information. Physical stream reach information was based on modeled stream network variables, as well as from the WDFW 2003 dataset. Actual fish distribution of fall chinook, spring chinook, winter steelhead, summer steelhead, coho, and chum includes aquatic habitat documented by Washington Department of Fish and Wildlife (WDFW) and local biologists as currently being used by each species or run-type, either for migration, spawning, or rearing (WDFW 2003). Based on WDFW documentation, distribution information is based on published sources, survey notes and first-hand accounts.

We also delineated potentially useable streams, streams with no documented use, but that are physically accessible to the particular species based on modeled physical reach variables. All streams were designated with a descriptive code that identified the stream segment as upstream or downstream of in-stream anthropogenic or natural barriers (Table A-1). The currently accessible documented and potential distribution for each species includes the WDFW (2004) documented tributaries and reaches, and all streams downstream of waterfalls, high gradient cascades, and manmade barriers to migration. The historical potential distribution includes all streams downstream of waterfalls and high gradient streams. Historical presence/absence documentation was also used to create the historical distribution (LCFRB 2004, Wade 2000) (Figure 6 of the main report).

Table A-1. Definition of codes used for classifying fish distribution and barrier distribution. These codes were combined with the SSHIAP-WDFW distribution guidelines and spatial fish distribution for each species to determine for each species what is currently accessible, what was historically accessible, and areas are naturally in accessible.

Code	Description	Distribution
A	generally accessible, currently	Current
H	historically accessible	Historical
IUCULV	Impassable section, upstream of culvert	Historical
IUCULV+DAM	Impassable section, upstream of impassable culvert and a dam	Historical
IUCULV+HYDRO	Impassable section, upstream of impassable culvert upstream of HYDRO facility	Historical
IUCULV_CONFLICTS	Impassable section, upstream of impassable culvert - using this in fishdist. likely will conflict w/ SSHIAP-WDFW fish distribution	Historical
IUDAM	Impassable section, upstream of impassable non-HYDRO dam	Historical
IUHYDRO	Impassable section, upstream of impassable HYDRO dam	Historical
IUNA	Impassable section, upstream of impassable natural barrier	None

Passable channel conditions for both current and historical distributions were defined as having channel width greater than 4 feet and channel gradients less than: 20% (winter and summer steelhead), 16% (spring and fall chinook, coho), or 5% (chum) (WDFW 2000). Distribution potential does not account for changes in water quality or habitat that may limit stream accessibility or productivity. Only totally impassable manmade barriers were considered to block upstream or downstream migration of salmon in the current distribution; streams with high concentrations of partially passable manmade or natural barriers may have limited migration and accessibility potential (Figure A-1).
Since there was not enough information above Lucia Falls when the original distribution data were created, literature was used to make corrections to fish distributions using the rules previously described. Occasional chinook and coho were seen above Lucia Falls but blocked at Sunset Falls whereas steelhead could go above Sunset Falls (LCFRB 2004, Washington State Department of Ecology 1999).

Adding recent restoration actions

To update our baselayer describing current conditions, we modified the GIS datalayers to reflect the restoration actions that occurred between 1998 and 2001. Completed restoration projects between 1998 and 2001 were collected from a variety of sources (Table A-1). Actions for each project included increasing habitat complexity, instream barrier removal, improving riparian forest, restoring floodplain connectivity and road decommissioning. After each project was identified, line and polygon data were converted to points and placed approximately at the center of each project. Some of the restoration actions were placed where base data displayed no changes. These were not included in our analyses.

The most current anthropogenic barrier information was obtained from WDFW in May 2004. Only barriers that are within 30 meters from streams were selected and snapped to the base stream layer, and our base barrier data was updated accordingly. Completed barrier restoration projects from SRFB and GP were already incorporated into the

WDFW's data. A modification to the mainstem of the East Fork Lewis River above Copper Creek was made based on literature (Clark County 2005; LCFRB 2004; WDFW 1999). Where barriers were opened up for fish access, fish distribution was also updated for each species without breaking the original rules of gradient, BFW, and potentially impassible waterfalls.

Road decommissioning data came in line, point, and polygon format. Existing roads were marked as decommissioned if the existing road lines overlapped with the decommissioned road lines. When data did not include decommission distance or did not intersect with the road, a 60-meter buffer was applied to them, and the road within the buffer was removed. If point data had a decommissioned distance, it was assumed that the point was located in the center of the restoration and half of the total restoration distance was removed on both sides of the point.

Stream segments for habitat structure restoration were selected based on the SRFB's project descriptions. REO's point data were converted to lines using stream miles indicated in the dataset. We applied a 30 m buffer from the point when there was no indication of distance. Spatial data were used where line data (REO) and project description did not match.

Riparian projects were not included in the update of current conditions because of obscure spatial information and lack of detailed project information.

Restoration activities and data sources are listed in Table A-2.

Table A-1. Restoration data sources. These data were collected between 1998 and 2001.

Organization	Source	Data Type
Gifford Pinchot National Forest (GP)	http://www.fs.fed.us/gpnf/forest-research/gis/	Point, Line, Polygon
NOAA	NOAA restoration center database in Sand Point, WA.	Point
Regional Ecosystem Office (REO)	http://www.reo.gov/restoration/	Point, Line, Polygon
Salmon Recovery Funding Board (SRFB)	http://www.iac.wa.gov/oiac/prism.htm	Point

Table A-2. Types of restoration activity and steps in the process.

Restoration Type	Data Source	Restoration Action
In Stream Barrier Removal	SRFB, GP	Barrier Removal, Barrier Update
Habitat Structure	SRFB, GP, REO	Boulders, Channel Connectivity, Deflectors, Gravel Placement, LWD, Off Channel, Rootwads, Weirs, Restored for Spawning
Riparian Forest	SRFB, GP, REO	Planting, Revegetation, Fencing, Restore Slide Areas, Streamside Revegetation, Streambank Stability, Riparian Thinning, Invasive Plant Control
Road Density	GP, REO	Road Decommission
Forest Cover	GP, REO	
Off-Channel/ Edge Habitat		

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Lewis River Case Study Final Report

A decision-support tool for assessing watershed-scale habitat recovery strategies for ESA-listed salmonids

Appendix B: Historical Lewis River Watershed Conditions

May 2007

Historical watershed conditions

Archival information, including the Journals of Lewis and Clark (Coues 1893) and the notes and maps of the Cadastral surveys that took place from 1853-1902 (BLM 2000), provide descriptions of soils, vegetation types, tree size, and channel conditions in the Lewis River watershed from the turn of the century. In addition, historical accounts (e.g., Rice 1996, Urrutia 1998) and historical photographs provide details of pre-European settlement conditions.

European settlement over the past 160 years has transformed the vegetation, soil properties, and flow regimes in the Lewis River watershed. The riverine environments in the Pacific Northwest are geologically young and physically dynamic because of glacial and tectonic activity such as volcanoes and earthquakes. Because of this, the freshwater habitat and riparian ecosystems are intricately linked with physical processes (Collins et al. 2003). Although geology and climate are primary controls on geomorphic and hydrologic processes and gross channel morphology, vegetation and soil are intermediary controls on physical processes, such as runoff regime and sediment erosion, as well as transport and delivery to a channel.

Archaeological evidence indicates that tribes fished for salmon up to the Lewis River falls (a barrier) in the Upper Lewis River watershed. Early settlers fished the North Fork Lewis and its tributaries, upstream of Cougar, for sport and sustenance (Rice 1996). The accounts say "trout," but that was the generic name frequently used: "the river is a mountain stream, pure and clear, abounds with the finest trout and other fish." Wade (2000) provides a comprehensive and detailed review of historical and current stock status and distribution limits of each salmonid species. The life-history diversity and spatial structure of Lewis River salmonid populations have declined in response to losses in available habitat for spawning and rearing, modified habitat conditions, and numerous other extrinsic factors.

The Pleistocene continental ice sheet did not extend to the Lewis watershed, but alpine glaciers and the Missoula flood left their mark on the landscape. Volcanic activity continues to influence the landscape. Mt. St. Helens has had numerous eruptions with tephra deposits, lahars, and debris and pyroclastic flows entering the Lewis watershed (Major and Scott 1988, PacifiCorp and Cowlitz PUD 2002). Pine, Muddy, Clearwater, and Smith watersheds are those more frequently affected by volcanic activity. Research indicates that hillslope sediment production on Mt. St. Helens blast-impacted areas decreases to background levels within 5-7 years (Dinehart 1997). Although hillslope erosion decreases, floods can remobilize the massive sediment inputs that were stored in the stream. The USGS sediment data indicate that stream sediment declined during the first decade but is still high. The farthest downstream known effect of Mt. St. Helens activity is downstream of Merwin Dam, 40,000 to 50,000 years ago. Lahar runout floods also extended toward Woodland approximately 440 years ago (Major and Scott 1988). There is no evidence of the volcanic flows extending past the Pine and Muddy Creek fans since then. It is therefore unlikely that volcanic activity before the dams substantially affected chinook salmon populations.

Dominant tree species in the Lewis watershed are Douglas fir and Western hemlock. Fire disturbance is an important factor in determining where Douglas fir becomes a co-dominant in the Western hemlock zone, and this has been the case in the Lewis watershed (Agee 1993). Archival journals, maps, and photographs were used to determine vegetation types, tree size, and channel conditions before significant settlement and watershed change occurred (Coues 1893; BLM 2000; Rice 1996; Urrutia 1998). Based on this information, the Lewis watershed consisted mostly of large conifer forests (e.g., Douglas fir, cedar, Western hemlock), with fertile meadows (e.g., Speelyai and Chelatchie Prairies). Throughout the lower half of the Lewis watershed, the vegetation was described as dense stands of fir, hemlock, some cedar, alder, and maple. The undergrowth had the same tree species with sallal, vine maple, dogwood, huckleberry, devils club, rosebush, hazel, elder, barberry, thimbleberry, salmon berry, and Oregon grape. Riparian areas had fir, cedar, maple, cottonwood, alder, hemlock, gooseberry, Oregon grape, hazel, vine maple, thimbleberry, and rose bush. Marker trees ranged from small to 144 inches dbh. In later surveys from 1902 (BLM 2000), large cut trees were frequently noted, as were logged areas covered with cut dead and green timber and dense undergrowth. Oak trees were abundant near the mouth of the Lewis and along the Lower East Fork. Black cottonwood and alder were important riparian species in the lower watershed.

Cadastral surveys noted numerous accumulations of “driftwood” in the Lewis, as did steamship captains in the late 1800s. The surveyors noted that the river was clear with a brisk current and ranged from 3 to 10 feet deep (at the Willamette meridian), and that the low bottoms overflowed annually. Soils along the river were mostly sandy. At the time of the cadastral survey in the Lower East Fork (1853; BLM 2000), the channel pattern was multi-channeled (anastomosing) with connected wetland habitat and side channels (starting at approximately RM 2). This pattern was still evident in the 1938 U.S. Geologic Survey (USGS) planar survey of the East Fork (to Lucia Falls) and the North Fork. Extensive wetlands in the scour channels from the Missoula flood are evident on the maps. The surveyors noted high, sandy eroding bluffs upstream of the multi-channeled reach. These are most likely remnant slack water deposits from the Missoula flood. The survey did not extend far past Lucia Falls because of the difficult terrain. Mass wasting events occurred throughout the watershed. The surveyors noted landslides and bank slumping along the Lower East Fork. Recent landslide inventories identified areas of natural instability and ancient deep-seated landslides (PacifiCorp and Cowlitz PUD 2002, PWI 1998).

Anthropogenic Modifications

Direct Channel Modification

In the late 1800s to early 1900s, the U.S. Army Corps of Engineers removed snags and deepened the North Fork to Speelyai Creek and the Lower East Fork channels for steamships. These actions, in addition to boat wakes, created extensive bank erosion, especially in areas where forests and shrubs had been cleared to the river for farming. One person reported that 3 acres were washed away from 1890-1908 (Rice 1996). In turn, the bank erosion led the farmers to demand bank protection, which changed the channel from multi-thread to single thread with disconnected sloughs and floodplains. Gravel mining increased this effect on both branches of the river. The combination of channel and floodplain modifications has led to a loss in floodplain connectivity (Wade 2000) and a straightening of the channels (PacifiCorp and Cowlitz PUD 2002, Report WTS3). This, in turn, has produced a loss in physical floodplain processes such as flood and sediment storage.

Levees

Levees were another modification that influenced the channel pattern and floodplain connectivity. Flooding was a problem for the settlers. Cadastral surveyors in 1853 noted that the river overflowed its banks on a yearly basis in the lower watershed. Before flood control was initiated, the Lewis flooded settlements in 1867, 1894 (which lasted several weeks), 1896, and 1917, suggesting that settled areas were in the active floodplain/floodway. This led to the construction of levees. The first successful levee was built near Woodland in 1921. The construction of levees not only modified channel pattern and floodplain connectivity, it also led to increased floodplain development along the Lower Lewis River (Rice 1996). Development in the floodplain continues to pose challenges for restoring important floodplain and side channel habitat.

Riparian Management

Prior to the railroad, the river was the primary source of transportation, and so the first trees to be logged were close to the river. Along with the large trees, smaller trees were removed from the riparian forest for cordwood to supply local, Vancouver, and Portland residences, schools, churches, and businesses. The riparian vegetation, along with the uplands, was also cleared for farmland by slashing and burning (BLM 2000, Rice 1996). Much of the riparian areas in the lower watersheds were cleared, and by 1860, commercial logging was done in earnest. Douglas fir was the large tree of choice. In 1871, the first commercial logging camp and sawmill in the watershed was started, followed by more mills. As the population increased, interest in cedar for roofing materials increased, and cedar became a heavily harvested species.

Roads

Roads are chronic sources of fine sediment (Reid and Dunne 1996), especially when they are not adequately maintained (NCASI 2002). In the Upper East Fork, the mainline unpaved roads are heavily used, which displaces aggregate surface material after a few weeks of vehicular travel (PWI 1998). While the recommended cross-drain spacing is

250-500 feet depending on road slope, cross-drain spacing on mainline roads in the watershed range closer to 900 feet. Spur roads that contour the slope have few to no cross drains and are supposedly out-sloped, although road segments on a number of spurs are in-sloped with no ditch. During high-intensity storms, segments of the road network in the Upper East Fork watershed (e.g., Forest Roads 53, 42, 4220, 41, and associated spur roads) transport water in the road treads, thereby effectively extending the channel network. Any one of these factors can increase road sediment production (NACASI 2002). The North Fork watershed analysis (USFS 1996) indicates that road conditions in upper hydrologic units (HUs) are similar to those found in the East Fork.

North Fork Lewis

Current vegetation in the Upper North Fork Lewis watershed is a mix of early, mid, and late seral stage forests, various aged clear-cuts, native grasslands, shrubs, burned areas, and rock and snow in the higher elevations. Vegetation in the lower North Fork Lewis River is dominated by agricultural uses, recreational grasses (e.g., golf courses), shrubs, native grasses, and forests. The natural fire regime is not known for the North Lewis; however, volcanic eruptions caused spot fires in the 1800s. It doesn't appear that any of these were widespread.

The construction of three large multi-purpose dams, beginning in the 1920s, was the primary change agent on the North Fork Lewis. Because the dams retain sediment and large woody debris (LWD), and alter the natural variation in peak and baseflow regime, they overshadow the influence of natural watershed processes on the freshwater habitat in the lower parts of the watershed. They are also barriers to upstream migration. The reservoirs, particularly Merwin and Swift, inundated what appears to be prime freshwater habitat, based on descriptions in the Cadastral surveys and photographs (BLM 2000), and descriptions of the area prior to the dam.

A photograph taken near Ariel before the dam shows that the channel was meandering, with a gravel-cobble substrate (Rice 1996). Before inundation, the river at Yale Reservoir had a meandering channel pattern with point bars. The floodplain was narrower than those in downstream reaches of the Lewis River (inundated by Merwin reservoir). Historical sources also indicate sloughs connected to the river where Merwin Reservoir is currently located. Throughout the river, gravel bars and islands loaded with LWD were noted, as were free and confined meanders (high bluffs).

Although the dams retain sediment, the channel below Merwin Dam does not appear to be sediment starved (PacifiCorp and Cowlitz PUD 2002, USFS 2002). Landslides may be the primary source of fine sediment to spawning gravels in the Lower North Fork. Fine sediment is delivered to the Lower North Fork through management-related mass wasting along Colvin and Johnson Creeks. Fines are approximately 92% of the total landslide sediment yield. The gravel sizes are within the range of preferred spawning sizes (PacifiCorp and Cowlitz PUD 2002, WTS3).

The channel pattern has changed from historical conditions. Sloughs and LWD accumulations are gone and the channel is straighter, possibly due to downstream gravel mining and efforts to reduce the migration of river meanders that threatened the highway in the 1940s and 1950s (PacifiCorp and Cowlitz PUD 2002). A reduction in active bars

and an increase in vegetated bars and islands occurred in this reach between 1939 and 1963-1974. The area of active bars has been relatively stable since 1974.

Near Yale Reservoir, the 1892 Cadastral survey described the channel as having rich bottomlands along the North Fork Lewis River and Speelyai Creek. The uplands on a glacio-fluvial terrace above the river were covered with a heavy growth of fir, cedar, hemlock, maple, and undergrowth similar to previous descriptions of the historical conditions. The loss of riparian areas has increased potential sediment delivery to channels and decreased sediment storage due to loss of LWD recruitment.

The reach downstream of Swift Dam has responded differently to the dam than the channel downstream of Merwin Dam. Temporal channel maps indicate that the active river channel width below Swift Dam decreased following closure (PacifiCorp and Cowlitz PUD 2002). Vegetation encroaches on the former active channel. The vegetation becomes uprooted during extremely large spill events that occur every decade or so, causing channel widening. This cycle continues to be repeated. USGS gauge height data indicate that the channel aggraded after the dam was constructed, probably due to reduced discharge and velocity, which caused a decrease in sediment transport capacity. Moreover, after the dam was built the sediment yields from the watersheds feeding the bypass reach increased due to logging. Substrate is dominantly cobble and small boulder. Most of the spawning-size gravel was located downstream from Ole Creek. Under existing conditions, median summer temperatures in the Swift bypass reach are at the upper end of preferred ranges for salmonids (PacifiCorp and Cowlitz PUD 2002). Existing maximum summer temperatures exceed the optimal ranges for all salmonid species, particularly bull trout. The reach at 1 to 1.5 miles downstream of Swift Dam does not have sufficient flow to provide any spawning habitat.

East Fork Lewis

Whereas the primary disturbance factor in the North Fork Lewis has been dam construction, fire has been a critical disturbance factor in the East Fork watershed. Fire disturbance is necessary to maintain Douglas fir as a co-dominant species in the Western hemlock (*Tsuga heterophylla)* zone (Agee 1993). Where Douglas fir is co-dominant with Western hemlock, the stand-replacement fire recurrence interval is 300-500 years. However, in Douglas fir-dominated associations, the recurrence interval is only about 100-150 years. From 1902-1927, three stand-replacing fires occurred within 10-20 years of one other. Human carelessness is suspected as the cause of these fires. However, logging practices exacerbated the fire intensity by creating more fuel (USFS 1986).

In 1902, salvage logging began after the Yacolt fire and continued until 1916. Salvage logging also occurred after the other fires that took place during the early 1900s. In the case of one large fire in the 1920s, salvage logging began before the fire was controlled (USFS 1986). To remove the dead trees, additional roads and skid trails were needed. Streams were often used as skid trails (PWI 1998). In 1953, a rehabilitation plan was developed to accommodate the logging and replanting of the burned areas. The plan included 50 miles of new access roads, 200 miles of bulldozed fire trails for vehicles, 200 miles of snag-free corridors, and development of 200 water holes for pumper and tanker trucks. On Forest Service land, many non-native species (e.g., Colorado blue spruce) were planted along bulldozed hillsides.

The fires, subsequent salvage logging, and the rehabilitation plan had more impact on the watershed-level processes in the upper East Fork than any other land change. The effects on soil and runoff patterns are still evident in portions of the watershed. The effect of the rehabilitation plan in the 1950s can be observed in the landslide records and peak flows in the East Fork (PWI 1998). Landslide frequency due to management-related actions accounted for 66% of total landslide activity that occurred between 1958 and 1997 (PWI 1998). The rehabilitation plan increased the drainage density by 40% through road and skid trail construction. The magnitude of high frequency (smaller) floods increased at the Heisson gauge (USGS gauge 14222500, RM 20, 1931-1996) during periods of road extension (1950s, 1970s-80s). No similar response was observed for gauging stations in the Upper North Fork.

Large wood was removed from the stream for salvage and to provide easier passage for machinery. Also, in the 1980s, the Washington Department of Fisheries removed many remaining logjams. The Upper East Fork channel response to removal of large wood was to change from forced pool-riffle morphology to plane bed morphology and to become armored (PWI 1998, Montgomery and MacDonald 2002). Both conditions create a situation where fine sediment is not readily stored even when channel gradient would imply storage. Fines are transported downstream to the Lower East Fork where channel gradient is substantially less. Excess fines in the Lower East Fork have been noted as a problem (Wade 2000, Rawding et al. 2001). Sources of fines include fires, gravel, surface erosion, and landslides, added road density, the natural high sand banks (part of the Missoula flood deposits), and bank erosion due to riparian changes.

The Upper East Fork appears to be sediment supply-limited (PWI 1998). Gravels are particularly in shorter supply from source areas, relative to other grain-size fractions, due to the nature of hillslope deposits and recent rates of sediment production. Based on the local geology and composition of older fluvial features (e.g., mainstem terraces), it is unlikely that the East Fork system has yielded an abundance of gravels in the past.

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Appendix C: Selecting Actions within Strategies

May 2007

Introduction

Details of how actions were selected for each of the 6 modeled watershed management strategies are described in this appendix. Appendix D describes how each type of action was modeled in the DSS.

Culvert removal

We removed the most cost efficient ($/km newly accessible habitat) barrier or culvert first. Barriers blocking the longest distance were often selected.

Riparian planting / Riparian restoration

Riparian planting only occurred on areas where riparian function was poor or fair and costs were not prohibitive. These included reaches for which more than 35% of the riparian area (within 20 m of the channel) was less than 5% slope, and more than 50% of the riparian area was not in bare ground, shrubs, or short grass. We started at the most upstream reach that met these criteria and worked our way downstream. A few expert panels specifically requested an alternative prioritization system. In these cases we also used agriculture ratio, fish distribution, sediment, and hydrologic input information.

Riparian protection

Protection of riparian areas only occurred where the riparian area (within 60 m of the channel) was not yet protected by county, state, or federal programs. We selected areas for protection starting at the most upstream reach that met all criteria. For the non-expert panel strategies, we protected good riparian habitat bordering streams that were suitable for spawning, as indicated from the remotely sensed habitat capacity and suitability model (Appendix I). Expert panels often provided additional instructions such as good riparian, all impaired riparian, in flood plain, lowest sediment input, etc.

Road decommissioning / Road improvements

We selected roads with the largest lateral road sediment input to the stream. If the amount of sediment was the same for 2 or more reaches, we selected the most upstream road. Some expert panels also requested that other information such as an agency list of prioritized roads be used in selecting roads for decommissioning or improvement.

Instream habitat improvement

All restore for spawning actions were dictated by an expert panel or by the EDT model (Appendix L). We selected the most upstream reach first where given a choice. Experts often instructed us to use additional information such as riparian condition or stream width. When the expert panel did not specifically define the selection area or selection process, we restored 50 m of habitat for every 10 km of stream.

Restore floodplain

All floodplain restoration actions were dictated by an expert panel or by the EDT model (Appendix L). We selected the most upstream reach first where given a choice. Experts often instructed us to use additional information such as reaches that currently have side channels and that currently have no dikes.

Dike removal

Only one expert panel under the expert panel strategy requested dike removals. We referred to the LCFRB map (WRIA 27 LEWIS-KALAMA WATERSHED Mass Wasting and Stream - Floodplain Connections Map A-4, LCFRB 2004a) to locate existing dikes and removed from upstream to downstream within budget constraints.

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Appendix D: Action Models

May 2007

Introduction

In this appendix, we describe how each type of restoration or preservation action was modeled. First, we selected stream reaches to be modified by a given restoration, protection, or degradation actions (Appendix C). Only stream reaches selected for an action were changed. Then, we modified the appropriate base datasets to account for the effects of actions in the selected reaches. Table D-1 provides an overview of what data are changed by each type of action. We then ran models on the modified datasets and summarized resulting metrics. Table D-2 details which models are affected by each type of action. After all actions for a particular strategy were modeled, it was then possible to evaluate the potential future landscape using the suite of landscape evaluation models described in the *Landscape Evaluation Models* section of this report.

Barrier removal

We used a two-step procedure that updated the amount of habitat accessible to anadromous fish based on the anthropogenic barriers selected for restoration or improved passage. The baseline barrier dataset contained information about whether each barrier was blocking or passable, and identified the next barrier downstream. When a barrier was restored, it was coded as passable, and the procedure recursively recoded all barriers that previously pointed to the newly restored barrier (Figure D-1). This information was then used to reclassify stream reaches as accessible, provided that all downstream barriers are passable. Only stream reaches that were historically accessible to a species were candidates for becoming accessible. Reaches with gradients over 20% or above natural barriers were considered inaccessible reaches.

Floodplain Restoration

Restoring connectivity between the main channel and side channels within the floodplain was modeled by increasing the length of the affected stream reach by 39.4%. This value was calculated as the average maximum change in length of channel versus associated side channels as measured from aerial photographs for 18 reaches from current (1990) to historical (1854, 1928, and 1955) conditions (Appendix M). We applied this action before any other strategy actions, so that the effects of any other actions would be reflected on the entire restored reach length (i.e., old reach length plus newly added length). The new reach length inherited habitat condition values from the original reach. This action could only be applied in reaches where floodplains were present historically.

Riparian Restoration and Protection

For riparian restoration and protection actions, we assumed a 50-yr time step. Benefits over 50 years were estimated and applied to the reach when the reach was restored or protected using the riparian model (Appendix H). To estimate the impact of protection over 50 years, we simply improved the riparian function scores (Table D-3) and seral stage values (Table D-4) by one value. If scores were already at the maximum value, there was no effect of protection. To estimate the impact of restoration over 50 years, riparian function scores and seral stage values were increased to the best possible level. If scores were already at the maximum value, there was no effect of restoration.

Additionally, riparian actions were reflected as changes in the riparian portions of the vegetation/land cover dataset. If protected or restored, the data code for riparian meta-polygons associated with the reach was improved to 20-yr forest for all classes except shrub, grass, or naturally bare ground, which we did not expect to become forested. Upland meta-polygon codes were not altered for these actions. In the rare case where restoration and protection actions were selected to occur on a single reach, protection occurred first and then restoration.

Instream Habitat Improvement

This action was designed to mimic placement of spawning gravel, and was the only static habitat improvement method directly employed. We reasoned that other habitat improvements (i.e., recruitment of wood and formation of pools) would be reflected by improvements in watershed processes affected by other actions (i.e., riparian restoration). Improvement of spawning habitat acted only on data inputs used to estimate spawner capacity (Appendix I). For small streams (≤25 m bankfull width), we improved mean redds/km on the affected portion of a reach to the 90th percentile value. New 10th (and 90th) percentile values were calculated as the new mean minus (or plus) the range from the old mean to the old 10th (or 90th) percentiles. For example, if we restored 4 m of a 6-m reach, of which mean redds/km was previously predicted to be 36.40 (7.97 to 61.30), new redds/km was calculated as follows: 2 m at old values, and 4 m at 61.30 (32.87 to 86.20) redds/km. The new redds/km for the entire reach was then 53.00, and ranged from 24.57 to 77.90 (10th to 90th percentiles). For large streams (>25 m bankfull width), we increased the area spawnable by 32%. Spawner capacity was then calculated based on these adjusted data values.

Road Improvement

We employed two levels of road restoration: decommissioning and improvement. This action was modeled by reducing the length of roads in the GIS layer. Decommissioning reduced length of roads by 95% (i.e., there was only 5% of road length remaining after restoration), whereas road improvements reduced road length by only 50%. Because actions acted on a reach-by-reach basis, the amount of roads remaining in the lateral drainage area of a stream reach was calculated as: (length of reach affected * level of restoration) + (length of reach unrestored). For example, if there were 5 km of roads decommissioned in the drainage area of a reach containing 7 km of roads, the resulting amount of roads in the reach was (5*0.05) + 2 = 2.25 km. Roads maintained their existing surface type (paved or unpaved) and underlying geology.

Table D-1. Data that are changed when modeling alternative conservation strategies. Actions (restoration, protection, or degradation) are in the left column. Data that change under each action type are listed in succeeding columns; actions happen only to reaches marked for conservation actions. See *Virtual Restoration and Degradation* section for details on specific changes made to data under each action category. All data not shown here do not change under modeled alternative strategies. Note that degradation only happens for strategies based on potential future conditions.

DATA→ ACTIONS↓	BARRIERS	RIPARIAN VEGETATION	VEGETATION/ LAND COVER	ROADS	STREAM REACH
upgrade or remove migration barriers	Changed to "passable," and corresponding fish distributions are updated to include newly opened habitat				
restore/protect riparian buffer		Modeled riparian function scores and seral stage values ↑ where possible to improve	In riparian areas, 20-year and 5-year forests ↑, clearcut and agriculture ↓		
restore/protect upland vegetation			In upland areas, 20-year and 5-year forests ↑, clearcut and agriculture ↓		
decommission or remove roads				Length of roads in marked drainage wings are decreased	
restore spawning gravels and instream large woody debris					Increased spawning capacity estimates (acts directly in model)
floodplain restoration (increase off-channel habitat by connecting to old channels)					Increased stream length available to fish
convert to urban or agriculture		Total tree cover and average conifer size ↓ in 75% of stream reach (25% automatically protected at existing value)	20-year and 5-year forests ↓, urban or agriculture ↑ (only on reaches without riparian protection measures)		

Table D-2. Models that are affected by data changes (see Table D-1) due to actions in conservation strategies. Actions (restoration, protection, or degradation) are in the left column. Models affected directly (or indirectly) by each action are listed in succeeding columns. See Appendix C through Appendix J for details on model development and parameters. Note that degradation only happens for strategies based on potential future conditions.

MODELS→ ACTIONS↓	RIPARIAN	SEDIMENT	HYDROLOGY	FISH MODELS
upgrade or remove migration barriers	Riparian Functions or Seral Stage (via opened habitat)	Sediment input and deposition (via opened habitat)	Runoff, Flood Discharge, and Scour (via opened habitat)	Accessibility, Suitability, Spawning Capacity, and Survival (all via opened habitat)
restore/protect riparian buffer	Riparian Function, Seral Stage	Surface Sediment, Road Sediment, Routed Fine Sediment	Runoff, Flood Discharge, Scour Potential	FishEye Suitability (via Riparian Function, Fine Sediment, and Scour Potential), Puget Suitability and Spawning Capacity (via Seral Stage), Survival (via Fine Sediment)
restore/protect upland vegetation		Surface Sediment, Mass Wasting, Routed Fine Sediment	Runoff, Flood Discharge, Scour Potential	FishEye Suitability (via Riparian Function, Fine Sediment, and Scour Potential), Survival (via Fine Sediment)
decommission or remove roads		Road Sediment, Routed Fine Sediment	Flood Discharge, Scour Potential)	FishEye Suitability and Survival (via Fine Sediment)
restore spawning gravels and instream large woody debris				Spawning Capacity
floodplain restoration (increase off-channel habitat by connecting to old channels)				Accessibility, FishEye Suitability, Spawning Capacity, and Survival (all via increased accessible reach length)
convert to urban or agriculture	Riparian Function, Seral Stage	Surface Sediment, Road Sediment, Routed Fine Sediment	Runoff, Flood Discharge, Scour Potential	FishEye Suitability (via Riparian Function, Fine Sediment, and Scour Potential), Puget Suitability and Spawning Capacity (via Seral Stage), Survival (via Fine Sediment)

Table D-3. New riparian function scores assigned to reaches that were protected or restored. If the existing score was naturally poor, the score did not change. If the existing score was already the highest possible, there was no change. Changes occurred in the poor and fair categories. Generally, protect actions increased scores by one level and restore actions increased scores to the highest potential where possible. The amount to which a score could be elevated depended on local conditions such as bankfull width, stream gradient, and existing percent of trees that are coniferous or deciduous; these are the same constraints used in the riparian function models used to create scores for current conditions.

Existing	Shade		Pool-forming conifers		LWD recruitment	
	Protect	Restore	Protect	Restore	Protect	Restore
naturally poor (0)	nat. poor	nat. poor	nat. poor	nat. poor	nat. poor	nat. poor
poor (1)	fair	good	fair	good	fair	good
fair (2)	good	good	good	good	good	good
good (3)	good	good	good	good	good	good
Exceptions	none		- ≤30% coniferous trees *or* - ≥40m bankfull width *or* - gradient >0.04 *or* - (gradient 0.02-0.04 *and* bankfull width 20-40m)		- >70% deciduous trees *and* ≥20m bankfull width	
Score for exceptions	n/a		naturally poor		fair	

Table D-4. New seral stage codes assigned to reaches that were protected or restored. Cottonwood-dominated areas were reaches designated as falling in floodplains or having Rosgen ratings of C; all other reaches were designated as conifer-dominated areas. Codes in bold represent changes; those in parentheses do not change. Naturally non-forested areas cannot change.

Existing	Protect		Restore	
	Cottonwood-dominated areas	Conifer-dominated areas	Cottonwood-dominated areas	Conifer-dominated areas
L (late seral conifer)	(L)	(L)	(L)	(L)
M (mid seral conifer)	(M)	**L**	(M)	**L**
E (early seral conifer)	**MIX**	**M**	**MIX**	**L**
MIX (mixed con/dec)	**D**	**E**	**D**	**E**
D (deciduous-dom.)	(D)	**MIX**	(D)	**MIX**
O (non-forested)	**D**	**E**	**MIX**	**E**

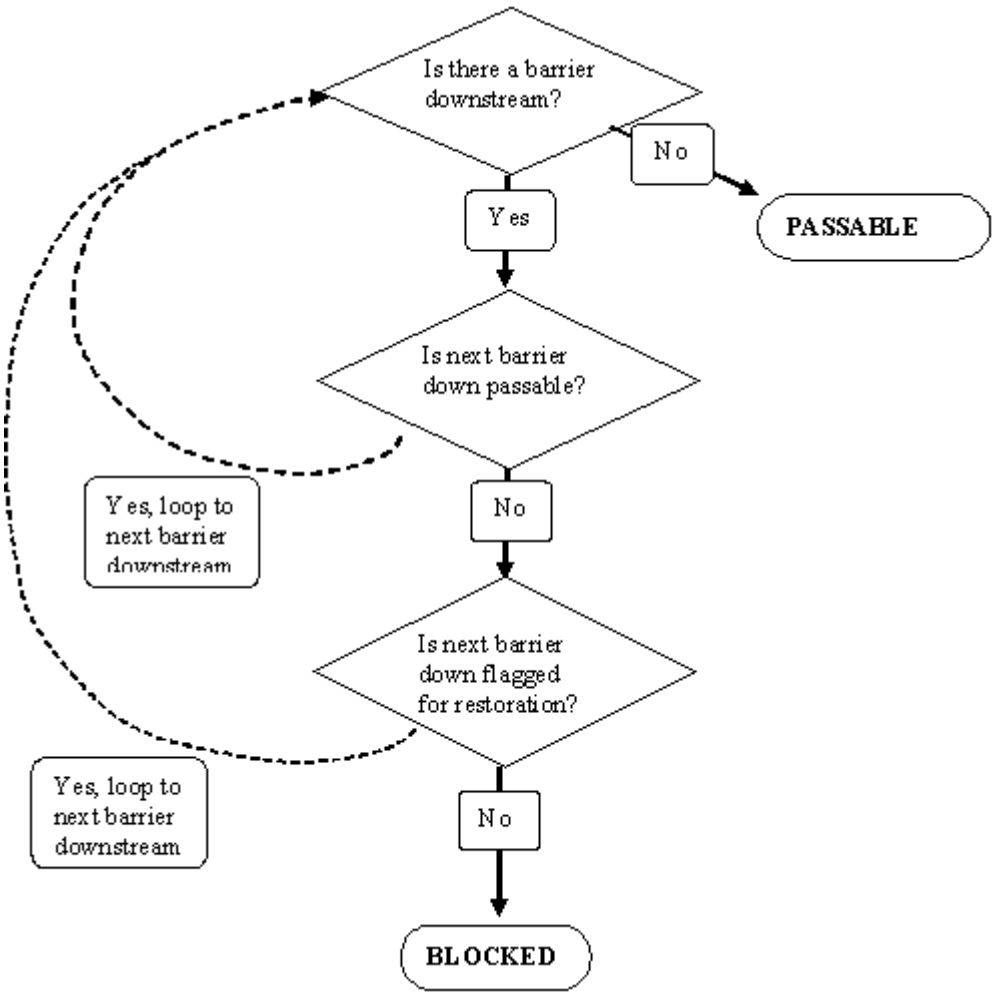

Figure D-1. Effect of barrier removals on accessibility of reaches to fish. After this barrier update procedure was completed, stream reaches were coded as currently accessible to fishes if all barriers in a reach were considered passable. Only reaches that were historically accessible to a given species were candidates for becoming accessible to that species.

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Appendix E: Surface Sediment, Erosion, and Runoff

May 2007

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Modeled Sediment and Runoff

We used a physically-based modeling approach to identify sediment sources and estimate sediment yield and runoff using GIS databases. Basin topography, soil type, and land-cover were the databases used in the models (described in Appendix A). Relative sediment and runoff production and impairment (low, moderate, high, none) were estimated based on the Water Erosion Prediction Procedure (WEPP) and the disturbed WEPP and WEPPROAD models (Flanagan and Livingston 1995; Elliot and Hall 1997; Elliot et al. 2000) as well as rules for landslide and road-erosion and runoff potential. The relative runoff change from historical to current conditions was based only on surface storm runoff. The impairment ratings were determined from the change in sediment yield and runoff between historical and current land cover conditions.

The primary objectives of the HU level analyses were to: 1) identify areas within subwatersheds which are more likely to deliver sediment and storm surface runoff to stream channels, 2) identify the relative contribution of three sediment sources—surface, mass wasting and road, delivered to streams, 3) provide a broad-scale context to reach-level sediment and hydrologic information and non-spatial historical stream hydrograph information, and 4) compare relative, qualitative differences between current and historical sediment and runoff levels to determine potentially impaired subwatersheds. In this appendix, we discuss field verification, sediment and runoff estimate corrections, and the tasks for developing estimates of sediment production and hydrologic runoff to be used in the Decision Support System (DSS). In addition to estimating sediment yield and runoff on the 7th field HU scale, dominant flood discharge data and land cover modifiers were determined. This information is used in the Sediment Routing model (Appendix F).

Verification of Runoff and Sediment Screens

Field verification and additional data were collected during the 2003 summer. The objective of field verification was to evaluate the results of the 7th field HU level sediment and runoff screens. We sampled 6 HUs to assess the assumptions used in the models (Figure E-1). The GIS datalayers examined for runoff were roads, soils, land cover, and DEM (slope). We visited the HUs noting evidence of surface runoff from hillslopes (e.g., gullies, slope rills/channels, headwater erosion, and material transported by overland flow) and roads. We also noted areas of groundwater emergence and erosion.

During field verification, we observed that in areas of ash-pumice soils, hillslope runoff and surface erosion responded differently than the general soil texture class (i.e., sand loam, silt loam, clay loam) attributed to these areas. Surface runoff and sediment transport were more common in these areas and were occurring in 10-20+ year old plantations (Figure E-2). Road runoff was more evident, including prism gullies on paved roads (Figure E-2). Field verification of modeled results indicated that another soil category, representing ash-pumice soils, was needed to improve surface and road sediment and runoff estimates. In addition, mass-wasting estimates needed revision. Based on these observations, the runoff and sediment yield rules for land cover, roads and mass wasting probability were modified to include a category for ash-pumice soil texture. The runoff and sediment yield values for these areas were changed and incorporated into the meta polygon level analysis (Appendix A). The WEPP models were used to estimate

surface runoff and sediment yield for the ash conditions. Riparian buffer modifiers were incorporated into the analysis. Where present, the riparian modifiers lessened surface sediment and runoff delivery.

Sediment and runoff adjustments

Model Storm Runoff and Surface Erosion

Hydrologic runoff and surface sediment yield were estimated using two versions of the WEPP model: WEPP hillslope model (http://topsoil.nserl.purdue.edu/nserlweb/weppmain/) and US Forest Service Disturbed WEPP and WEPPROAD models (http://forest.moscowfsl.wsu.edu/fswepp/). The surface erosion and runoff data were adjusted by adding another soil class for ash-pumice soils. Runoff production from agricultural lands was estimated using the WEPP hillslope model a numerical distributed process-based model (Lane et al. 1989, Flanagan and Livingston 1995). This model calculates surface and shallow subsurface storm runoff, i.e., runoff from the rooting zone, and surface sediment yield. Surface erosion and storm runoff rates from forest, shrub, native grasslands, clear cuts, and fire were estimated using the Forest Service Disturbed WEPP model (Elliot and Hall 1997; Conner, et al. 2000; Elliot et al. 1995; Elliot et al. 2000).

Both models are mostly used for estimating sediment erosion and water availability on a single hillslope or small watershed scale. The lateral drainage wings meet the scale criteria for small watershed (e.g., Lane et al. 1997). WEPP simulates the conditions that impact erosion for every day in a multiple-year run based on the amount of vegetation canopy, the surface residue, and the soil water content. Variables used are vegetation type, vegetation management, cover, soil texture or series, slope and climate. For each day that has a precipitation event, WEPP determines whether the event is rain or snow, and calculates the infiltration and runoff based on local precipitation and storm patterns. When there is excess precipitation, WEPP routes the excess precipitation as surface runoff, calculating erosion or deposition rates for at least 100 points on the hillslope. It then calculates the average sediment yield and runoff from the hillslope. The U.S. Forest Service Disturbed WEPP model assumes two overland flow elements. Users can specify buffers below a skid trail, prescribed fire, or harvesting activity in forests.

The soil texture classes, slope classes and land cover used in the WEPP model are described in Appendix A, as well as Table E-1 and Table E-2. The models were used on multiple land cover scenarios, historical, current, and potential (13 landscapes). Based on the available information, the historical scenario is a forested watershed with interspersed meadows, wetlands, and other covers, and no large stand-replacing fires. Natural disturbances such as fire and volcanic activity can produce much more sediment than some management activities. However, there is not adequate information to incorporate these into the model with the exception of ash-pumice soils. No extensive natural stand-replacing fires were evident in the recent history (since the late 1700s). There have been extensive stand-replacing fires caused by human error and logging practices in the early to mid 1900s. Historical eruptions from Mt. St. Helens caused spot fires but most were of small extent (Agee 1993). The soils are assumed to be the same as for the current condition.

The USFS Disturbed WEPP model did not have an option for old growth so the 20-year forest category was used to represent mid-to-late seral stage. Although 20-year forests are not defined as hydrologically mature (WFPB 2000), WEPP runoff and sediment estimates approach background levels for this land cover. We assume for this analysis that sediment yield and surface storm runoff estimates in 20-year forests are in the same order of magnitude as data from old growth areas in similar geology and in the Pacific Northwest (Larsen and Sidle 1980, Reid and Dunne 1996, PWI 1998, PacificCorp and Cowlitz PUD 2002). Agricultural and rangeland areas were considered to have only one slope class, so sediment yield and runoff only varied by type of agriculture and not by topographic class. The agricultural model incorporates bare ground. Urban areas are treated separately. Zero sediment is assumed for rocky areas.

Mass Wasting and Road Erosion

Natural sediment production from mass wasting was estimated using landslide data from unmanaged HUs and landslide inventories in the Lewis watershed (PWI 1998, PacificCorp and Cowlitz PUD 2002). Management-related landslide sediment yield was extrapolated from existing landslide inventories. Field verification indicated that the 7th field HU assessment mass wasting estimates were inaccurate in areas of ash-pumice soils. Unlike surface runoff and erosion, mass wasting appeared to occur less often than predicted. New rules concerning presence or absence of ash-pumice soils were incorporated into the DNR mass wasting GIS model to better represent conditions on the ground. For ash-pumice soil, the percent slope was increased for each mass wasting probability class, e.g., for concave slopes the percent slope for high mass wasting probability was increased from 50% to 80%. New rules on land cover and road density were added to the decision support mass wasting model (Figure E-2).

In the HU level sediment screen, road erosion rates were calculated from rates previously estimated (PWI 1998, PacificCorp and Cowlitz PUD 2002) and modified by road surface and presence of stream adjacent roads. Road conditions in the upper watershed were extrapolated from the road survey done for the Upper East Fork (PWI 1998). The road survey was based on the protocol outlined in Washington Forest Practices Board Watershed Assessment Manual (1997).

Following field verification, road runoff and surface erosion models were refined from the HU estimates using the U.S. Forest Service programs WEPPROAD (Elliot et al. 1995, Elliot et al. 2000) and an ash-pumice soil class was added (Figure E-3). The road parameters were based on previous road surveys (PWI 1998) and road surveys conducted during 2003 field verification. Paved roads are assumed to contribute sediment from road fill slopes only. Based on estimates in the Washington Forest Practices Manual (WFPB 2000), fill slopes contribute 20% of unpaved road sediment. From road surveys, we adjusted this to 10% of unpaved road sediment for paved roads.

Distance and riparian modifiers were developed from WEPPROAD models (Figure E-3, Table E-3). Riparian buffer width was 33 meters for the analysis. These modifiers were used to attenuate both runoff and sediment delivery to streams as distance from road crossings and riparian buffers increased (Table E-3).

Sediment size distribution

The sediment routing model, described in Appendix F, used sediment size distributions. Sediment size distributions for surface and road sediment yields were obtained from the SSURGO databases and soil surveys for each county in Lewis basin (McGee 1972, Call 1974, Haggen 1990, NRCS 2004). In the database, each soil series (MUKEY in the database) has a distribution based on percent of size greater or less than a given sieve size. Soil information from the Natural Resource Conservation Service (NRCS) was not available on USFS lands. Sediment size data for similar soil series were extrapolated to the USFS soil database. Sediment sizes were distributed into 6 size classes (Table E-4). The SSURGO database does not provide information to estimate sediment sizes from mass wasting however. The sediment size distribution was estimated from mass wasting assessments from the Tilton and East Fork Lewis watershed as described in Appendix F.

Estimated 2.3-year flood discharge

We estimated the 2.3-year recurrence-interval flood discharge ($Q_{2.3}$) as an indicator of the mean annual flood and channel forming and bankfull flow (Black 1991, Whiting et al. 1999). Flood frequency and sediment transport analysis in the East Fork indicate that this flood is the average flood that initiates bedload transport (PWI, 1998). A sediment movement study done in Ole Creek indicates that the 2-2.5 year floods initiate bedload transport (PacificCorp and Cowlitz PUD 2002).

We used regression analyses to develop equations for estimating the 2.3-year flood discharge. The regression variables are 2.3-year flood measured at all USGS gauging stations in the Lewis watershed and drainage area above the gauges. In the East Fork Lewis River we had additional flood discharge information to develop a separate equation for that subwatershed. The best-fit equation for the $Q_{2.3}$ is a power function of drainage area:

> $Q_{2.3}$ = 4.235*A^0.929, see=0.449, r^2adj=0.90 (from gages above Merwin reservoir)
>
> $Q_{2.3}$ = 4.4003*A^0.9132, see=0.46, r^2adj=0.91 (for the upper East Fork Lewis)
>
> Where $Q_{2.3}$ is in ft^3/sec and drainage area A is in mi^2.

The average duration of the 2.3-year flood, 2.1 days, was estimated from the peak flow records from gauges in the Lewis River watershed. The duration and equations used in the sediment routing model are described in Appendix F. The 2.3-year flood discharge equations are used for the “unaltered case” describing hydrologic and hydraulic conditions in the watershed. Although the landscape had been disturbed before most gauges were established, we assume that over time the effects of disturbance on flood magnitude are attenuated. Land use, such as urbanization, conversion to agriculture, and timber harvest activities alter runoff processes over shorter time spans than the gauged periods of record in the Lewis River watershed (e.g., Booth and Jackson 1997, Jones and Grant 1996, Harr 1986). Analyses of temporal homogeneity showed no long-term trends in discharge data at gauges or in climatic data.

Land use modifiers (2.3-year flood)

An estimate of potential modification to the 2.3-year flood magnitude from land use activities is useful for providing information to the sediment routing model for evaluating the effect of restoration scenarios related to land management. The WEPP-generated runoff provides an indicator of changed conditions for only surface and shallow subsurface components (e.g., root zone) of stream runoff. However, surface storm runoff is only a small percentage (1-10%) of total stream runoff in forested watersheds in western Washington (Harr 1986, Dunne 1990). For example, the WEPP generated surface storm runoff contributions to total 2.3-year flood runoff were estimated to evaluate the reasonableness of the WEPP generated surface storm runoff contributions to total flood runoff (Table E-6). The results indicate that the WEPP generated runoff is less than 10% of the discharge. Consequently we needed an order of magnitude method to estimate increase in all storm related runoff (surface and subsurface) that does not require a precipitation-runoff model or generating hydrographs.

Three land use categories—forest harvest activities, agriculture, and urban, were used to estimate relative increases in 2.3-year floods from these activities. The modifiers developed were added to the DSS database. The estimates would be relevant for questions such as:

- If trees are grown in cluster x, what would be the relative decrease in 2.3-year flood, sediment transport, and potential spawning gravel scour in the associated reach, downstream reaches?
- If urban areas increase in cluster y by x%, what would be the relative increase in 2.3-year flood, sediment transport, and potential scour in the associated reach, downstream reaches?

Forest Harvest

Approximate increases in the 2.3-yr flood magnitudes for forested and roaded areas were estimated from storm precipitation, peak flow data, and stream runoff response to storms (e.g., Black 1991, Leopold and Dunne 1978):

$$Sr = \text{Storm runoff} / \left(\sum \text{precipitation}_{to \cdots t5}\right)$$

Where:

Sr is storm response

Storm runoff is in mm of water

Precipitation $_{to \cdots t5}$ is daily precipitation on the day of peak discharge ($_{to}$), back to 5 days preceding the peak ($_{t5}$).

The storm response index gives an indication of the response of a particular watershed to storm events. The higher the value, the more precipitation contributes to peak flow. The 5-day precipitation sum is from standard NRCS procedures on estimating storm volume (US Soil Conservation Service 1972). The storm response index is mostly associated with smaller flood events (<10-year flood). A similar analysis was originally done for the East

Fork at the Heisson gage (1927-1998 data) using precipitation data from Wind River climate station (PWI 1998).

Gauges in North Fork Lewis watersheds were evaluated for similar storm runoff response as found in the East Fork. Analyses of temporal and spatial homogeneity showed there was sufficient similarity in hydrologic patterns to use the data from East Fork to estimate the relative change to 2.3-year flood throughout the Lewis watershed.

The percent increases in flood discharge are storm response averages from known burn, road building, and road and harvest periods identified in the East Fork data (Table E-7). The increases include all storm runoff—surface and subsurface. The percent increases are similar to those found in other studies (e.g., Bowling and Lettenmeir 1997, Bowling and Lettenmeir 2001, Lewis et al. 2001).

Agricultural

The full WEPP model was designed to evaluate runoff, soil moisture conditions, and erosion on agricultural lands. This model addresses subsurface storm runoff more thoroughly than the Disturbed WEPP for forested areas. Accordingly we used the full WEPP model to estimate the relative increase in the 2.3-year flood for agricultural land uses. Two climate stations, Battleground and Packwood, were used in the analysis. The WEPP hillslope model was run for the 5 dominant soil series in the agricultural area in the Lewis and 3 agricultural covers—grass, row crop, and fallow. The soil series were grouped into 4 textural classes—silt, silt loam, sand loam, and clay loam. The modifiers are an average of the two climate stations and 3 agricultural covers classified by soil texture (Table E-8). The modifiers can be used for other grass conditions such as golf courses or play fields.

Urban

While “effective impervious area” (EIA) provides a measure of urban impact on streams, it does not provide a means to estimate potential increase in peak flow. We used data from a study conducted in the Puget Lowlands (Moscrip and Montgomery 1997) to develop the 2.3-year flood urban modifier. The modifier is a regression equation that equates percent of area in urban to a ratio of the 2.3-year flood post-urbanization to the 2.3 year flood pre-urbanization:

$$Q_{2.3post} / Q_{2.3pre} = 0.0298x+0.9255, \text{ see}=0.264, r^2\text{adj}=0.73,$$

where x is the percent urban area.

The prediction results compare favorably to data from other studies (Dinocola 1989; Hollis 1975, Richey 1982). The equation shows that when percent urban is less than 2.5%, the modified $Q_{2.3}$ is less than the unmodified $Q_{2.3}$ (Table E-8). In these cases, the ratio between the two flows is assumed to be 1.0. Forcing the x-intercept to equal 1 (ratio equal to one is assumed to be the origin) provides similar results (Figure E-4). In effect using the equation as is will slightly underestimate (<1%) the discharge ratio for areas where % urban is less than 20% and slightly overestimate (<2%) where % urban is greater than 50%. Not forcing the regression model to go through 1 avoids difficulties resulting from forcing the model when it may not be appropriate. These results are not

unlike the research on the effects of effective impervious area on peak flow (Booth and Jackson 1997, Hollis 1975). These studies indicate that when EIA is less than 3% than there is no impact on hydrologic conditions. Percent EIA is most often less than percent urban area.

Integration into the DSS

For running scenarios, values from modeled current conditions were permanently stored in lookup tables (Table E-1, Table E-2). To calculate surface sediment and hydrologic runoff in lateral drainage areas of individual segments, we summed area-weighted values from the lookup table for each land use category. Sediment input to streams was reduced by 45% on non-ash soils and by 38% on ash soils when riparian conditions were deemed to be functioning (Appendix H).

For running scenarios, values of road-derived sediment and runoff were calculated for the lateral drainage wings of each stream reach. Road sediment and runoff input to streams was reduced by functioning riparian conditions, as was surface sediment and runoff. Model parameters are detailed in Figure E-3. Values of mass wasting-derived sediment were calculated for the lateral drainage wing of each stream reach. Model parameters are detailed in Figure E-2.

The sediment models provided estimates of sediment yield by source to each drainage wing stream reach. The 2.3-year flood modifiers were also added to the DSS database. All output variables were then incorporated into the sediment routing model (see Appendix F).

Table E-1. WEPP estimates for sediment surface erosion (kg/m2/yr).

Vegetation	Soil Type	SlopeCode					
		≤10%	10-20%	20-30%	30-40%	40-50%	>50%
Clearcut	silt loam	0.008	0.020	0.033	0.045	0.055	0.068
Clearcut	clay loam	0.002	0.005	0.010	0.018	0.025	0.035
Clearcut	sand loam	0.000	0.002	0.005	0.010	0.011	0.015
Clearcut	ash, pumice subsoil	0.016	0.041	0.065	0.090	0.111	0.135
20_year	silt loam	0.002	0.005	0.010	0.012	0.015	0.020
20_year	clay loam	0.000	0.000	0.002	0.002	0.005	0.008
20_year	sand loam	0.000	0.000	0.000	0.002	0.002	0.002
20_year	ash, pumice subsoil	0.005	0.009	0.020	0.025	0.029	0.041
5_year	silt loam	0.008	0.020	0.033	0.045	0.055	0.068
5_year	clay loam	0.002	0.005	0.010	0.018	0.025	0.035
5_year	sand loam	0.000	0.002	0.005	0.010	0.011	0.015
5_year	ash, pumice subsoil	0.016	0.041	0.065	0.090	0.111	0.135
Shrubs	silt loam	0.000	0.002	0.010	0.025	0.030	0.037
Shrubs	clay loam	0.005	0.005	0.010	0.010	0.015	0.020
Shrubs	sand loam	0.000	0.000	0.000	0.000	0.000	0.002
Shrubs	ash, pumice subsoil	0.001	0.002	0.010	0.025	0.030	0.037
Grass	silt loam	0.055	0.132	0.198	0.253	0.294	0.328
Grass	clay loam	0.024	0.059	0.102	0.143	0.174	0.208
Grass	sand loam	0.002	0.010	0.027	0.043	0.058	0.071
Grass	ash, pumice subsoil	0.110	0.264	0.396	0.506	0.588	0.656
Urban_rock	silt loam	0.000	0.000	0.000	0.000	0.000	0.000
Urban_rock	clay loam	0.000	0.000	0.000	0.000	0.000	0.000
Urban_rock	sand loam	0.000	0.000	0.000	0.000	0.000	0.000
Urban_rock	ash, pumice subsoil	0.000	0.000	0.000	0.000	0.000	0.000
ag_alfalfa	silt loam	0.628	0	0	0	0	0
ag_alfalfa	clay loam	0.628	0	0	0	0	0
ag_alfalfa	sand loam	0.628	0	0	0	0	0
ag_alfalfa	ash, pumice subsoil	0.628	0	0	0	0	0
ag_row	silt loam	0.373	0	0	0	0	0
ag_row	clay loam	0.373	0	0	0	0	0
ag_row	sand loam	0.373	0	0	0	0	0
ag_row	ash, pumice subsoil	0.373	0	0	0	0	0
ag_recgrass	silt loam	0.201	0	0	0	0	0
ag_recgrass	clay loam	0.201	0	0	0	0	0
ag_recgrass	sand loam	0.201	0	0	0	0	0
ag_recgrass	ash, pumice subsoil	0.201	0	0	0	0	0

Table E-2. WEPP estimates for hydrologic surface runoff (mm of water/ yr/m2).

Vegetation	Soil Type	SlopeCode					
		≤10%	10-20%	20-30%	30-40%	40-50%	>50%
Clearcut	silt loam	100.584	100.584	100.076	100.076	100.076	100.584
Clearcut	clay loam	147.828	150.622	150.622	152.146	152.146	152.146
Clearcut	sand loam	42.164	43.434	43.434	43.434	43.434	43.434
Clearcut	ash, pumice subsoil	204.216	207.772	207.772	206.248	207.772	205.994
20_year	silt loam	6.35	5.842	6.096	6.35	6.604	6.604
20_year	clay loam	14.986	12.446	14.224	14.224	14.224	13.716
20_year	sand loam	2.286	2.286	2.286	2.286	2.286	2.286
20_year	ash, pumice subsoil	133.812	135.228	135.228	135.228	135.582	135.582
5_year	silt loam	9.906	10.668	10.668	10.668	10.668	10.922
5_year	clay loam	19.812	19.304	19.812	19.558	19.812	19.05
5_year	sand loam	2.794	2.794	3.048	3.048	3.048	3.048
5_year	ash, pumice subsoil	147.828	150.622	150.622	152.146	152.146	152.146
Shrubs	silt loam	0.508	1.778	4.064	6.604	6.858	6.604
Shrubs	clay loam	15.24	16.256	16.764	14.986	15.24	15.748
Shrubs	sand loam	0.508	1.016	1.27	1.778	1.778	1.778
Shrubs	ash, pumice subsoil	15.24	16.256	16.764	14.986	15.24	15.748
Grass	silt loam	22.098	22.606	22.606	22.606	22.352	22.352
Grass	clay loam	36.830	38.862	38.608	38.354	36.830	38.862
Grass	sand loam	6.096	6.858	7.112	7.366	7.366	7.366
Grass	ash, pumice subsoil	36.830	38.862	38.608	38.354	36.830	38.862
Urban_rock	silt loam	233.426	244.094	244.094	244.348	244.348	244.348
Urban_rock	clay loam	233.426	244.094	244.094	244.348	244.348	244.348
Urban_rock	sand loam	233.426	244.094	244.094	244.348	244.348	244.348
Urban_rock	ash, pumice subsoil	233.426	244.094	244.094	244.348	244.348	244.348
ag_alfalfa	silt loam	213.36	0	0	0	0	0
ag_alfalfa	clay loam	213.36	0	0	0	0	0
ag_alfalfa	sand loam	213.36	0	0	0	0	0
ag_alfalfa	ash, pumice subsoil	213.36	0	0	0	0	0
ag_row	silt loam	224.3667	0	0	0	0	0
ag_row	clay loam	224.3667	0	0	0	0	0
ag_row	sand loam	224.3667	0	0	0	0	0
ag_row	ash, pumice subsoil	224.3667	0	0	0	0	0
ag_recgrass	silt loam	191.3467	0	0	0	0	0
ag_recgrass	clay loam	191.3467	0	0	0	0	0
ag_recgrass	sand loam	191.3467	0	0	0	0	0
ag_recgrass	ash, pumice subsoil	191.3467	0	0	0	0	0

Table E-3. Sediment and runoff delivery modifiers are based on road distance from stream or stream crossing and riparian modifiers assuming a 33-meter buffer. Distance and riparian modifiers were developed from the WEPPROAD model.

Unpaved roads	Sediment			Runoff		
Road distance from stream or road crossing of stream (m)	Distance reduction	kg/m2/yr per unit of road prism	Riparian modifier	Distance reduction	mm/m2/yr per unit of road prism	Riparian modifier
Clay loam						
0-62		6.2	45%		0.86	62%
62-155	36%	2.21		23%	0.20	
155-248	19%	1.16		14%	0.12	
248-371	11%	0.70		9%	0.08	
371-495	8%	0.48		6%	0.05	
495-681	5%	0.30		5%	0.04	
>681	0%	0.00		0%	0.00	
Silt loam						
0-62		6.4	62%		0.69	32%
62-155	36%	2.30		22%	0.15	
155-248	19%	1.21		13%	0.09	
248-371	11%	0.73		9%	0.06	
371-495	8%	0.50		6%	0.04	
495-681	5%	0.35		4%	0.03	
>681	0%	0.00		4%	0.02	
Sand loam						
0-62		4.5	38%		0.38	72%
62-155	29%	1.32		24%	0.09	
155-248	15%	0.67		15%	0.06	
248-371	9%	0.39		9%	0.04	
371-495	6%	0.27		7%	0.03	
495-681	4%	0.19		4%	0.02	
>681	0%	0.00		0%	0.00	
Ash soils						
0-62		24.7	38%		0.9	9%
62-155	36%	8.78		36%	0.32	
155-248	19%	4.62		19%	0.17	
248-371	11%	2.78		11%	0.10	
371-495	8%	1.90		8%	0.07	
495-681	5%	1.20		5%	0.04	
>681	0%	0		0%	0	

Table E-4. Sediment size distributions (mm) for surface and road sediment yields were obtained from the SSURGO databases and soil surveys for each county in Lewis basin (NCRS gis database). In the database, each soil series (MUKEY in the database) has a distribution based on percent of size greater or less than a given sieve size. Sediment sizes were distributed into 6 size classes that were then incorporated into the DSS Access database

MUKEY	> 78mm (Cobble) GT78_PRC	>4.8-78mm (Coarse gravel) GT4.8_LT78	1.0-4.8mm (V. Coarse sand to gravel) LT4.8_GT1	<1.0-0.5mm (Coarse sand) LT1_GT.5	<0.5-0.25mm (Med sand) LT.5_GT.25	<0.25mm (Fine sand and less) LT.25_PRC
71952	0.0	28.0	2.5	3.6	5.0	60.9
71953	0.0	28.0	2.5	3.6	5.0	60.9
71954	2.7	22.7	10.0	12.4	11.7	40.5
71955	4.7	23.7	0.0	0.1	0.5	71.0
71956	0.0	0.0	0.1	0.4	1.0	98.5
71957	0.0	0.0	2.8	3.6	5.9	87.8
71958	0.0	0.0	2.8	3.6	5.9	87.8
71959	0.0	13.3	3.5	6.8	7.1	69.2
71960	0.0	13.3	3.5	6.8	7.1	69.2
71961	0.0	13.3	3.5	6.8	7.1	69.2
71962	0.0	13.3	3.5	6.8	7.1	69.2
71963	0.0	13.3	3.5	6.8	7.1	69.2
71964	0.0	13.3	4.2	6.1	6.4	69.9
71966	0.0	0.0	0.1	0.4	1.0	98.6
71967	1.7	38.3	2.2	3.0	4.2	50.6
71968	1.7	38.3	2.2	3.0	4.2	50.6
71969	1.7	38.3	2.2	3.0	4.2	50.6
71970	0.0	1.3	4.8	14.2	19.0	60.8

Table E-5. The WEPP generated surface storm runoff contributions were compared to total 2.3-year flood runoff to evaluate the reasonableness of the WEPP generated surface storm runoff contributions to total flood runoff. The results indicate that the WEPP generated runoff is less than 10% of the discharge.

Surface runoff as a % of the 2.3-year flood runoff				
Soils/land cover	Ash soils	Clay loam	Silt loam	Sand loam
Clearcut	2.3%	1.9%	2.2%	1.9%
5-year forest	2.2%	1.4%	1.95%	0.5%
20-year forest	1.9%	1.2%	1.6	0.5%

Table E-6. The percent increases in the 2.3-year flood discharge are storm response averages from known burn, road building, and road and harvest periods identified in the East Fork data. The harvest increase is the difference between harvest and roads and road categories. The increases include all storm runoff—surface and subsurface

Flood frequency	% Increase in flood discharge			
	Burn	Harvest	Road	Harvest & Roads
2.3 year	19%	8%	20%	28%

Table E-7. Data generated from the WEPP hillslope model was grouped in soil texture classes. Agricultural covers—grass, row crop, and fallow were averaged for Battleground and Packwood climate stations. The unadjusted 2.3-year flood is multiplied by the modifier to get new 2.3-year flood values.

Agricultural modifier for increasing 2.3-yr flood (Average)				
All soil types	Silt loam	Clay loam	Sand loam	Ash
1.65	1.36	1.12	2.34	N/A

Table E-8. Values of post-urban discharge for a given pre-urban discharges. Values are calculated using the regression equation for the ratio of post-urban 2.3-year flood discharge to pre-urban 2.3-year flood. The equation, based solely on data from Moscrip and Montgomery (1997), without the x-intercept forced to equal one is used. When percent urban area is less than 2.5 than the discharge ratio is less than 1. Since this is not a realistic case, we assume that for areas with less than 2.5% urban that the ratio is equal to 1.

% urban	Pre urban Discharge				
	100	**200**	**300**	**500**	**1000**
	Post Urban Discharge results				
2.45	99.9	199.7	299.6	499.3	998.6
10	122.4	244.8	367.1	611.9	1223.8
20	152.2	304.4	456.6	761.0	1522.1
30	182.0	364.1	546.1	910.2	1820.4
40	211.9	423.7	635.6	1059.4	2118.7
50	241.7	483.4	725.1	1208.5	2417.0
60	271.5	543.1	814.6	1357.7	2715.4
70	301.4	602.7	904.1	1506.8	3013.7
80	331.2	662.4	993.6	1656.0	3312.0
90	361.0	722.1	1083.1	1805.2	3610.3
100	390.9	781.7	1172.6	1954.3	3908.6

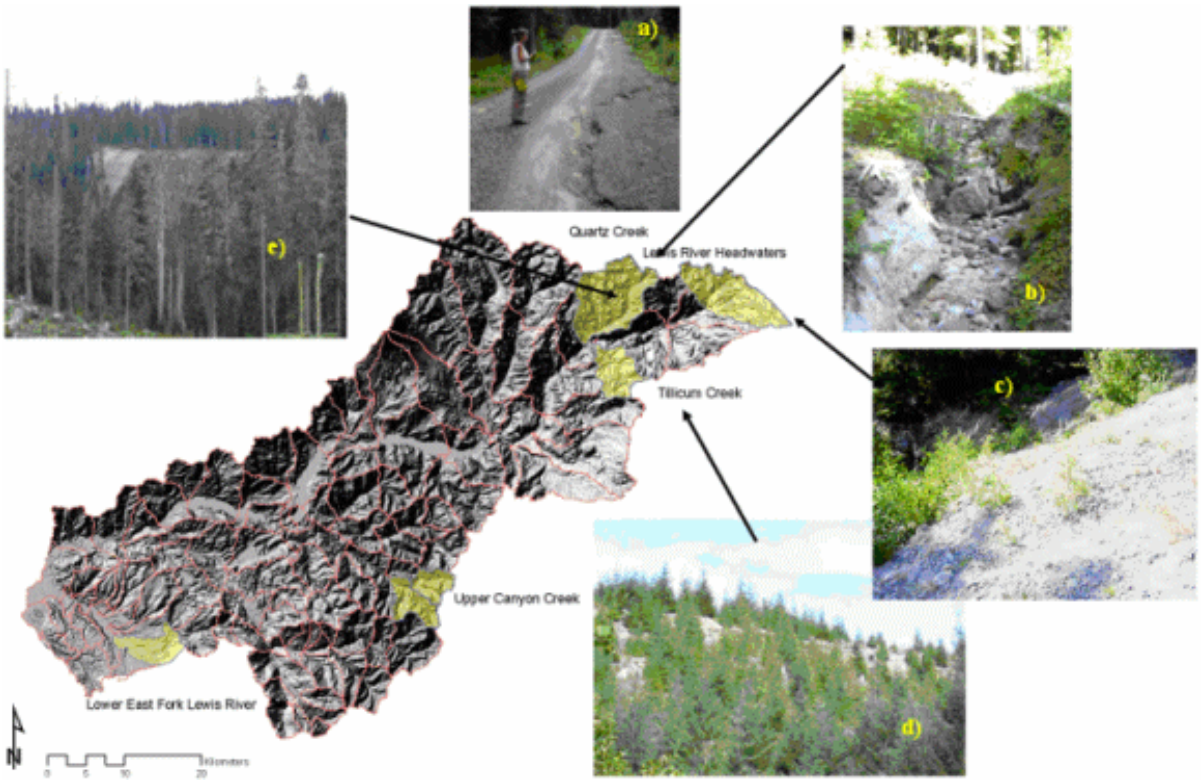

Figure E-1. Field verification watersheds are in yellow. Photo series show a) road erosion; b) runoff erosion in 20 year plantation; c) paved road prism erosion in ash-pumice area; d) continued surface erosion in 15+ year plantation in ash-pumice areas; e) surface slumping/erosion in clearcut in ash-pumice area.

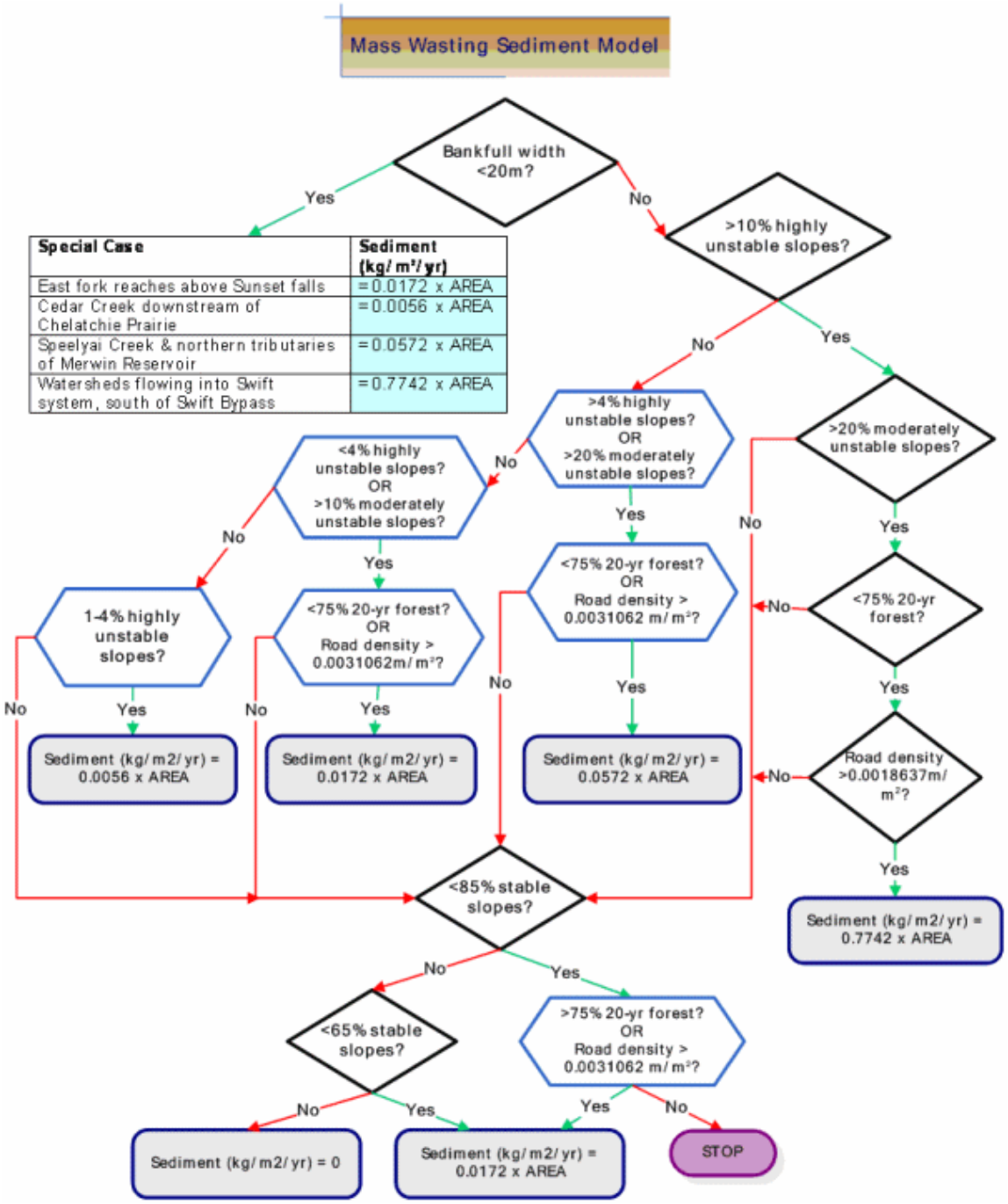

Figure E-2. The revised DSS mass wasting model after field verification incorporates changes in land cover and road density, and presence or absence of ash soils by increasing %slope values in the DNR mass wasting model. The DNR model provides the information on slope stability used in the DSS mass wasting model.

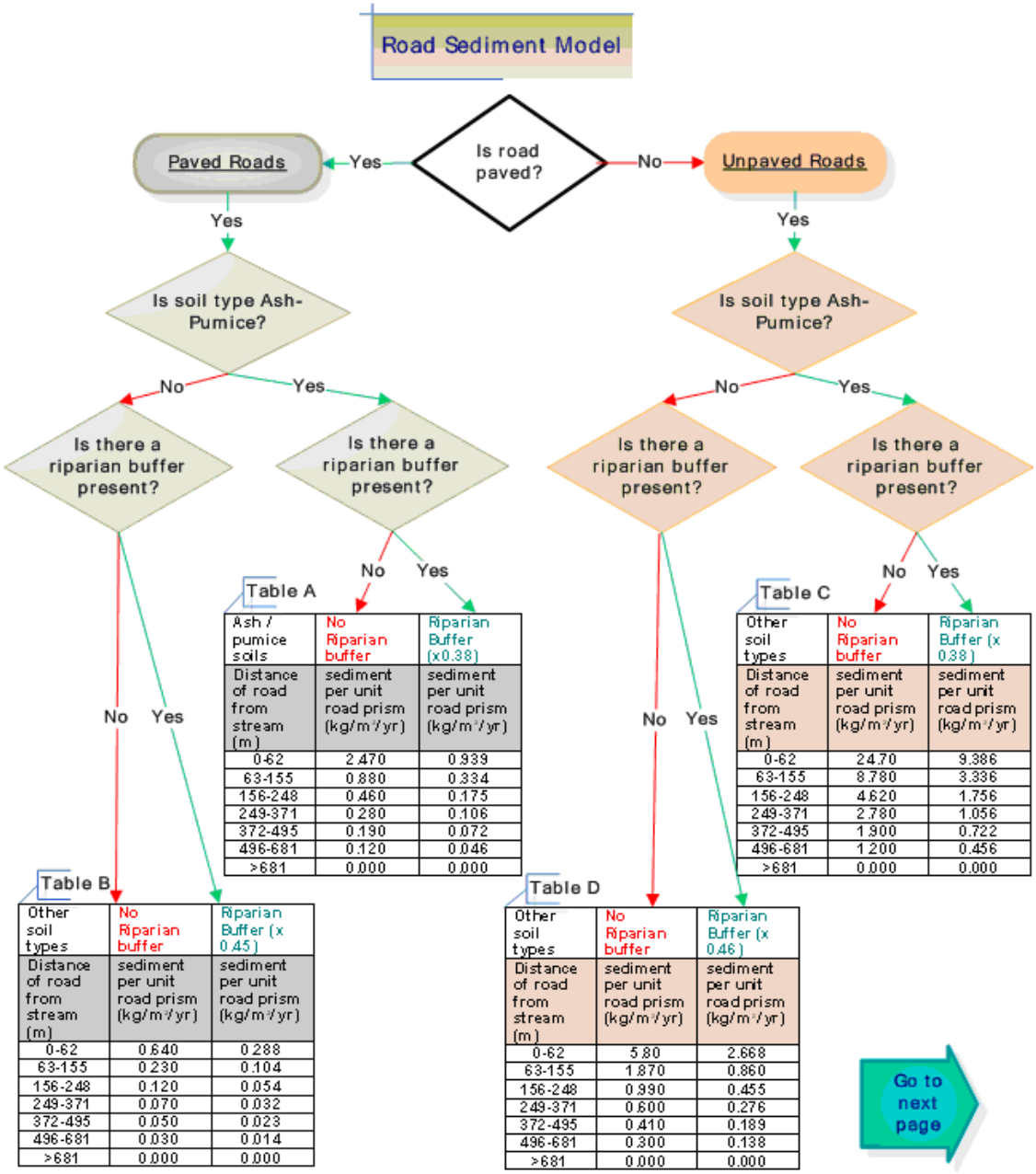

Table A

Ash / pumice soils	No Riparian buffer	Riparian Buffer (x0.38)
Distance of road from stream (m)	sediment per unit road prism (kg/m²/yr)	sediment per unit road prism (kg/m²/yr)
0-62	2.470	0.939
63-155	0.880	0.334
156-248	0.460	0.175
249-371	0.280	0.106
372-495	0.190	0.072
496-681	0.120	0.046
>681	0.000	0.000

Table B

Other soil types	No Riparian buffer	Riparian Buffer (x 0.45)
Distance of road from stream (m)	sediment per unit road prism (kg/m²/yr)	sediment per unit road prism (kg/m²/yr)
0-62	0.640	0.288
63-155	0.230	0.104
156-248	0.120	0.054
249-371	0.070	0.032
372-495	0.050	0.023
496-681	0.030	0.014
>681	0.000	0.000

Table C

Other soil types	No Riparian buffer	Riparian Buffer (x 0.38)
Distance of road from stream (m)	sediment per unit road prism (kg/m²/yr)	sediment per unit road prism (kg/m²/yr)
0-62	24.70	9.386
63-155	8.780	3.336
156-248	4.620	1.756
249-371	2.780	1.056
372-495	1.900	0.722
496-681	1.200	0.456
>681	0.000	0.000

Table D

Other soil types	No Riparian buffer	Riparian Buffer (x 0.46)
Distance of road from stream (m)	sediment per unit road prism (kg/m²/yr)	sediment per unit road prism (kg/m²/yr)
0-62	5.80	2.668
63-155	1.870	0.860
156-248	0.990	0.455
249-371	0.600	0.276
372-495	0.410	0.189
496-681	0.300	0.138
>681	0.000	0.000

Figure E-3. The revised Road Sediment model incorporates ash/pumice soils and distance from stream crossings. The model is applied for each road in the lateral drainage wings. The model decision tree is continued on the next page. The same model is used for runoff with the change in sediment table values to runoff table values.

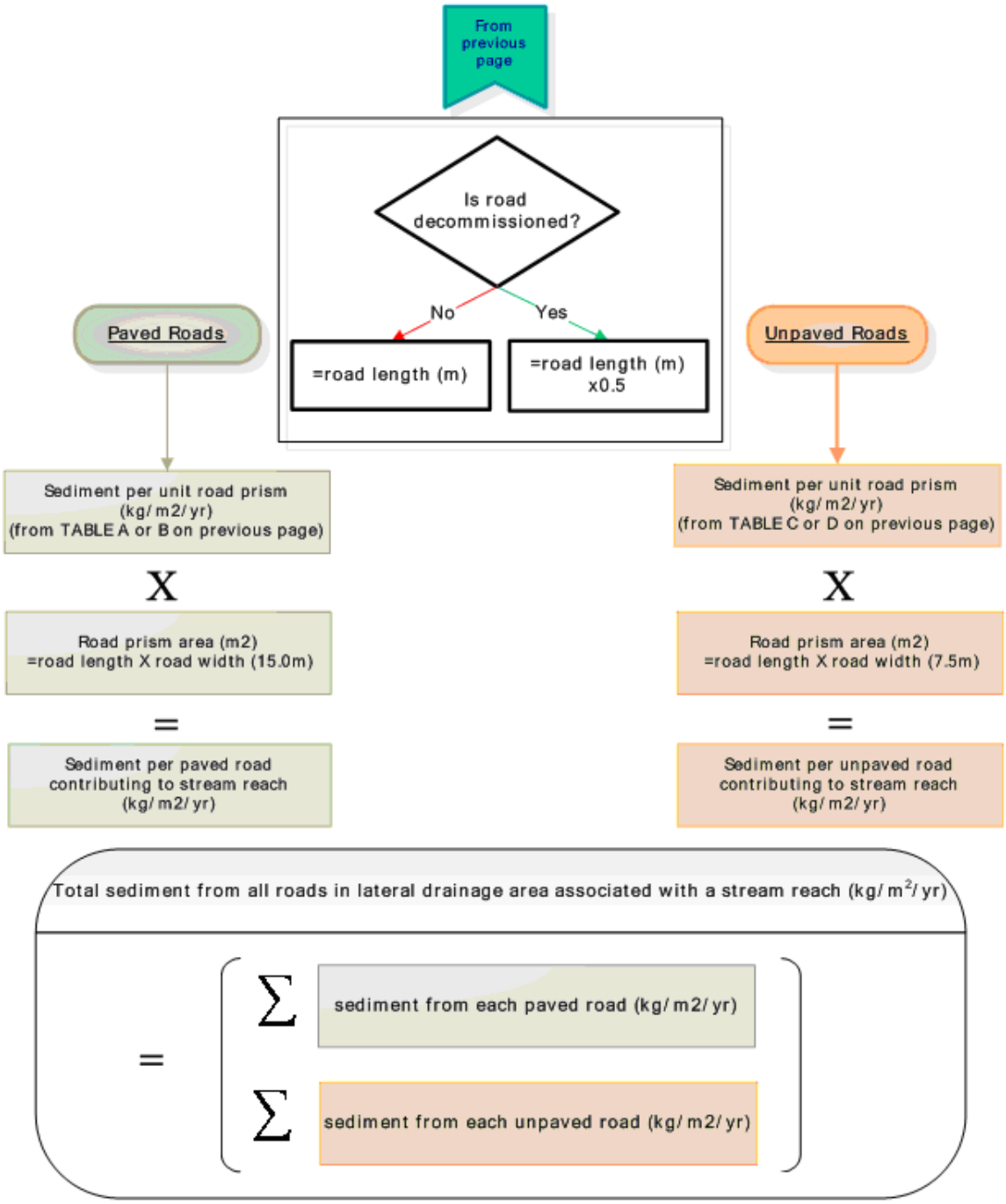

Figure E-3. (Continued): The revised Road Sediment model incorporates ash/pumice soils and distance from stream crossings. The model is applied for each road in the lateral drainage wings.

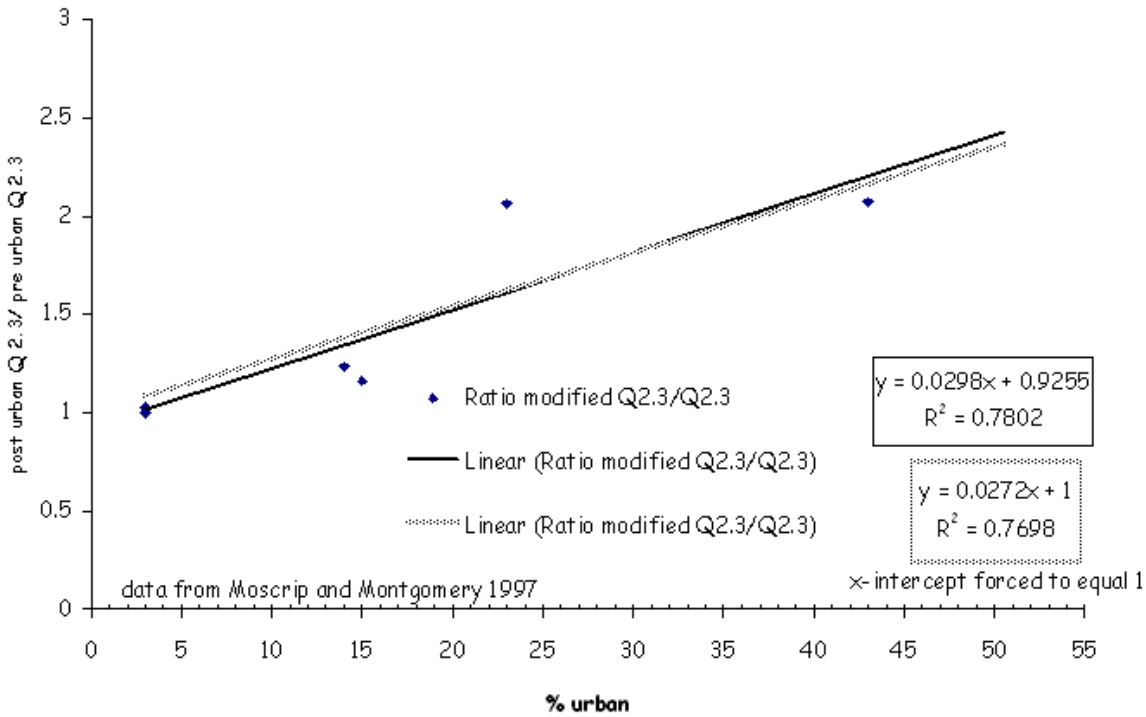

Figure E-4. Graph shows the relationship between the ratio of post urban 2.3-year flood to pre urban 2.3 year flood versus the percent area in urban. The regression equation, based solely on the data from Moscrip and Montgomery (1997), is the solid line. The dotted line and equation is the case where the x-intercept is forced to equal 1.

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Lewis River Case Study Final Report

A decision-support tool for assessing watershed-scale habitat recovery strategies for ESA-listed salmonids

Appendix F: Sediment Routing

May 2007

Introduction

The sediment routing model is used to route sediment delivered to each channel segment from the lateral hillslopes and upstream segments. The lateral and upstream sediment influx is cumulatively routed through each segment to the next downstream segment. The routed sediment estimates are output as habitat metrics and also used as inputs for evaluation models such as the sediment survival model (Appendix K).

Our goals are to identify the sources and transport pathways for sediment entering the river system. Ideally, we would also determine corresponding transport rates and sediment transit times, with associated channel morphologies, bed textures, and stability, for all points in the channel system.

The first goal is approachable: basin topography, together with geologic, soil type, and land-cover mapping, can be used to identify source locations and estimate sediment input rates to channels. Topography can also be used to delineate the channel network and determine flow paths for all points in the basin. The second goal is less approachable: controls on sediment transport are numerous and complex, far beyond our data sources to characterize completely. Nevertheless, we can generalize.

Using gross morphologic indicators of drainage area, channel gradient, and valley width, we can broadly delineate different channel types and identify potential zones of transport and deposition (e.g., Montgomery and Buffington 1997). When coupled with basin-specific empirical data, such generalizations can be used to predict channel conditions and responses to changes upstream.

Here, we describe the processes of sediment routing for the Lewis River basin. Many steps entail GIS exercises to relate disparate data sources and to extract needed information for the actual sediment routing. All GIS programming tasks were done in FORTRAN95. Microsoft Excel was used for plotting graphs and estimating functional relationships between variables.

Link Segments

The channel network is represented as a series of channel reaches, each reach is represented in a GIS database as a unique geo-referenced arc (referred to here as a segment, in reference to the SHIAPP (2004) terminology), and each segment has associated data that describe attributes of the reach. The SHIAPP segment data shapefile provided information concerning segment locations and attributes, but lacked information indicating how the segments are connected spatially, such as which segments were connected and the direction of flow between them.

All segments are identified with a unique integer value (ID) and segment topology is defined in terms of the up- and downstream segment IDs. In most cases, there is only one upstream segment and one downstream segment; although there are multiple upstream segments at channel confluences and multiple downstream segments where anastomosing channels diverge. Two steps were used to identify the up- and down-stream segments:

1) *Use segment-endpoint locations to identify contiguous segments*.
   Adjacent segment endpoints are coincident. I compared endpoints of all segments to find those that matched. There were no matches for ten reaches. Some of these were completely isolated from the channel network, others crossed another segment, but the endpoints did not match. I visually matched all but two of these.
2) *Use flow direction to determine the up and downstream segments.* Flow directions were determined with the program bld_grds (Miller 2002, 2003), part of the NetStream program set, which constructs a raster file specifying flow direction at each pixel, with pixels defined by the 10-m DEM.

These steps were done using short programs written in FORTRAN95. Up- and down-stream segment ID for each segment is stored in a binary data file, which can be written to an ASCII output file if required.

Delineate Contributing Areas

Sediment inputs from surface erosion, road erosion, and mass wasting are estimated per unit area as functions of surface gradient (and curvature for mass wasting), soil type, land use, forest cover, and road density. These models were developed separately and are described in Appendix E. Estimates of mean annual sediment influx to a segment, excluding inputs from upstream, are calculated for the unique contributing area to the segment, based on its gradients, soil types, land uses, forest cover, and road density. The contributing areas form closed polygons adjacent to the segment, typically one on each side, and are referred to here as “drainage wings.”

Drainage wings were delineated using the flow direction estimated for each DEM pixel. Using a new raster file with pixel values initialized to zero, pixels containing channel segments were flagged with the segment ID. All non-channel pixels were then given the ID of the segment they flow into. Drainage wings are thus delineated at a resolution of 100 m^2, the surface area of a 10-m DEM pixel. The resulting raster file was later used to define a polygon coverage of drainage wings to use for sediment and runoff modeling (Appendix E).

Using this algorithm, drainage wings for some very short segments were not delineated. These segments then have no unique contributing area, but still serve as conduits for water and sediment delivered from upstream. Although a small proportion of the segments ended up with no drainage wings, those that were delineated account for all surface area of the basin.

Cumulative routing of sediment inputs

Output from the erosion models is provided in a row-column format, ASCII data file that gives mean annual sediment inputs (kg) from each drainage wing. Sediment inputs for three sources are reported: surface erosion, mass wasting, and roads (including both surface erosion and mass wasting associated with roads). We further divided sediment inputs into six size classes (Table F-1).

Table F-1. Sediment size classes

Size Class	Size range (mm)	Description
1	<0.25	Fine sand, assumed to be transported as wash load
2	0.25-0.5	Medium sand
3	0.5-1.0	Coarse sand
4	1.0-4.8	Very coarse sand to fine gravel
5	4.8-78	Medium-coarse gravel
6	>78	Cobble

These inputs are summed cumulatively downstream through the channel network, so that the total upstream mean-annual sediment yield can be estimated for each of the sources and for each of the size classes. This information provides an estimate of mean annual sediment yield (in kg) from all upstream sources to each segment (with zero transport assumed through reservoirs).

Downstream fluvial transport results in abrasion of sediment clasts. Because much of the abraded material is small enough to be carried as wash load, bed load volume is systematically reduced with downstream travel. Estimates of mean annual wash load and bed load must therefore account both for inputs to the channel and for changes in volume (reductions to bed load, increases to wash load) associated with abrasion.

Volume loss to abrasion is difficult, perhaps impossible, to measure directly in a river, so tumbling-mill experiments are used to estimate abrasion rates. Volume loss of coarse grains is found to be proportional to the distance traveled:

$$dV = -\beta V dx \quad (1)$$

where V is volume of bed-load-sized material in the mill and x is distance traveled (product of mill circumference, rotations per unit time, and time the mill is run).

Abrasion causes both a loss of total bed load volume and a change in grain-size distribution.

For a given size class, some sediment clasts are lost to the next lower size class, as their size is reduced by abrasion, and some clasts are gained from the next larger size class. The change in volume *v* in a single size class during mill transport is estimated as (Parker 1991):

$$\frac{dv}{dx} = \frac{1}{3}\frac{\partial(v\beta)}{\partial \ln(D)} - v\beta \quad (2)$$

where ln(D) is the natural logarithm of the mean diameter of clasts in the size class. (Clast size is commonly characterized using a log scale, because of the typical abundance of small clasts). If we assume that abrasion in the mill can be used to approximate abrasion during fluvial transport, this equation can be integrated over travel distance x for all size classes to estimate both the change in the grain-size distribution and the loss of bed-load volume to wash load.

The rate of volume loss, specified by β, varies with rock type and abrasion process (Mikos 1995) and is estimated from repeated sieving of material after increments of tumbling using either total volume loss (Eqn. 1, assuming that β is independent of clast size), or from the evolution of the grain-size distribution (Eqn. 2). Tumbling-mill data (Figure F-1) from Tilton River sediment were used (unpublished data, Earth Systems Institute). The Tilton is a tributary of the Cowlitz River to the north of the Lewis River Basin. Both the Tilton and Lewis river basins are underlain by similar volcanic rock types. Using Equation 1, we obtain a value for β of 0.034 km^{-1}, i.e., a volume loss of 3.4% per kilometer of transport.

To apply Equation 2 to estimate changes in grain-size distribution associated with abrasion requires a finely subdivided set of size classes. Sediment volumes in the six size classes specified above were subdivided into smaller logarithmically distributed classes using the following procedure. The proportion of sediment volume in size class *i* was used to estimate a probability:

$$p_i = (v_i/\Sigma v)/(\ln(D_i)-\ln(D_{i-1})) \quad (3)$$

where the sum over all size classes, Σv, gives total sediment volume and $(\ln(D_i)-\ln(D_{i-1}))$ gives the width of the ith class. A cubic spline was fit to these values to estimate a probability density function over the range of grain sizes. The spline was constrained so that integrated value over a size class equals the value given by Eqn. 3. The spline was then integrated over smaller increments to provide a larger number of grain-size classes for numerical integration of Eqn. 2.

For a channel segment, sediment flows in via fluvial transport from the upstream segment and erosion from adjacent hillslopes. Changes in the grain size distribution and losses to wash load (size-class 1, Table F-1) by abrasion are estimated through integration of Eqn. 2 over the length of the segment. This gives a total volume (or weight) by size class, for each segment.

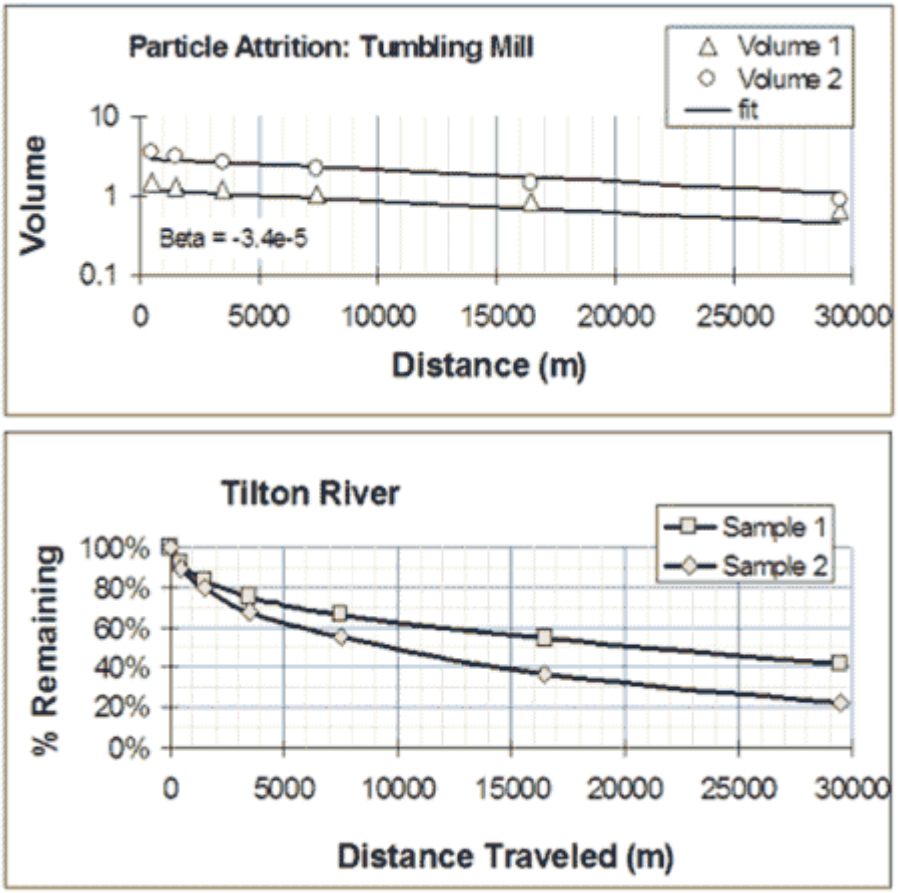

Figure F-1. Results of tumbling mill experiments for colluvium from the Tilton River basin (Earth Systems Institute, unpublished data).

Flood discharge

We use the estimated 2.3-year recurrence-interval flood discharge ($Q_{2.3}$) as an indicator of the channel-forming and bank-full flow (Wolman and Miller 1960, Whiting et al. 1999). The $Q_{2.3}$ was used rather than the $Q_{1.5}$ or Q_2 because cross-section measurements in the watershed indicated that effective discharge ranged from $Q_{1.5}$-Q_5 with $Q_{2.3}$ discharge the median. The $Q_{2.3}$ regression equations described for Sediment and Runoff (Appendix E) are a power function of drainage area:

$Q_{2.3} = 0.895317A^{0.925}$ (from gages above Merwin reservoir)

$Q_{2.3} = 0.967492A^{0.9132}$ (for the upper East Fork Lewis River)

Here $Q_{2.3}$ is in m^3/sec and drainage area A is in km^2.

These estimates are altered by land use in the watershed. Timber harvest or conversion of land from forest to agriculture, roads, or urban areas results in a proportional increase in $Q_{2.3}$, based on models described in Appendix E on sediment and runoff modeling. The proportional increase is calculated individually for each drainage wing, and then cumulatively summed over all upstream drainage wings to calculate land-use modifications to $Q_{2.3}$ for each segment.

Proportional increases in $Q_{2.3}$ for timber harvest and agriculture use are listed in Table F-1. Increases associated with agriculture vary with soil type, of which four broad types are delineated in the basin (described in Appendix A and Appendix E).

Table F-1. Increases in the estimated 2.3-year recurrence-interval flood discharge ($Q_{2.3}$) by land use and soil type

Land Use	Proportional increase in $Q_{2.3}$
Timber Harvest	0.08
Agriculture:	
Soil type 1	0.36
Soil type 2	0.12
Soil type 3	1.35
Soil type 4	0

The proportional increase in $Q_{2.3}$ associated with agriculture is calculated as $m_{ag} = \Sigma P_i m_i$, where P_i is the proportion of the area in agriculture in soil type i, m_i is the coefficient specified in Table F-1, and i varies from 1 to 4 (note that $\Sigma P_i = 1$).

Increases associated with roads are based on road density: if road density (ρ_{road}) exceeds 0.0012 m/m^2, $Q_{2.3}$ increases by 20%: $m_{road} = 0.0$, if $\rho_{road} \leq 0.0012$; $m_{road} = 0.2$, if $\rho_{road} > 0.0012$.

Increases associated with urban areas are calculated as $m_{urban} = P_{urban}*3.0 + 0.93$. Here P_{urban} is the proportion of area in urban cover types and m_{urban} is the proportional increase by which to modify $Q_{2.3}$. This equation gives a value less than one for values P_{urban} less than 0.023, in which case m_{urban} is set to one.

For each drainage wing, a modifier (DW_{mod}) for $Q_{2.3}$ is calculated as:

$$DW_{mod} = P_{Forest}*m_{forest} + P_{Ag}*m_{ag} + P_{urban}*m_{urban} + m_{road},$$

where P_{Forest} is the proportion of area in forest (and harvest) land cover, P_{ag} is the proportion of area in agriculture, and P_{urban} is the proportion of area in urban areas. Note that P_{Forest}, P_{ag}, and P_{urban} sum to one.

To estimate the increase in $Q_{2.3}$ for a segment, we use an area-weighted sum over all upstream drainage wings:

$$Q_{2.3} = Q_{2.3Base}\left(\frac{\sum\left(DW_{\mathrm{mod}(i)}A_{DW(i)}\right)}{\sum A_{DW(i)}}\right) \quad (4)$$

Here $Q_{2.3Base}$ is the baseline (unadjusted) discharge, $A_{DW(i)}$ is the area of the ith upstream drainage wing, and $DW_{mod(i)}$ is the modifier for the ith upstream drainage wing.

Bed Texture

Channel sediment texture is important for at least two reasons: 1) fry survival is affected by the proportion of fine sediment in the bed (Bjornn and Reiser 1991), and 2) the extent and depth of channel scour correlates with bedload transport rates, which are reflected in bed-sediment texture (Dietrich et al. 1989, Montgomery et al. 1996, Haschenburger 1999); increased transport rates correspond to finer surface grain sizes.

A simple empirical approach was used to characterize channel-bed sediments. First, we identified the physical factors that might affect sediment supply, transport, and deposition (restricted to those factors that can be quantified with available data). Then, we sought correlations between these factors and the bed textures (e.g., percentage of fine sediment) reported in available channel surveys.

Fine sediment is here defined as that large enough to be carried as bed load, but small enough to fill interstices of larger clasts and thereby impede water flow through the channel bed or hinder fry emergence. For the size classes listed in Table F-1, class 1 (< 0.25mm) is considered small enough to travel as wash load during high flows and classes 2-6 (> 0.25) are considered the bed load, with classes 2-3 (0.25 – 1.0mm) comprising the "fines" portion of the bed load.

Two sets of geo-referenced channel-survey data were used as general indicators of bed texture and abundance of fine sediments in channel substrates. One provided reach-averaged information for 16,133 m of channel over 12 reaches (HabRate Surveys conducted during field verification for this project, Table F-1). Two sets of these reaches were adjacent, so the survey provided information from 10 sites in the Lewis watershed. The other data were for a continuous habitat survey along 27,782 m of Cedar Creek (Jennifer Stone, USFWS, personal communication), a tributary to the North Fork below Merwin Dam used heavily by spawning steelhead. Data from both sources were aggregated to average values over reaches corresponding to the SHIAPP segments. Both surveys contained estimates of % fines (sand), % gravel, % cobbles, and % boulders for channel bed surfaces. The Cedar Creek surveys also included % bedrock. For the Cedar Creek dataset, Wolman pebble counts were summarized into percent substrate classes for each reach. Percent fines were calculated as the percent of total substrate measurements <6mm. For the HabRate dataset, % fines were assessed visually as the proportion of the substrate that was silt and organic matter, rather than from substrate size measurements.

A variety of field studies point to numerous factors that affect bed texture and % fines (Lisle 1989, Montgomery and Buffington 1997). The resolution and types of data available for characterizing channel attributes limit us, however, to a fairly small subset of these factors. The erosion models provide an indication of the locations and relative average rates of sediment inputs to the channel system; topographic information that can be inferred from the 10-m DEM provides an indication of gross channel morphology in terms of channel size, gradient, and valley confinement.

Sediment inputs, upstream sediment yield, and the 2.3-year-recurrence-interval flood discharge were estimated as described in Appendix E. Estimates of channel gradient, drainage area, and valley width were obtained with the NetStream program set (Miller 2003). Channel widths were provided from a separate calibrated model. While there was a regression equation of mean bank-full depth to drainage area based on channel geometry at Lewis basin gaging stations, we were interested in a depth estimate that responded to other attributes as well. Discharge through a cross section is

$$Q = WHU \qquad (5)$$

where

W is channel width

H is mean channel depth (cross-section area divided by width)

U is mean flow velocity through the cross section.

Mean flow velocity can be estimated with Manning's equation:

$$U = \frac{S^{1/2} R^{2/3}}{n} \qquad (6)$$

where

S is channel gradient

R is the hydraulic radius (cross section area divided by wetted perimeter: ~WH/(W+2H)), equal approximately to mean channel depth H

n is Manning's roughness coefficient.

Using the 2.3-year recurrence-interval discharge ($Q_{2.3}$) and an estimate of channel roughness, we can solve Eqn. 5 for mean flow depth H. In general, channel roughness is found to vary systematically with other channel attributes. Small, steep, coarse-bedded channels tend to have high roughness coefficients; large, low-gradient, fine-bedded channels tend to have low roughness coefficients. An empirical equation for mountain rivers that relates Manning's n to channel gradient and hydraulic radius was used (Jarrett 1984, Wohl 2000):

$$n = 0.32\, S^{0.38}\, R^{-0.16} \qquad (7)$$

Eqns. 5, 6, and 7 give H as

$$H = \frac{Q_{2.33}}{WU} = \frac{Q_{2.33}(0.32 S^{0.38} R^{-0.16})}{W S^{1/2} R^{2/3}} \qquad (8)$$

Eqn. 8 can be solved iteratively (R is also a function of H) to provide a flow-depth estimate for every segment. This estimate is a function of channel width, channel gradient, and discharge (which is itself a function of drainage area and upstream land use).

If we assume equilibrium between estimated sediment inputs and fluvial transport rates, we can use a simple bed-load transport model to make further inferences about bed texture. Empirical measures of bed-load transport are commonly characterized in the form

$$q_s = a(\tau - \tau_c)^b \quad (9)$$

where

q_s is bed-load transport rate per unit channel width

τ is available shear stress at the channel bed

τ_c is the "critical" shear stress required to initiate transport

a and *b* are empirical coefficients

The Meyer-Peter and Müller equation, for example, gives a value for *a* of approximately $8\rho/(g(\rho-\rho_s)\rho^{3/2})$, where ρ is water density, ρ_s is sediment density, and g is gravitational acceleration, and a value for *b* of 3/2. The critical shear stress is a function of bed texture, and is estimated from the empirical Shields stress:

$$\tau^* = \frac{\tau_c}{g(\rho - \rho_s)D} \quad (10)$$

where D is a characteristic grain size for channel-bed sediments. From 10,

$$\tau_c = \tau^* g (\rho - \rho_s) D. \quad (11)$$

Equations 10 and 11 give

$$D = \frac{\tau - \left(\frac{q_s}{a}\right)^{1/b}}{g(\rho - \rho_s)\tau^*} \quad (12)$$

Table F-1. Reach-averaged results of HabRate surveys conducted in the Lewis River watershed.

Stream	Length (m)	Width (m)	Area (m2)	Slope (%)	Depth (m)	% Silt/Org	% Sand	% Gravel	% Cobble	% Boulder	% Bedrock
Clear Creek	1397.0	8.82	242.5	1.17	0.79	8.28	22.21	35.33	32.22	2.05	0.00
Copper Creek	2031.4	7.37	208.9	4.21	0.62	0.35	3.41	19.47	25.36	24.45	27.05
EF Lewis River	1363.2	19.07	2104.6	0.85	0.78	1.93	9.93	19.93	42.73	25.47	0.00
Johnson Creek 1	848.2	4.09	88.4	7.37	0.41	2.40	16.38	32.20	21.03	5.28	22.63
Johnson Creek 2	908.7	4.24	72.5	1.89	0.29	3.3	15.77	34.35	33.35	12.77	0.44
Mason Creek	1318.63	4.36	233.9	0.48	0.6	40.21	8.46	33.68	17.11	0.46	0.00
Miller Creek	1086.5	2.67	31.6	10.94	0.29	2.55	12.58	13.94	25.25	44.58	1.07
NF Lewis River 1	1547.4	19.19	1826.6	0.87	1.42	0.00	9.60	15.70	26.45	27.40	20.80
NF Lewis River 2	2606.8	22.68	3169.2	0.50	0.83	0.00	16.92	25.00	34.72	21.20	1.72
Pine Creek	3399.7	8.37	376.5	4.10	0.58	0.05	19.24	20.79	28.04	31.73	0.13
Rock Creek	1413.87	10.39	513.2	0.86	0.58	3.44	9.94	33.59	29.53	10.84	12.78
Siouxon Creek	1295.20	15.89	800.71	3.01	0.96	0.85	6.19	15.52	19.78	38.59	18.81

For gravel-bed channels, τ^* is nearly constant in the range of 0.03 to 0.045 (Buffington and Montgomery 1997). Using a value for τ^* of 0.03 (Buffington and Montgomery 1999a), values for *a* and *b* from the Meyer-Peter and Müller equation, and the depth-slope product ($\tau = \rho gSH$) for shear stress, Eqn. 12 provides an estimate of the characteristic grain size associated with a transport rate q_s.

If we assume that most sediment transport occurs during bankfull discharge, approximated in the Lewis basin with $Q_{2.3}$, transport rate q_s is approximated as

$$q_s \approx Q_S / (W*T)$$

where

Q_S is total sediment input from upstream, estimated as mean annual sediment yield

W is channel width

T is the duration of bankfull flow, estimated as 2.1 days, the average for Lewis River basin streams.

The assumption of equilibrium between sediment inputs and outputs, so that q_s can be estimated from mean annual sediment yield, is a purely conceptual construct. Sediment production and transport is highly stochastic (e.g., Benda and Dunne 1997) and transit times through valley segments can vary dramatically (Reid and Dunne 1996). Likewise, sediment transport may occur over a range of discharges. Indeed, with these assumptions, Eqn. 12 specifies a negative characteristic grain size for many segments. Nevertheless, this calculation serves a useful purpose. It provides a spatially distributed estimate of potential supply relative to potential transport. Small values of D correspond to segments where supply is large relative to transport potential (transport-limited, response reaches); large values correspond to segments where supply is low relative to transport potential (supply-limited, transport reaches). The value of D obtained with Eqn. 12 responds both to changes in $Q_{2.3}$, via changes in the estimated flow depth used to calculate shear stress τ, and to changes in estimated sediment supply. Increased discharge results in an increase in D; increased supply causes a decrease in D.

We can use the same approach to evaluate the potential effect of local sediment supply on bed texture. Rather than sediment yield for the entire upstream watershed, q_s can be based on some measure of local sediment input. I defined a local measure of sediment flux using a distance-weighted sum of upstream sediment inputs, with a weighting that decreases exonentially with distance upstream:

$$Q_{Slocal} = \Sigma(Q_{Si}\, e^{-\alpha x_i}) \quad (13)$$

Where

Q_{Slocal} is a measure to characterize total sediment flux through the channel from local upstream sources

Q_{Si} is sediment input from the ith upstream drainage wing

x_i is distance to the ith upstream segment.

The coefficient α is set to cause the weighting term to decrease by one half over a specified distance $x_{1/2}$, i.e., $\alpha = \ln(0.5)/x_{1/2}$. We found a value of $x_{1/2}$ = 500m worked well for correlating observed channel textures to local sediment inputs. Together, these methods provide channel attributes such as proportion of fines (Table F-2).

Table F-2. Estimated channel attributes

Attribute	Source
Gradient (S)	DEM (NetStream, using intersection of channel arcs and 40-ft contour lines)
2.3-yr recurrence interval flood ($Q_{2.3}$)	Function of drainage area, modified by watershed land use
Mean annual upstream bed load (Q_S)	Erosion models, abrasion
Stream power (Q_S)	Product of $Q_{2.3}$ and gradient S
Local sediment inputs (Q_{Slocal})	Erosion models, distance upstream
Proportion of bed load sediment in fine size class (P_F for total watershed yield and P_{Flocal} for locally derived sediment)	Based on grain size distribution for incoming fluvial sediment and local upstream drainage wings
Channel width (W)	Calibrated function of drainage area, mean annual precipitation, and channel gradient
Mean bank-full depth (H)	Function of $Q_{2.3}$, channel gradient, and channel width
Valley width	DEM (NetStream)
Maximum reach-averaged shear stress (τ)	Calculated from gradient and flow depth (ρgHS)
Valley-width index (VWI)	Ratio of valley width to channel width
dS/dx, dW/dx, dVWI/dx	Longitudinal changes in channel gradient, channel width, and valley width
Characteristic bed grain size based on transport rate (D)	Function of sediment supply and maximum reach-averaged shear stress

Proportion Fine Sediment

Correlations between channel attributes (Table F-2) and the % fines reported in the HabRate and Cedar Creek surveys (noting that many of the attributes are themselves correlated) were evaluated. The HabRate data exhibited relationships with three attributes (Figure F-2), listed in order of decreasing importance:

- P_{Flocal}, the proportion of local sediment inputs (using the distance-weighted sum defined above) in fine size classes (classes 2 and 3 in Table F-1, 0.25-1.0mm),
- VWI, the valley width index, and
- Q_S, stream power.

No significant correlations were found for any of the other variables. These observations led us to infer that the % fines reported in the HabRate survey depends primarily on the grain-size distribution of local upstream sediment inputs (as represented by the erosion

models, which respond to soil types, hillslope gradients, and land cover), modulated by channel characteristics represented with the valley-width index and estimated stream power:

$$\%Fines = a_1 P_{Flocal}^{b1} (1 + (a_2 + c_2 VWI^{b2}) + (a_3 + c_3 QS^{b3})) \qquad (14)$$

$$= a_1' P_{Flocal}^{b1} (1 + a2' VWI^{b2} + a_3' QS^{b3})$$

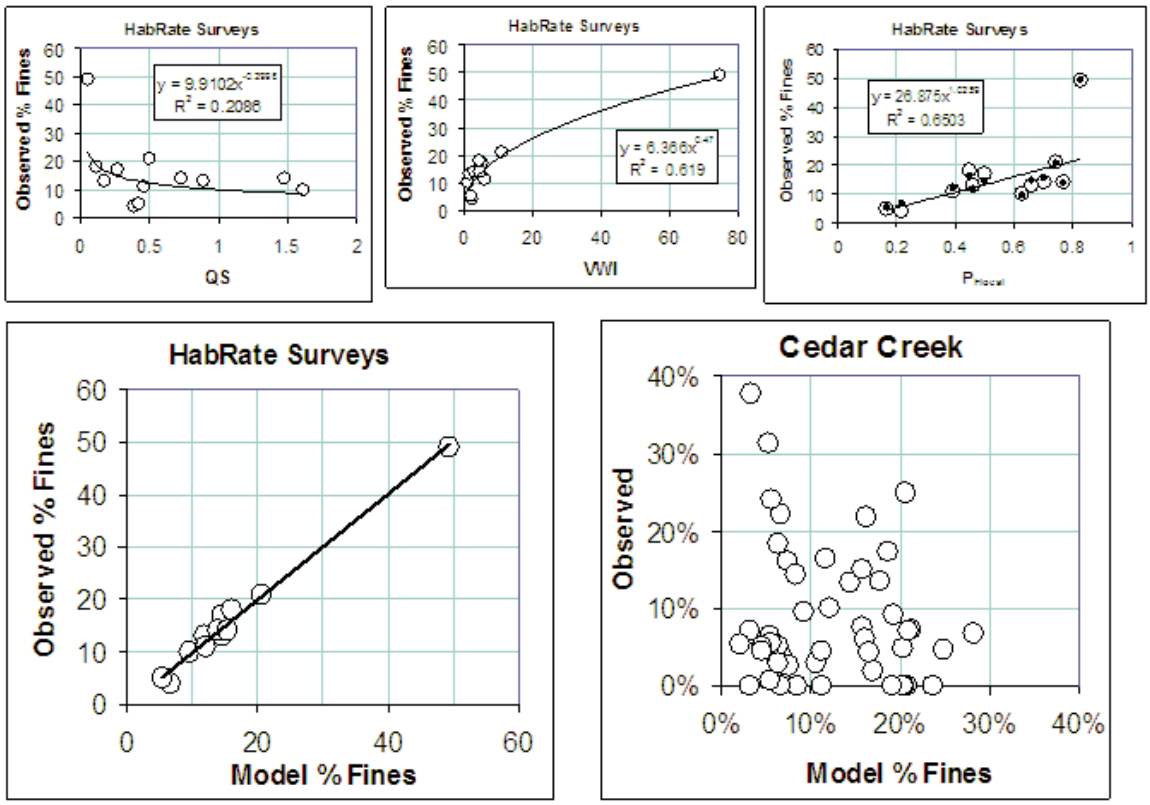

Figure F-2. Upper graphs show relationships between stream power (QS), valley-width index (VWI), and the proportion of local sediment inputs in fine (0.25-1.0mm) size classes (PFlocal). The black lines show power-function regressions to the data. The small black dots in the right-most graph show adjustments in estimated values when effects of stream power and channel confinement (via VWI) are included in the model. Lower graphs show correlation between modeled and observed values. The model was calibrated to the HabRate surveys. The Cedar Creek values showed no relationship to any geomorphic attributes (Table F-2) and the HabRate model bears no relationship to observed values at Cedar Creek.

Coefficients were obtained using the "Solver" module in Microsoft Excel to minimize RMS (root-mean square) differences between the %Fines calculated with Eqn. 14 and reported values. Power functions are used so that the influence of large values can be reduced (e.g., the influence of valley width on channel confinement changes little beyond a valley width index of 10 or so, as indicated by the small exponent given by value b2 in Table F-3). Although this model fits the observations well, it is based on few samples, has half as many parameters as data points, and is heavily influenced by one large data value. The conceptual model for Eqn. 14 and the relative magnitude of the fit parameters do correspond to physical expectations, but our ability to predict observations elsewhere in the basin may be very limited. While analyses of significance for these coefficients were not done, we can evaluate the predictive ability of the model using the HabRate and

Cedar Creek data. As shown in Figure F-2, the HabRate model has a positive relationship but the Cedar Creek model has no relationship to the observed %sand.

This failure is not necessarily an indication that the model has no worth; it may simply be a consequence of the great potential heterogeneity exhibited by processes of sediment supply and transport (Benda et al. 1998) as reflected in resulting substrate texture. It might also indicate differences in survey techniques. The Cedar Creek survey, although of longer total length than the HabRate surveys, is limited to a single channel, whereas the HabRate data reflect ten different locations in the watershed. These ten sites may provide a better indication of channel response than obtained with data from a single channel. Moreover, the value representing % fines in the Cedar Creek dataset included sediment particles of up to 6mm, which is likely larger than that considered to be "silt" in the HabRate dataset. In the absence of additional data, Eqn. 14 provides a model for estimating the proportion of fines expected in channel-bed sediments. It is responsive both to changes in local sediment supply and to changes in discharge, which serves the need for simulating relative effects of different land use scenarios in the basin.

Table F-3. % Fines model coefficients (pre-test model)

Coefficient	Value
a_1'	0.231
b1	0.610
a_2'	47.600
b2	0.292
a3'	5.200
b3	-0.928

Test and Revision of the % Fines Model

Initial scenarios to examine the effects of road decommissioning resulted in increased percent fines in some channel reaches. The model for percent fines is based on correlations with three independent variables: the proportion of locally derived sediment in fine size classes, channel width to valley width (valley width index), and an estimate of stream power (the product of the modeled 2.33-year-recurrence interval flood discharge and DEM-estimated channel gradient). Roads enter the sediment models in two ways: there is erosion from road surfaces and road density enters calculations of mass wasting. Removing road segments therefore reduces both surface erosion and mass wasting sediment sources, which each contribute different sediment grain-size distributions.

For most channel segments, surface erosion (excluding roads) contributes the majority of all sediment, and this sediment is predominately fine grained. Coarse-grained sediment comes primarily from mass wasting. Eliminating the mass-wasting sediment associated with roads reduces the total sediment input, but by eliminating a source of coarse sediment, can increase the proportion of the sediment supplied that is in fine size classes. Since this proportion is the factor that influences the predicted percentage of fines in the bed, and not the total amount of sediment supplied, the result is an increase in the predicted percentage of fines in the channel bed.

Two potential problems with the modeling methods were identified. First, surface erosion, particularly from roads, occurs with every rainstorm, whereas landsliding is

considerably less frequent. Within any single drainage wing, most years probably have no landslides. Our models utilize estimated long-term mean annual sediment inputs, but what one finds in the channel bed might depend more on what has occurred in the last few years. We may capture average conditions better using only the chronic sediment sources – surface erosion and roads – and not using mass wasting inputs for predicting the percentage of fines in the channel bed. Second, surface erosion of road surfaces predominately produces sediment that is fine enough to be suspended by overland flow over the road, generally less than 1.0 mm. We have been using the grain-size distribution associated with surface erosion, which although primarily of finer size classes, includes a proportion of coarse sediment.

To address the first point, mass-wasting sediment was excluded from the analysis. The model was recalibrated using only sediment from surface and road-erosion sources. The resulting coefficients were very similar and the goodness of fit unchanged. However, because surface erosion and road erosion both contribute the same size-class distribution within our models, this change would not result in any reduction in the percentage of fine sediment in channel beds associated with road removal

To address the second point, we altered the grain-size distribution for sediment derived from roads to exclude size classes greater than 1.0 mm. The incoming sediment was divided between the < 0.25, 0.25-0.5, and 0.5-1.0 mm size classes in the proportions specified for these three classes for surface erosion. This time, recalibration of the model resulted in a very slightly better fit.

These changes give us a model with coefficients that fit our limited observations just as well as the previous set, but that should result in a reduction in predicted percent fines in the channel bed when roads are decommissioned (Table F-4). This results because 1) roads now contribute only fine sediment, and 2) changes in mass wasting associated with removing (or adding) roads do not affect the predicted percentage of fines in the bed.

The revised model (Equation 14):

$$\%Fines = a_1 P_{Flocal}^{b1} (1 + (a_2 + c_2 VWI^{b2}) + (a_3 + c_3 QS^{b3}))$$

$$= a_1' P_{Flocal}^{b1} (1 + a2' VWI^{b2} + a_3' QS^{b3})$$

Table F-4. Updated % fines model coefficients:

Coefficient	Value
a_1'	0.207
b1	0.860
a_2'	54.093
b2	0.342
a3'	4.955
b3	-0.493

Proportion Coarse Sediment

Bed-surface texture can provide an indication of bed load transport rate, with finer textures indicative of larger transport (Dietrich et al. 1989, Buffington and Montgomery 1999). Texture is also influenced by local inputs (e.g., the boulders found in debris-flow deposition zones). Transport rate and local sediment inputs are important determinants

for many channel attributes (Church 2002), including depth of scour. As described above, we can use modeled sediment inputs to the channel system to estimate transport rates (assuming equilibrium with upstream sediment yield) and local sediment inputs. We can use the sediment models, together with other estimated channel attributes that affect transport capacity (channel gradient, channel width, and flow depth), to predict variations in bed texture, in terms of a characteristic grain size, throughout the channel network (Eqn. 12). This hypothesis can be tested against the HabRate and Cedar Creek survey data.

The D value was calculated for all segments using both total sediment yield and local sediment inputs, as described previously (Eqn. 12). Using the sum of % cobbles and % boulders (and % bedrock for Cedar Creek) as an indicator of bed texture (i.e., % coarse), comparison with surveyed values are shown in Figure F-3. Using total sediment yield to estimate transport rate in Eqn. 12 provides no relationship between the calculated grain size D and observed bed textures. However, characteristic grain size based on local sediment supply correlates with observed textures in both cases. There is a large degree of scatter, but a clear trend exists. For the HabRate survey, correlations are also found with the proportion of locally supplied sediment in size classes 4 and 5 (>4.8mm, Table F-1) and with valley width index. Such correlations are evident, but much weaker for the Cedar Creek data.

These results suggest that estimates of shear stress and local sediment supply (Eqn. 12) provide a useful indicator of bed texture. This model is responsive to changes in flow, which by altering estimated flow depth (and associated shear stress), indicates a coarsening of bed texture with increased discharge, and changes in local sediment supply, with increased supply indicating a fining of bed texture. Although this model ignores other types of channel response, such as changes in width or gradient, it provides an indication of the magnitude of response that may be associated with changes in the watershed that affect rates of runoff and erosion.

Using the HabRate data, we can incorporate predicted characteristic grain size (Eqn. 12), the proportion of locally supplied sediment in the fine size fraction (0.25 – 4.8mm), and the valley width index into a calibrated model to predict the coarse fraction of channel bed sediments. Following the reasoning for Eqn. 14, we define a model:

$$\text{Coarse_Fraction} = a_1 D^b (1 + a_2 P_{Flocal} + a_3 VWI) \qquad (15)$$

Here P_{Flocal} is the proportion of local sediment inputs ("local" as defined previously, using an exponential weighting with distance) in size classes 1-3 (0.25-4.8 mm, Table F-1). As before, coefficients are estimated using Excel Solver to minimize the RMS error. Residuals indicated no nonlinear dependence on P_{Flocal} or VWI (with VWI cut off at a value of 10, i.e., MIN(VWI,10)), so no exponents were used with these variables. Coefficient values are supplied in Table F-5.

Table F-5. Coarse-fraction coefficient values.

Coefficient	Value
a_1	108.800
b	0.143
a_2	-0.192
a_3	-0.0524

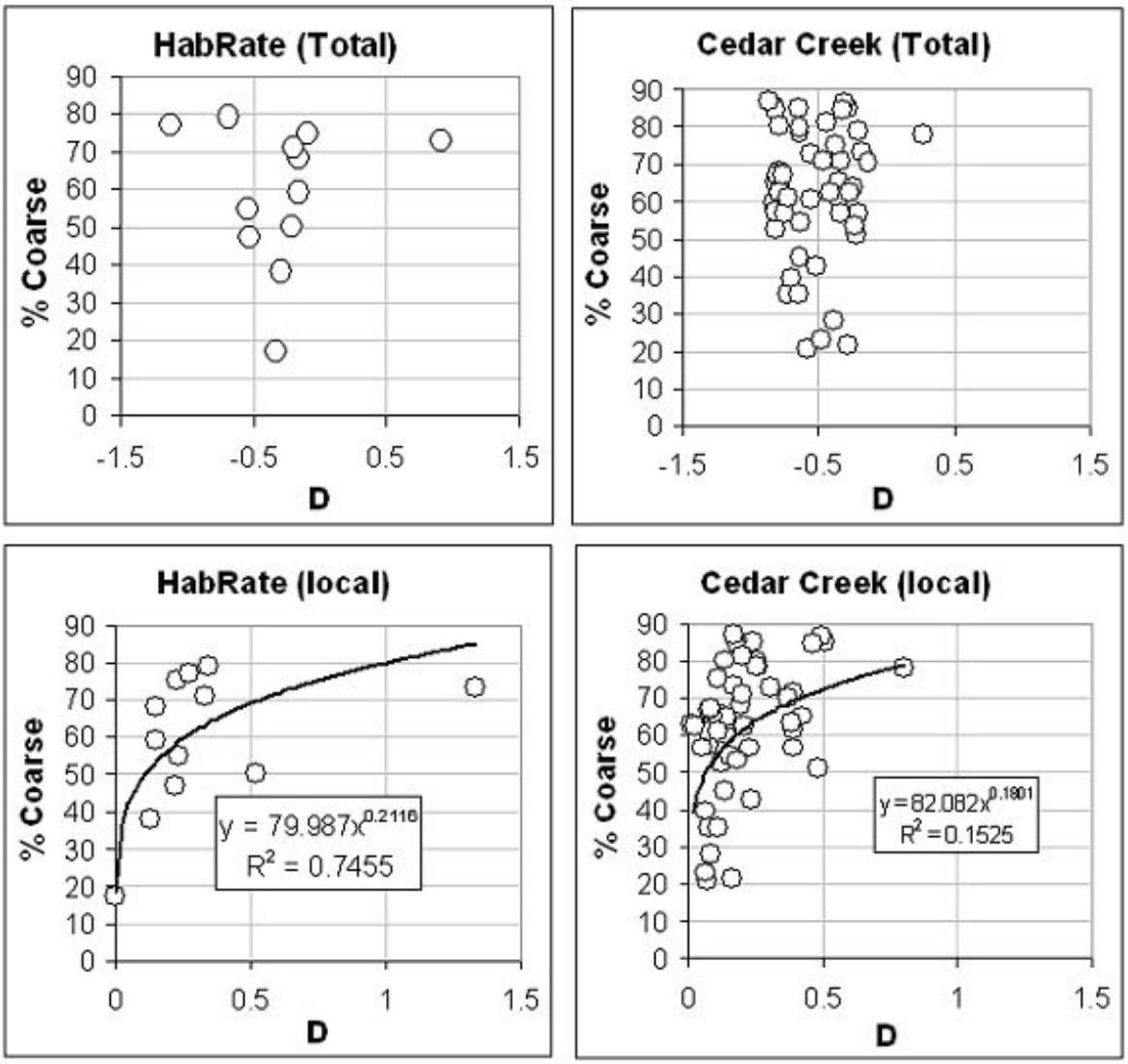

Figure F-3: Characteristic bed grain size based on bed load transport rate (Eqn. 12). Use of total watershed yield shows no relationship (upper graphs); use of local sediment inputs to estimate transport rate shows consistent trends for both data sets (lower graphs).

Bed Scour

Over reach-averaged length scales (tens to hundreds of meters), the depth, frequency, and spatial extent of scour is observed (or expected) to vary with numerous factors, including bedload transport rates and associated shear stress (channel depth and gradient) and bed texture (Montgomery et al. 1996, Haschenburger 1998), channel confinement (Schuett-Hames et al. 1994), flood discharge (Haschenburger 1998), and the amount of stable large woody debris. These factors are themselves inter-related.

The channel attributes and models for characteristic grain size described above incorporate the first three of these factors, bed load transport rate, channel confinement, and flood discharge. We therefore have the ingredients for a model of relative bed scour.

Following is a list of factors available to us that we might include in a model of scour:

- Flood discharge $Q_{2.3}$ (increased discharge causes increase in mean scour depth)
- Shear stress $\tau \approx \rho hHS$ (increased shear stress causes increase in mean scour depth)
- Characteristic grain size D (function of shear stress and sediment supply, smaller D corresponds to increased mean scour depth)
- Fine fraction in bed surface (related to D, higher proportion fines corresponds to increased mean scour depth)
- Valley width index (less confined channels tend to exhibit greater scour)

All factors are inter-related and exhibit non-linear relationships to scour. A simple model using some combination of these that might be related to scour can be hypothesized, but we currently lack data to evaluate or calibrate such a model. A combination of the above factors is given by an index

$$I_S = QS/WD_{pred} \qquad (16)$$

where D_{pred} is the characteristic grain size predicted by regressing the coarse-fraction model (Eqn. 15) against the D value from Eqn. 12. D_{pred} incorporates all the factors in Eqn. 15: local bedload supply, the fine fraction of the local supply, and channel confinement (via the valley-width index). D_{pred} relates to bed texture, in that larger D_{pred} values are associated with coarser beds, but its relationship to any quantified aspect of the grain size distribution is unknown, and perhaps unwarranted. Thus I_S has all the components required for an index of scour potential, but with no quantitative relationship established to actual scour.

To establish a relative relationship, we can use the distribution of calculated I_S values as a guide. Figure F-4 shows a cumulative distribution of I_S for channels of drainage area greater than one square kilometer and DEM-estimated gradients of less than 7% (approximating the fish-bearing portion of the network). We can set thresholds to define "high," "moderate," and "low" ranges. For example, for the current scenario, 50% of the fish-bearing network is low, 30% moderate, and 20% high. The value of such arbitrarily defined thresholds is in comparing the proportion of the channel network falling within these classes under different scenarios. Additionally, because a value is defined for each segment, the locations experiencing changes in scour potential between different scenarios can be identified.

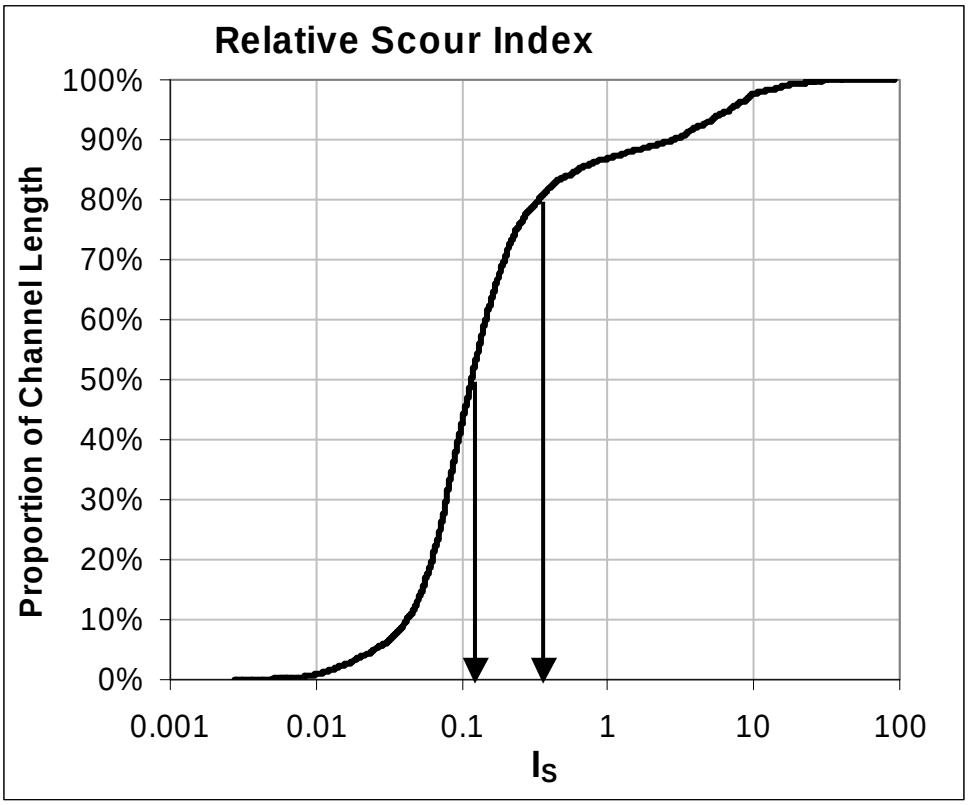

Figure F-4. Cumulative distribution of "scour index" (I_S, Eqn. 16) over low-gradient (< 7%, potentially fish-bearing) channels. This distribution can be used to define thresholds to compare scour potential between different scenarios. In this case, for example, 50% of all values fall to the left of the left-most arrow and 20% of all values fall to the right of the right-most arrow. Using the corresponding I_S values, the proportions will differ with different scenarios: if the proportion falling to the right increases, scour potential in the basin has increased.

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Lewis River Case Study Final Report

A decision-support tool for assessing watershed-scale habitat recovery strategies for ESA-listed salmonids

Appendix G: Bankfull Width Model

May 2007

D. Jensen

Introduction

Many of the models in the Lewis River Case Study DSS required an estimate of stream width for all reaches within the study area, to allow predictions of habitat parameters and thresholds models. Bankfull width (BFW) measurements were necessary for defining riparian habitat thresholds (Appendix H), fish habitat thresholds and suitability (Appendix I, Appendix J), and calculating spawner capacity (Appendix I). In addition to the DSS, BFW measurements were also used for a study designed to assess the accuracy and precision of BFW field estimates. The specific sampling methods for this study were used for both applications, and are described in this appendix.

We developed a predictive bankfull width model for the Lewis (applicable to similar watersheds) from field BFW measurements provided by the U.S. Forest Service, U.S. Fish and Wildlife Service and Pacific Watershed Institute (PWI). In addition to these, 282 BFW measurements were calculated from ortho photos by the Pacific Watershed Institute, and limited field measurements were collected by the LRCS group in the summer of 2003 (n = 44). We applied the final models to all stream reaches, and used the predicted BFW values to calculate some of the final DSS results.

LRCS Field sampling

Field sampling by LRCS members (N=88) was conducted near several bridge crossings over both the East Fork and the North Fork of the Lewis River (5 of 11 bridges). In addition, an inventory was taken of all tributaries to the Lewis River that empty directly into the mainstem (i.e., other tributaries and branches were not included). The inventory excluded first order streams, as shown on 1:24,000 topographical maps published by the U.S. Geological Survey and did not include intermittent streams in the determination of stream order. BFW measurements were taken both upstream and downstream of 3 bridges over the Lewis River, and in one direction (chosen randomly) from the two other Lewis River bridges and from all tributary crossings where the stream bed was reasonably accessible. The protocol was to go between 180 and 220 meters up or down from the Lewis River Bridge or 80-120 meters from the tributary crossing. This distance was chosen randomly. BFW measurements were taken at this location with a laser range finder, and at 5 meters above and below this point. Two observers were present on most surveys, and both took the measurements independently. Subsequent to these measurements, the survey crew moved to the adjacent habitat unit (either pool to riffle or vice versa). The direction was randomly selected, as was the location for the center sampling location. Measurements were then taken 5 meters above and below the second center point. The average of all measurements in both habitat units was used to fit the model.

Geographic locations of each site were recorded in Universal Transverse Mercator (UTM) coordinates using a hand-held GPS. The GPS accuracy is less than 15 meters[1] and the DGPS (WAAS) accuracy is 3-5 meters. The DGPS was used whenever possible. For sites where we were unable to obtain a GPS fix, the site description was recorded

[1] Subject to accuracy degradation to 100 meters 2DRMS under the U.S. DOD-imposed Selective Availability Program.

with respect to nearby landmarks, and the laser rangefinder was used to obtain an estimate of distance from the landmark. Digital 1:24,000 topographic quadrangle maps were used to manually locate and create GIS points for missing site data, and these were joined with the GPS-derived site locations. Field data were linked to the spatial data. All site locations were snapped to the base stream coverage (WDFW-SSHIAP 1:24,000 streams), and spatial and non-spatial stream attributes were linked to the BFW data. In addition, a spatial join was performed, linking field sites to NetStream, a modeled stream coverage containing additional stream attributes of interest.

Calibration BFW Data

The three sources of calibration data required some manipulation and analyses in order to maintain consistency between data sources. Details for each source are as follows:

- **Mainstem Measurements (PWI).** Bankfull width field data (N=28) from PWI was available for the mainstem channel of the upper East Fork River above Sunset Falls and Green Fork River and was joined to both the SSHIAP-WDFW and NetStream spatial data. Data were collected by PWI using tapes. Three to seven bankfull widths were taken in each geomorphically-defined reach and averaged (PWI 1998).
- **Aquatic Habitat Survey data (USFS, USFWS).** Multi-year tabular aquatic habitat survey unit-level stream data (N = 481) were obtained from Gifford Pinchot National Forest (GPNF) and the U.S. Fish and Wildlife Service (one stream) (USFS 1995; USFWS 2000). Tabular data were converted to GIS by calibrating survey reaches and unit measurements, and dynamically segmented onto the GPNF or the WDFW-SSHIAP 1:24,000 routed stream coverage, depending on the survey source (ESRI 2004; USFS 2003; WDFW 2000). Width sites were converted to points, and joined to both WDFW-SSHIAP and NetStream stream habitat variables using methods described above.
- **Determining width remotely (PWI).** Bank full width measurements were manually extracted from 1:12,000 digital ortho-photographs (N=282). Bankfull channel edge was defined as the active channel with vegetation type and age used to delineate the boundaries. On-screen digitizing was used to measure channel features and to record site locations. Channel type and vegetation density was recorded to provide an uncertainty metric on channel visibility. Previous work indicates that photo measurements tend to underestimate bankfull width by an average of 5.9 meters (Beechie et al 2005), so orthophoto measurements were adjusted by this amount.

Geographic variability

The eruption of Mt. St. Helens in 1980 impacted a portion of the North Fork Lewis River watershed, particularly the area around Muddy, Pine and Smith Rivers. The large debris flows following the eruption resulted in BFWs that are larger than they would have been prior to the eruption and thus could not be modeled with the same relationship used to model the non-impacted streams. The volcano impacted reaches were identified from aerial photos and a separate model was generated for these areas.

Model details

The upstream watershed area for each stream segment was calculated by Dan Miller, and cumulated mean annual precipitation (mm) and reach gradient parameters were available from an earlier version of a Netstream stream network (Miller 2004). These three variables were used as potential predictors. If the positional distance between the BFW measurement location and the nearest NetStream reach was greater than 100 m (n= 2789; 8% of reaches), the NetStream data was considered unrepresentative of the BFW site location. At these sites, NetStream variables were not used, and only drainage area was used to model bankfull width.

Prior to fitting the models, the distribution of the data was assessed. The BFW data were extremely skewed, so a log transformation was used. A normal probability plot indicates some departures from normality, though these are not serious. Each of the predictor variables was also transformed to the log scale to increase the linearity between the predictor and the response.

A series of models was fit to the log BFW data that included all combinations of the three predictors, and all interactions among them. Plots revealed that the relationship between drainage area and BFW was curvilinear, even on the log-log scale, so a quadratic term was included in the candidate set of models as well. The model fit was evaluated by comparing AICs, the model with the smallest AIC was selected for making the basin-wide predictions.

A separate BFW model was needed for volcano-affected reaches in the watershed. For the volcano-affected reaches, the smallest drainage area in the calibration data set was 0.392 km^2, and 25.5% of the predictions were for reaches with drainage areas less than this area. However, the problem with increasing BFW predictions in small watersheds was not present, presumably because there was no interaction between drainage area and either precipitation or gradient.

Another model that only had drainage area and the square of drainage area was also fit to the data. This model decreases to the lower limit of drainage area. A graph of model predictions from the two models revealed that the predictions from both were nearly equal when drainage area was about 1 km^2, regardless of precipitation or gradient, thus this was a good transition point from the full model to the drainage area only model. This model was also used for stream reaches that did not have precipitation and gradient estimates.

Model Results and Explanation

The models were subsequently used to predict mean bankfull width throughout the Lewis River basin. An evaluation of the predictions revealed several problem areas. In both of the models, drainage area was clearly the most influential predictor.

The model for the non-volcano affected reaches was a quadratic function of drainage area, and while the model does a good job of predicting BFW for reaches within the range of values used for calibrating the model, the predictions for reaches with drainage areas less than 1 km^2 show a tendency to increase with drainage area for most reaches. This arises because of the 751 observations used to calibrate the model for non-volcano affected reaches, the smallest drainage area was 0.293 km^2, while 46% of the stream reaches in the Lewis River watershed had drainage areas less than this value. Many of the model predictions in stream reaches with drainage areas less than 1 km^2 were very large, particularly in areas of low precipitation and steep channel gradients.

The difference in predictions across the range of gradient and precipitation decreased substantially with increasing drainage area, the differences were only moderate for reaches with drainage areas greater than 1 km^2. The large predictions in small watersheds likely arose because of the interaction terms in the model and the fact that there were no observations to calibrate the model in small watersheds.

A second problem was a tendency for BFW predictions to sometimes change dramatically from one reach to the next. This latter behavior was determined to be largely due to changes in channel gradient between adjacent reaches. These changes were amplified by the interaction terms involving gradient. Gradient was removed from the model and the predictions compared to those from the model with gradient. The AIC from the reduced model was 906.1 while that from the full model was 874.1, indicating that the reduced model is likely not the best model for generating the observations (Burnham & Anderson 1998). However, the root mean squared prediction error (RMSPE) for the reduced model was slightly lower than that for the full model (4.10 vs. 4.00) suggesting that the reduced model described mean BFW as well as the larger model, at least in the reaches with observations. Gradient was removed from all models and the models were recalibrated. A reexamination of the predictions from the two models from non-volcano affected areas indicated that the drainage area where the predictions coincided had shifted upward, to 1.43 km^2, so that value was used as the transition point.

There were approximately 500 reaches that did not have drainage area estimates. If the reach was at the upstream terminus of the stream (indicated by an increase in the drainage area of the next segment), then the reach was given the same BFW as the adjacent reach. If the reach was between two reaches, then the logBFW from the two adjacent reaches was averaged and back-transformed. Figure G-1 shows predictions from the three models.

For the stream reaches not impacted by Mt. St. Helens, the model for drainage areas larger than 1.43 km^2 was:

$$\log(BFW) = 3.43 - 5.20\ \log DA + 0.94\ \log DA^2 - 0.23\ \log Precip$$

(1.77) (1.23) (0.20) (0.23)

$$+ 0.68\ \log DA * \log Precip - 0.12\ \log DA^2 * \log Precip$$

(0.16) (0.025)

For the non-volcano impacted reaches with drainage areas smaller than 1.43 km^2 and reaches without reliable precipitation estimates the model used was:

$$\log(BFW) = 1.65 + 0.28\ \log DA + 0.018\ \log DA^2$$

(0.062) (0.035) (0.0046)

For the volcano impacted reaches, the model used for all reaches was:

$$\log(BFW) = 12.30 + 0.22\ \log DA - 1.20\ \log Precip$$

(6.03) (0.049) (0.76)

where:

log(BFW) is the natural log of bankfull width in meters

logDA is the natural logarithm of watershed area above the reach in km^2

logPrecip is the natural log of the cumulative annual precipitation in mm

Standard errors of parameter estimates are shown in parentheses

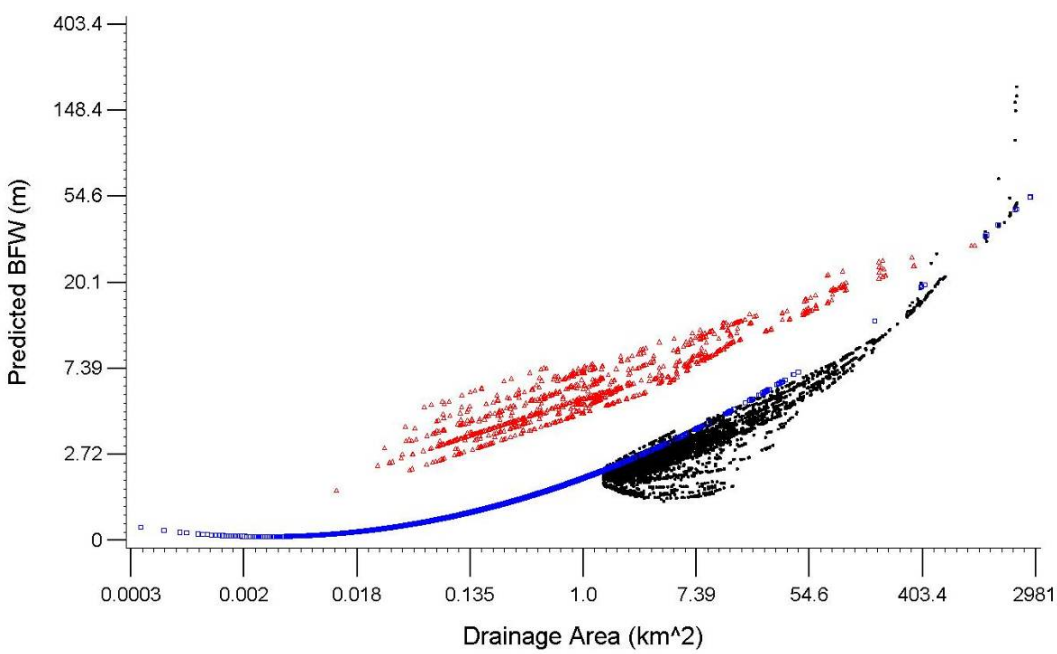

Figure G-1: Predicted BFW vs. drainage area from each of the three models used in the Lewis River basin. Blue squares are predictions from non-volcano affected areas with drainage area < 1.43 km^2, black dots are predictions from non-volcano affected areas with drainage area > 1.43 km^2 and red triangles are predictions from volcano affected areas. Note: scales on both axes are logarithmic.

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Appendix H: Riparian Function Model

May 2007

Riparian Function Model

The assessment of historical conditions and field verification of vegetation showed that deciduous trees, particularly Black Cottonwood, were the dominant tree species in the lower Lewis River watershed. Red alder, cottonwood, and big leaf maple are important riparian species in the upper watershed as well (Appendix B). Thus, we include effects of deciduous tree cover in our models of riparian function. We have three sub-models to describe riparian function: shade, recruitment of pool-forming conifers, and wood recruitment. Each submodel considers both coniferous and deciduous tree cover (i.e., tree crown cover) and conifer size (diameter at breast height, dbh). In each sub-model, we use bankfull width (Appendix G) to separate streams into small and large size categories; this division differs for each of the modeled riparian functions. For each submodel, we also identify naturally poor conditions separately from management-related poor conditions. Naturally poor conditions represent areas such as high altitudes or prairies where large trees are not expected to grow.

The shade submodel is modified from the Washington Forest Practices Watershed Board's Assessment Method (WFPB 2000). Our first approximation used the westside temperature equation for the entire watershed. However, based on recommendations given to us by the experts convened to develop the expert and landscape strategies, we learned that this model was not relevant to areas affected by volcanic activity. Therefore, we applied the eastside temperature equation to the upper Lewis watershed that was affected by Mt. St. Helen's 1980 eruption, and the westside temperature equation to the remaining areas in the watershed. The model is based on elevation (high: >1400 m; moderate: $\leq$1400 m and $\geq$500 m; and low: <500 m), stream width (small: <30 m bankfull width; large: $\geq$30 m bankfull width), and percent total tree cover. Shade ratings are determined based on percent tree cover for four categories: 1) streams at high elevations; 2) large streams at moderate or low elevations; 3) small streams at moderate elevation; and 4) small streams at low elevation. Model parameters and relationships are detailed in Figure H-1.

The pool formation submodel is based on tree size for conifers in relation to bankfull width and channel gradient (Montgomery et al. 1995). Equations relating tree size (dbh) with bankfull width were developed from research data for the Pacific Northwest (Bilby and Ward 1989: Bilby and Ward 1991; Beechie and Sibley 1997; Beechie et al 2000, USFS 1999). Pool formation ratings are determined based on conifer size for three categories, all of which have >30% conifer cover and $\leq$4% gradient : (1) <4 m bankfull width, (2) 4-20 m bankfull width, and (3) $\geq$ 20 m bankfull width and gradient $\leq$ 2%. Model parameters and relationships are detailed in Figure H-2.

The wood recruitment submodel was modified from the Washington Forest Practices Watershed Board's Assessment Method (WFPB 2000) using similar criteria for size, dominant tree type (deciduous, conifer or mixed), and density (percent cover was used as the density metric). Modifications are mostly related to the role of Black Cottonwood providing large wood functions in the lower Lewis watershed, where we included deciduous and mixed categories, separated by stream size. Large woody debris recruitment ratings are determined for conifer-dominant, deciduous-dominant, mixed-

dominant, and naturally poor cover riparian areas. Model parameters and relationships are detailed in Figure H-3.

Qualitative ratings from each submodel were either used as final metrics, combined into an overall riparian function score for use as a modifier that reduces sediment input, or a combined score for input into the FishEye model (Appendix J).

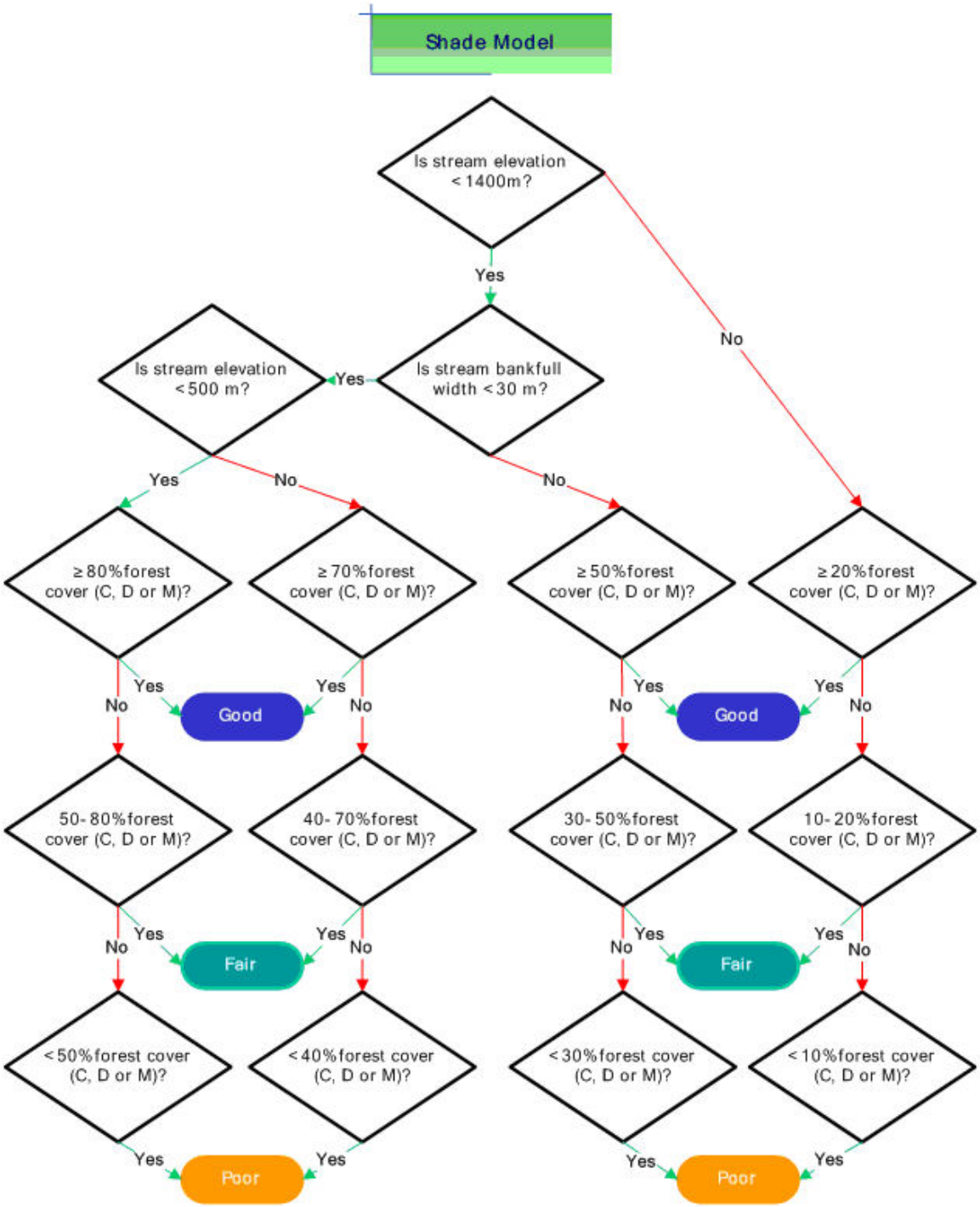

Figure H-1: Diagram of the DSS decision tree for the shade component of the riparian function model.

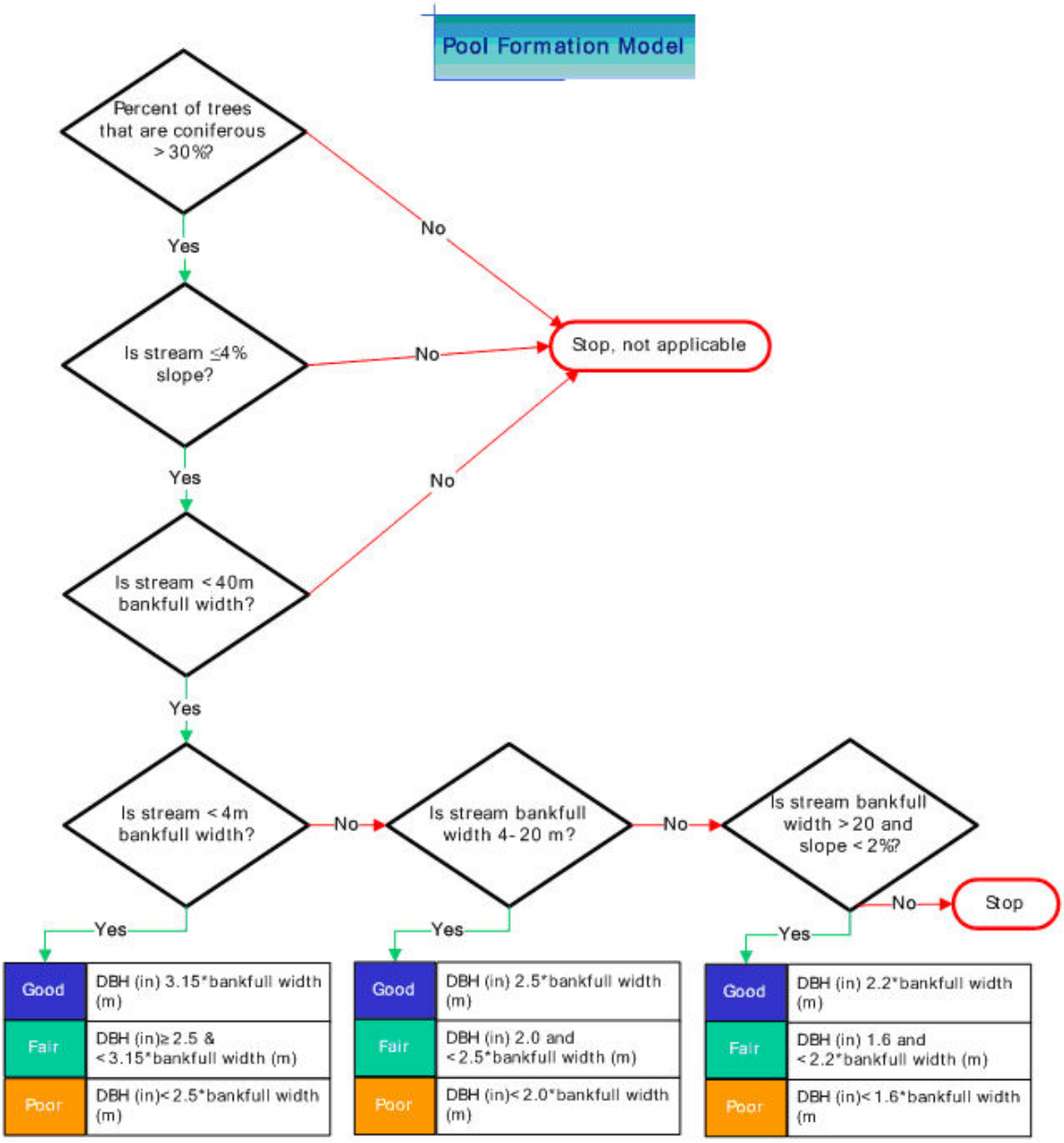

Figure H-2: Diagram of the DSS decision tree for the pool formation component of the riparian function model.

Figure H-3: Diagram of the DSS decision tree for the large woody debris recruitment component of the riparian function model.

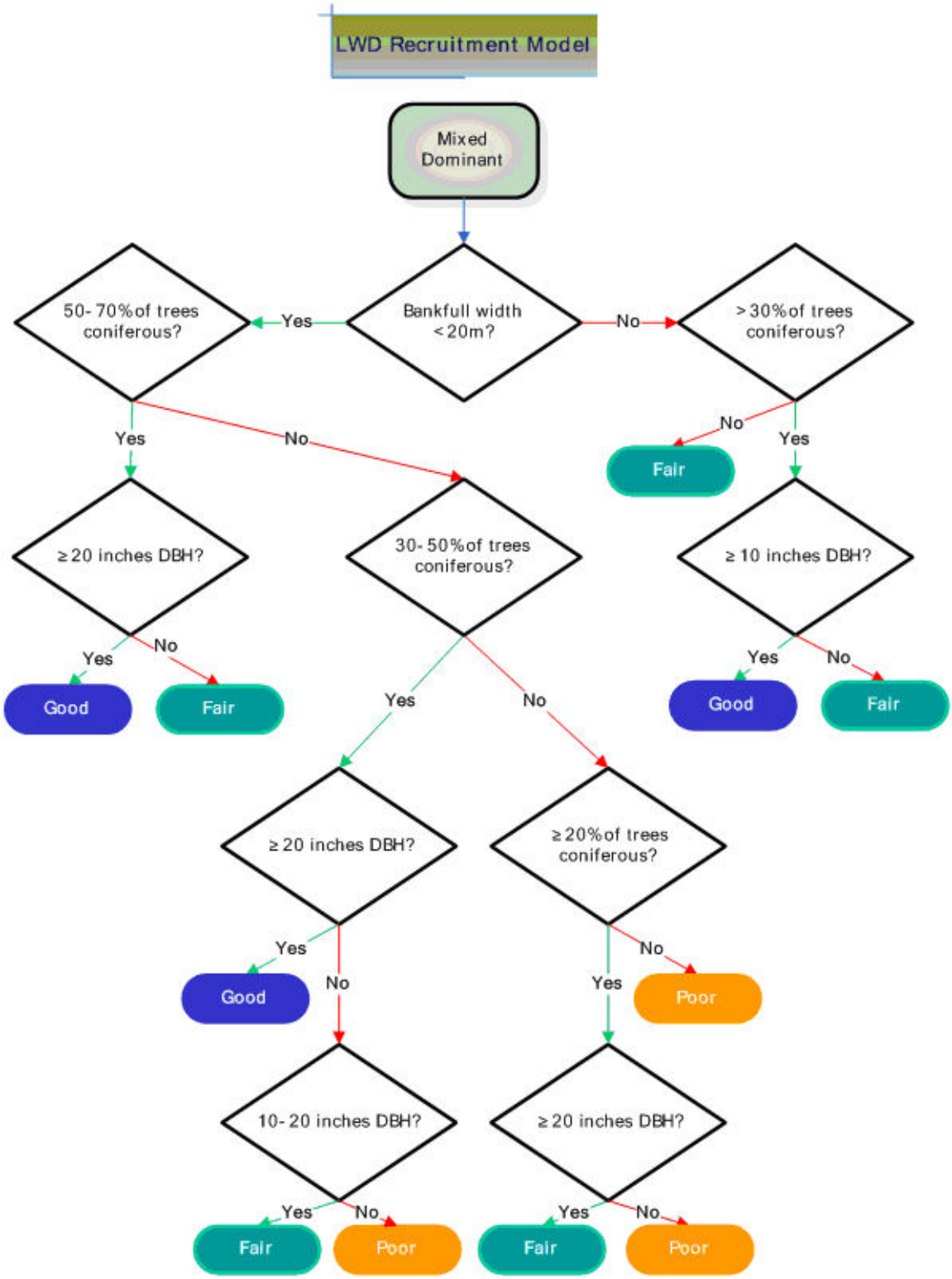

Diagram of the DSS decision tree for the large woody debris recruitment component of the riparian function model (continued).

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Appendix I: Chinook Spawner Suitability and Potential Capacity Estimates

May 2007

Introduction

To estimate spawning suitability of each stream reach for chinook salmon, and to calculate potential capacity of adult spawners per reach, we adapted methods developed in the Puget Sound region (Lunetta et al 1997; Beechie et al. 2006, Sanderson et al. In Prep.). These analyses used remotely sensed data to estimate habitat conditions over a large spatial extent in the absence of field measurements. Data required for this approach included stream gradient, bankfull width, and riparian seral stage (as an index of habitat complexity). Stream channel gradients were estimated following methods in Miller (2003) using 10-m digital elevation model and stream network data. We used bankfull width estimates modeled from precipitation, stream gradient, and field measurements throughout the Lewis watershed (Appendix G). The analyses developed in Puget Sound used a predominantly conifer-dominated classification of riparian seral stage. Below, we describe how we derived seral stage information that incorporated riparian conditions dominated by hardwood and shrub riparian species such as those found in lowland regions in the Lewis River watershed.

Riparian Seral Stage

Initially, we obtained seral stage information for streams in the Lewis River watershed under current conditions from Wade (2000); hereafter, the "original dataset." This information was derived from the same source data and approach as the Lunetta et al. (1997) study (Table I-1), and indicated the proportion of each stream reach in coniferous late, mid, and early seral stage, as well as land cover categories of deciduous, non-forested, and water. We also summarized this information into dominant seral stage values for each reach in the watershed. We determined dominant seral stage by assigning the seral stage or cover type covering >60% of the reach. Reaches without a dominant seral stage class were considered "mixed."

To estimate potential future habitat spawning suitability and capacity resulting from each management strategy, it was necessary to develop a method for estimating seral stage for each reach after we altered the underlying vegetation dataset to represent effects of actions (i.e., riparian restoration) (Appendix D). We derived seral stage values from a vegetation dataset composed of canopy cover class and conifer tree diameter from the Interagency Vegetation Mapping Project (IVMP; BLM 2001). We chose this dataset because we also used it for all other vegetation components in the Decision Support System (to develop land cover classes used in sediment and hydrology models (A) and to derive riparian function ratings (Appendix H). We summarized the pixel-based IVMP dataset into reach-specific metrics of percent canopy cover, canopy cover type, and average conifer size (Appendix A).

We developed a simple model to classify tree cover and conifer size from the summarized IVMP dataset into seral stage categories (Table I-1). To determine appropriate thresholds for IVMP information that best approximated seral stage categories in the original dataset, we compared dominant seral stage values for each reach in the original dataset to summarized metrics for corresponding reaches in the IVMP dataset under current conditions. In the model fitting stage, we initially set threshold parameters of cover class and conifer size in the IVMP dataset to those used to create the original dataset using the Lunetta et al. (1997) approach. For example, we distinguished between late and mid-seral stage categories based on requirements for the proportion of the reach in conifer cover and on conifer size (Table I-1). We then adjusted the parameters based on how well the metrics in the summarized IVMP dataset (e.g., average conifer size, percent conifer cover, percent total tree cover) fit into original dataset seral stage categories. To do this, we plotted each IVMP metric against each original seral stage category to determine the best threshold value. For example, the apparent best cutoff between mid-seral and late seral for average conifer size measured in diameter at breast height (dbh) was 20 inches (most of the records with sizes >20 inches fell into the original dataset category 'late seral' and vice versa). Finally, we adjusted parameters using MS Excel Solver to optimize the number of matched results classified by our model (using summarized IVMP metrics) and by the Wade (2000) approach used to create the original dataset. We used this model to develop seral stage values for each reach in the watershed for each management strategy. Results of the two methods under current conditions were spatially similar (Figure I-1).

Habitat Suitability

We estimated suitability of each reach for chinook spawners following methods developed by Sanderson et al. (In Prep.) and classified each reach as not suitable, poor, fair, or good for spawning (Table I-2). Stream reaches with gradients >4% or <5 m bankfull width were considered not spawnable by chinook salmon (Lunetta et al. 1997; Beechie et al. 2006). Streams 5–25 m wide were assumed to be predominantly forced pool-riffle, pool-riffle, and plane-bed habitats (Montgomery et al. 1999). Stream channels of this size are likely influenced by riparian conditions; hence we used seral stage as an indicator of potential stream habitat complexity. We calculated habitat suitability for chinook for each reach in the watershed for current conditions and under each management strategy.

Potential Capacity

We calculated potential capacity of reaches throughout the watershed suitable for spawning by chinook salmon based on bankfull width, stream gradient, and riparian seral stage (Table I-3) (Beechie et al. 2006; Sanderson et al. In Prep.). Values for equation parameters are from published studies (Table I-4). Equations used to calculate capacity estimates (N) for each reach (Table I-3) are:

Equation 1

N = ((stream area * % spawnable)/redd area) * (spawners/redd)

where stream area = reach length * bankfull width.

Equation 2

N = stream length * spawners/redd * redds/$km_{FPR/PR}$

Equation 3

N = (stream length * proportion FPR/PR * spawners/redd * redds/$km_{FPR/PR}$) + (stream length * proportion PB * spawners/redd * redds/km_{PB})

where FPR = forced pool-riffle, PR = pool riffle, and PB = plane-bed habitat

Table I-1. Seral stage categories described by Lunetta et al. (1997) and by our model using summarized IVMP (BLM 2001) metrics.

Model	Late seral	Mid seral	Early seral	Deciduous	Mixed	Other (vegetated)	Non-forest
Lunetta et al. (1997)	>50% conifer trees with DBH >20”	<50% conifer trees with DBH >20”	<75% of land cover deciduous forest, remainder conifer	NA	NA	>70% of land cover deciduous forest	>70% of land cover non-forest (agriculture, urban, or barren)
Our model (IVMP data)	≥69.8% conifer cover and DBH >22.5”	≥69.8% conifer cover and DBH >6.8”	≥69.8% conifer cover and DBH ≤6.8”	>39.3% deciduous cover	<69.8% conifer cover and <39.3% deciduous		total tree cover ≤24.2%

Table I-2. Components that determine reach-specific habitat suitability ratings for chinook spawners (after Sanderson et al. In Prep.).

Bankfull Width	Stream Gradient	Riparian Seral Stage	Suitability Score
any	> 0.04	any	not suitable
< 5 m	any	any	not suitable
5 – 10 m	< 0.01	any	fair
	0.01 – 0.04	late	good
		mid	fair
		not late/mid	poor
10 – 25 m	< 0.01	any	good
	0.01 – 0.04	late	good
		mid	fair
		not late/mid	poor
> 25 m	< 0.01	any	good
	0.01 – 0.04	any	fair

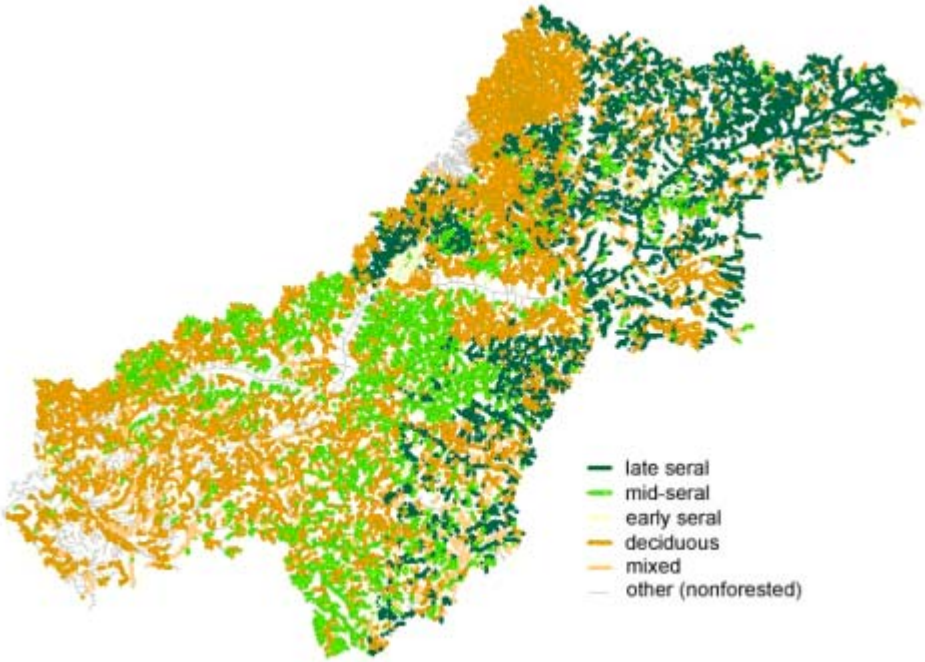
Table I-3. Equations used to determine reach-specific potential capacities for chinook spawners (after Sanderson et al. In Prep.).

Bankfull width	Stream gradient		
	<1%	1-4%	>4%
>25 m	Equation 1	Equation 1	Not suitable
10-25 m	Equation 2	Equation 3 (influenced by riparian seral stage)	Not suitable
5-10 m	Equation 2	Equation 2	Not suitable
<5 m	Not suitable	Not suitable	Not suitable

Table I-4. Parameters used in Equations 1-3 for estimating capacity of a reach for adult chinook spawners (after Sanderson et al. In Prep.). Values are means with 10% and 90% confidence intervals in parentheses.

Equation Parameter	Value	Reference
% spawnable	6.24%	Beechie et al. In Press (data for the Skagit and Stillaguamish Rivers in Puget Sound)
redd area	15.25 (27.90, 4.90)	Beechie et al. In Press (D. Hendricks, WDFW unpublished data)
spawners/redd	2.33 (1.35, 3.50)	Beechie et al. In Press (D. Hendricks, WDFW unpublished data)
redds/$km_{FPR/PR}$	36.40 (7.97, 61.30)	Montgomery et al. 1999
redds/km_{PB}	1.77 (0.0,6.0)	Montgomery et al. 1999
proportion forced pool-riffle/pool-riffle	late seral conifer riparian: 1.0	Lunetta et al. 1997
	mid seral conifer riparian: 0.78	
	early seral conifer, mixed conifer/deciduous or deciduous riparian: 0.74	
	non-forested or other riparian: 0.35	
proportion plane-bed	1.0 – proportion forced pool-riffle	Montgomery et al. 1999

A. Wade (2000)



B. Our model

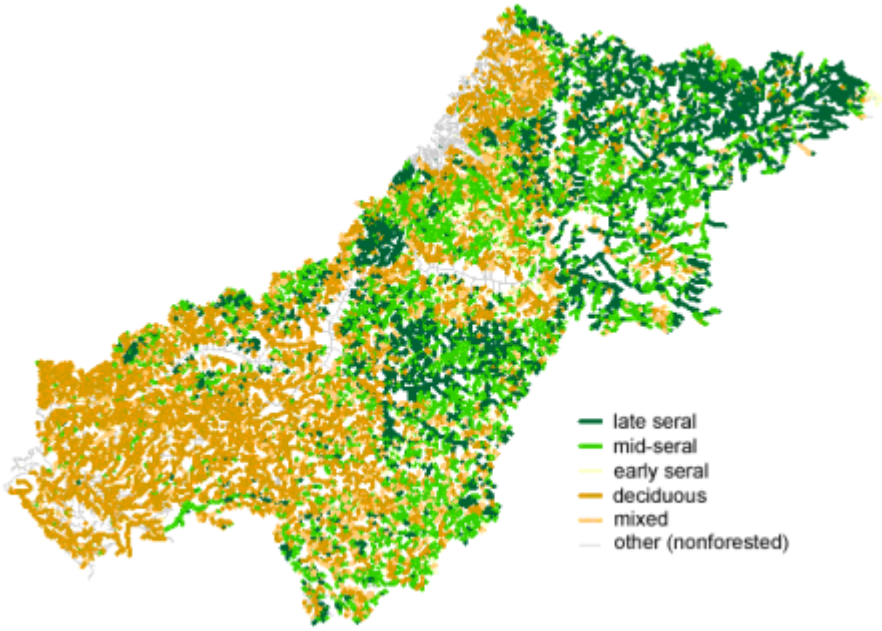


Figure I-1: Spatial comparison of seral stage-classified reaches. (A) Dataset described by Wade (2000), classified using the approach of Lunetta et al. (1997), and (B) our seral stage model using summarized IVMP data (BLM 2001).

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Appendix J: FishEye

May 2007

Introduction

The distribution of salmonids on the landscape is influenced by physical habitat variability and the distribution of anthropogenic development. Habitat quality natural varies with the physical and biological landscape. Migration barriers, restoration activities and natural disturbances can modify the extent of and quality of migration, spawning, or rearing habitat. In the context of watershed restoration and fisheries management, it is useful to differentiate natural and anthropogenic-derived variation in hydrology, sediment, and riparian habitat over entire watersheds.

The natural physical, biological, and biochemical conditions of a stream that support migration, spawning, or rearing of salmonids can be described as the natural, or intrinsic potential of a stream (Burnett et al 2003a; Burnett et al 2003b). Intrinsic potential of streams incorporates features that are driven predominantly by general watershed characteristics that vary over the landscape, such as stream gradient, channel form, channel size, and flow regime (Burnett et al 2003a, Montgomery et al. 1999). These characteristics are driven by geology, geomorphology, and topographic variation, which can be predicted using spatial and hydrological modeling.

The purpose of the FishEye model is to describe the intrinsic potential and modified anthropogenic potential of each reach in the watershed with respect to channel width, gradient, and hydrological regime, and to identify reaches whose current potential is limited by either sediment deposition and scour or poor riparian conditions, either intrinsically or as a result of anthropogenic influences. It is a simple assessment tool that provides a reach-level habitat quality rank for the purpose of making relative comparisons among streams within a watershed. The FishEye framework is a system of assigning and combining habitat quality ranks for physical stream variables and habitat preference ranges for multiple species. We designed this model to determine relative rankings for potential salmonid habitat quality independent of documented fish use, which can vary, depending on data quality.

General Approach

We determined the basic biological and physical parameters needed to reflect both intrinsic potential and anthropogenic effects for habitat variables modeled in the Lewis River Case Study Decision Support System (DSS). Hydrologic modeling and GIS methods were used to derive physical stream-based parameters at the stream reach scale (WDFW 2003; Miller 2005). Methods for creating reach-scale estimates of bed scour, hydroregion, fine sediment, scour potential, and riparian habitat are described or referenced in the following sections.

FishEye ranks each stream segment with an estimate of the base intrinsic potential or natural suitability (using the base potential variables) and an anthropogenic rank based on the anthropogenic variables. The natural suitability is differentiated by species, though the anthropogenic rank is not. The final result of FishEye is a series of ratings per species, delineated by the two components, then classified into suitability rankings.

DSS Application

FishEye is used in the LRCS-DSS as a relative measure of fish habitat quality under current, historical, and a variety of restored watershed conditions or scenarios. Final FishEye scores provide a qualitative method of describing habitat quality that is useful across multiple reaches, or at the sub-watershed scale. It is not predictive at the scale of a single reach (due to modeling and preference uncertainties) and is only appropriately applicable over a length of stream >1 kilometer (depending on predominant reach length) or at the scale of a 7th or 8th field watershed. FishEye scores and results should be clustered or grouped by multiple similar reaches or patches of reaches.

Scores are appropriate to use in conjunction with potential fish capacity predictions in the framework of the associated report (detailed in Appendix I), or can be compared to these predictions. Scores are also appropriate to use as measures of general habitat variability and diversity throughout the watershed under current and historical conditions, and under various modeled restoration scenarios. A few of the metrics used in FishEye are similar to those used in Sanderson et al (In Prep.) and Lunetta et al. (1997). The source of the base stream reach data for FishEye was the Washington Department of Fish and Wildlife (WDFW 2003).

Description of Variables and Scores

The natural suitability component is composed of bank full width, stream gradient, and scour potential. The anthropogenic component includes shade, large woody debris, pool forming conifers, fine sediment, and modified scour potential**.** Each was broken into qualitative categories based on available references for the purpose of designing a weighting and ranking scheme. Classification into ordinal categories was necessary in FishEye to better reflect the nature of the source references. The primary references provide only general habitat range values; we did not have the appropriate empirical data to warrant the development of curves.

Channel Width

Species-specific accessibility and passability information for various salmonids was available for bank full width (Table J-1). Smaller stream channels (< 4 feet) and the higher gradient channels (see next variable) are equally important and both serve as basic limiting factors to fish access, and also function as indicators of general habitat quality. A width score of 0 indicates that this section of stream is unusable, because of severely limited access or low streamflow. This is a limiting factor. The bank full width threshold value is based on an estimate of limits to anadromous fish used by Washington Department of Fish and Wildlife (2000) which, in turn, is based on the definition of stream thresholds for anadromous and non-anadromous fish as defined by Washington Forest Practices Board (WFPB 2000).

Mainstem spawning fish (fall chinook) are limited to channels greater than 10 feet (3 m) (WDFW 2000). A score of 1 indicates less favorable conditions for summer steelhead and chum, based on flow conditions that are typically associated with streams less than 10 feet in width (WDFW 2000; Burnett et al 2003a; Salo 1991). Scores of 2 indicate generally favorable conditions. Scoring of channel width is highly correlated with flows and gradient, so our purpose in including width is to indicating areas with poor potential or that are completely unusable by most adult salmon of the species. Species were assigned only 0's and 2's where source information only indicated two conditions – limited or no access, or width suitable for fish passage. The assessment area for chum in the DSS only includes areas downstream of natural and manmade barriers, so chum results do not include any streams <2 m in width.

Gradient

Gradient indicates species-specific differences in stream accessibility, passability, and physical habitat preference (Table J-1). Boundaries between gradient categories should be considered as general transition zones between suitability, rather than firm threshold breaks in the natural environment. Uncertainty in gradient estimation techniques (both remote and field), scale of reach-level gradient estimates (length of reach), and variation in the size of adult fish, all affect confidence levels in ranking gradient quality for fish. Gradient alone is not an absolute indicator of excellent habitat, obviously, and should be considered in combination with the other variables in FishEye, as well as with multiple environmental variables we are unable to represent in FishEye (fish density, fish behavior, flow, aquatic quality factors, etc.).

Similar to channel width, a 0 indicates complete lack of accessibility or use and is a limiting factor rather than a preference. Information on gradients that are limiting was more consistent between sources than the gradient preference information, and in most cases reflects the upstream natural gradient barrier to fish. Typically, these represent waterfalls or cascades. Longer reach lengths in the GIS stream source can result in an overestimation of gradient rank per reach segment, as the features are not necessarily homogenous within these reaches, but gradient is calculated as an average over the length of the reach (WDFW 2003). Scores 1 - 3 indicate that the stream is useable to the fish, with 3 indicating the most preferred gradient habitat.

Multiple sources were used as references for gradient rankings. Ranges indicated to be indicative of good habitat (3) are positively correlated with fish use and spawning in various reference sources (Steel and Sheer 2003; Cramer, Galovich, Hunt, Hymer, Schroeder and Kenniston, Wade, Ziller – Pers. Comm.; Burnett et al 2003a). Moderate quality gradient habitat (2) was defined using ranges outlined in WDFW (2000) and Burnett et al (2003a). Poor quality or streams indicated to be barely passable (1) are streams with highly variable seasonal access, or are have features that make gradient predictions difficult, or where adjacent habitat indicated high gradient levels (WDFW 2000; Burnett et al 2003a). Cutoffs or limits to accessibility (0) were also obtained from these sources.

Table J-1. Gradient and channel width preference ranges and scores by species. A dash (-) indicates that no information was available, or that data sources did not indicate any preference or information on the particular species or physical characteristic. Column headings indicate species; both score number and score descriptor are provided for each species by gradient level. Score numbers described in the text were converted to descriptors "Good," "Moderate," and "Poor" for ease of interpretation. Asterisks (*) indicate that the reference source differed from the other source indicating "moderate" quality habitat for this species (WDFW 2000).

Gradient (%)	Chinook (spring)		Chinook (fall)		Steelhead (winter)		Steelhead (summer)		Steelhead (Burnett 2003)		Chum	
0 - 1	Fair	*2*	Fair	*2*	Fair	*2*	Fair	*2*	Fair	*2*	Good	*3*
1 - 2	Good	*3*	Good	*3*	Good	*3*	Fair	*2*	Fair	*2*	Good	*3*
2 - 3	Fair	*2*	Fair	*2*	Good	*3*	Fair	*2*	Good	*3*	Fair	*2*
3 - 4	Fair	*2*	Fair	*2*	Good	*3*	Good	*3*	Fair	*2*	Poor	*1*
4 - 5	Fair*	*2*	Fair*	*2*	Good	*3*	Fair*	*2*	Fair	*2*	Poor	*1*
5 - 6	Fair*	*2*	Fair*	*2*	Fair	*2*	Fair*	*2*	Fair	*2*	-	*0*
6 - 7	Fair*	*2*	Fair*	*2*	Fair*	*2*	Fair*	*2*	Fair	*2*	-	*0*
7 - 8	Poor	*1*	Poor	*1*	Fair*	*2*	Fair*	*2*	Fair	*2*	-	*0*
8 - 12	Poor	*1*	Poor	*1*	Fair*	*2*	Fair*	*2*	Poor	*1*	-	*0*
12 - 15	Poor	*1*	Poor	*1*	Poor	*1*	Poor	*1*	Poor	*1*	-	*0*
15 - 16	Poor	*1*	Poor	*1*	Poor	*1*	Poor	*1*	-	*0*	-	*0*
16 - 20	-	*0*	-	*0*	Poor	*1*	Poor	*1*	-	*0*	-	*0*
> 20	-	*0*	-	*0*	-	*0*	-	*0*	-	*0*	-	*0*
Channel Width												
< 2m	-	*0*	-	*0*	-	*0*	-	*0*	-	*0*	-	*0*
2-10m	Good	*2*	-	*0*	Good	*2*	Fair	*1*	Fair	*1*	Fair	*1*
> 10m	Good	*2*	Good	*2*	Good	*2*	Good	*2*	Good	*2*	Good	*2*

Hydrologic Scour Potential

Natural variability and hydrologic regimes in rain-dominant versus snow-dominant systems can impact spawning success through differences in bed scour and egg survival (Montgomery et. al 1999). Hydrologic regime variations across a population's range may indicate spatial diversity and life history adaptations (Beechie et al. 2006). Hydroregion is correlated with differences in natural hydrologically-driven scour and reflects the hydrologic landscape independent of land cover. Hydrologically-based factors have implications for spawning quality based on bed scour and relative success of redds (Montgomery et al. 1999).

In FishEye, we used details from a study by Montgomery et al (1999) to determine hydrologic scour scores by fish species (Table J-2). The dominant hydrologic regime influencing the segment and gradient were used to characterize an overall seasonal scour potential for each species. Based on Montgomery et al. (1999), a high gradient was defined as those stream reaches with >3% channel slope, and low gradient as reaches with < 3% channel slope. It is important to note that gradient is a natural suitability variable, so it is incorporated twice in the FishEye model—under general gradient and in scour potential rules. A score of 0 indicates a negative rank for scour, indicating that adult fish of this species (especially the larger fish) have a high risk of experiencing a negative effect (i.e., low productivity) due to egg mortality from scour events. A score of 1 indicates a neutral effect, or no documented negative or positive effect for this condition. A score of 2 indicates that this condition is a generally positive condition for egg survival, though actual conditions are subject to behavior and environmental variability, such as egg burial depth and actual fish size. The positive score would primarily apply to the larger fish in the cohort, which would have a very low risk of a negative scour impact, due to their deeper egg burial depths (Montgomery et al. 1999).

Table J-2. Hydrologic region scour scores based on Montgomery et al (1999). A score of 1 indicates a neutral or undocumented effect. Chum only have two possible scores, since source data only provided information on possible negative impacts.

Precipitation	Other	Rain		Rain-on-Snow	Snow	
Gradient	**All**	**<3%**	**>=3%**	**All**	**<3%**	**>=3%**
Spring Chinook	1	2	2	1	1	-
Fall Chinook	1	1	-	1	2	2
Winter Steelhead	1	2	2	1	1	-
Summer Steelhead	1	2	2	1	1	-
Steelhead (Burnett 2003)	1	2	2	1	1	-
Chum	1	1	-	1	-	-

Dominant upstream hydrologic regime was calculated for individual stream reaches using existing base hydrologic zone spatial data from Washington State Department of Natural Resouces (WDNR 1991). This hydrologic data layer was derived from information on locale climate, elevation, average January snowpack, and latitude in the Lewis watershed, as defined by Washington DNR.

Hydroregion was determined for each segment using GRID functions in ArcGIS. A grid was created for each of the five dominant hydrologic/ precipitation classes (highland, snow, rain on snow, rain, or lowland). Hydrologic flow direction was determined from

the 10 m digital elevation model. Upstream connectivity and identification of drainage source was identified for all segments by tracking the source and amount of contributing cells to each stream channel. Weighted and non-weighted flow accumulation grids were generated for each of the five hydrologic landscape categories. These grids were summed together, and the hydrologic landscape source was determined for each pixel by dividing the summed flow accumulation values by the original landscape source code number. The dominant region code flowing into the stream for each segment was assigned to each stream segment using a spatial overlay.

Riparian

Riparian condition scores are determined without species-level differentiation. Available references and the limited precision of riparian vegetation data were not appropriate for determining species-level scores. Riparian condition is used to indicate the general quality of the riparian forest with respect to large woody debris recruitment, pool forming conifers, and shade (Appendix H). All species in the analysis have the same score for these riparian variables. A score of 0 indicates poor habitat, 1 indicates fair or moderate with respect to the parameter, and 2 indicates that the riparian conditions meet the rules designated for good riparian conditions for that feature.

Fine Sediment

Fine sediment is the second anthropogenic variable. Sediment deposition from surface run-off or mass-wasting is a natural occurrence that can be amplified by anthropogenic activities. The fine sediment score is an indicator of deposition in the stream channel. It is based on the difference between the natural sediment deposition for the reach (assuming fully vegetated land cover) and the estimated fine sediment deposition for the current conditions. The variable is designed to measure relative differences in the deposition of fines in stream channels in the Lewis. The amount of fines in a reach was measured as the proportion of all sediments with a grain size < 1 mm that are deposited in a stream reach.

The relationship between egg-to-fry survival and sediment particle size was used to determine the fine sediment ranking categories used in FishEye (Table J-3). Figures 1 and 2 of Appendix K refer to the proportion of sediment within streams from field-collected substrate core samples. The sediment information is available as the overall amount of fines deposited in the reach from both lateral and upstream. Although comparable, these ranks are only appropriate for comparisons between reaches, and do not represent within-reach variability. We use these relationships between fish density and egg-survival for relative ranking (between stream reaches) purposes. Because the sediment particle size thresholds were determined from empirical studies, the translation of sediment particle size thresholds from field-measured sediment to modeled sediment predictions includes some uncertainty.

Similar to other FishEye variables, the sediment scores (Table J-3) are designed to reflect the relative quality of habitat and survival based on published studies. As noted in Appendix K, the original studies are based on dividing the percent of fines by size class, with fines defined as substrate less than 0.85 mm. The sediment modeling procedure was based on general soil horizon distributions, and estimation of the proportions of the soil horizons comprised of substrates with a grain size ≤ 1 mm (described in Appendix A).

Table J-3. Sediment scores were determined from the egg-to-fry survival functional relationships as described in Appendix K. Table includes survival by species, associated percent fines, and FishEye score value. Survival by percent fines was used to determine the rank score for FishEye. The rank score, not the survival rate, was used as the final FishEye score.

Species Survival	Percent fines in reach *(proportion of 0 – 1.0mm grain size fine substrate)*		
	Good (%fines/reach) Score = 2	***Fair (%fines/reach) Score = 1***	***Poor (%fines/reach) Score = 0***
Steehead	**0-10**	**10-14.6**	**> 14.6**
Egg-to-fry survival	> 64.7	64.7-32.4	< 32.4
Chinook	**0-5.9**	**5.9-13.3**	**> 13.3**
Egg-to-fry survival	> 37.3	37.3-18.6	< 18.6
Coho	**0-10.2**	**10.2-18.5**	**>18.5**
Egg-to-fry survival	> 56.5	56.5-23.8	< 23.8

Modified Scour

The modified scour potential rank is defined as the relative risk of scour based on the

2.3- year flood flow levels. The 2.3-yr flood magnitude is described further in Appendix E. This variable provides information on runoff and discharge as it relates to in-stream scour potential. The scour potential may change if upstream land use is modified extensively enough to affect runoff / discharge for the 2.3-yr flood magnitudes. A scour index was calculated as part of the sediment modeling procedure, converted into percentile ranks, then grouped into categories of relative quality (Table J-4). Modified scour is related to the hydroregion, except that it incorporates runoff and discharge from the hillslopes and does not incorporate physical channel and precipitation conditions. Modified scour scores are determined using results from the dynamic sediment routing procedure (Appendix F). We used the distribution of scour index (I_s) values for Lewis River streams (Appendix F) under historical conditions within the winter steelhead distribution habitat extent, including current and historical habitat to define natural value thresholds (Figure J-1 and Figure J-2). The distribution of scour indices (I_s) is primarily affected by changes in lateral vegetation (riparian habitat) and variation in land cover type and soil type. We used the distribution of historical landscape conditions to determine the natural percentile breaks in the environment for runoff, discharge, and bed scour, in order to assess anthropogenically-driven changes under current or potential conditions.

Base threshold values were set at proportions of 0.5, 0.3, and 0.2 in the distribution (Figure J-1 and Figure J-2; Figure 4 of Appendix F). Bin ranks assigned based on this distribution were too general to allow effective differentiation between scour risk; I_s values were then differentiated by 20th percentile categories. Index values greater than or equal to 0.11 were given a score of 0 (very high risk) indicates a very high probability of potential scour from land use and corresponding negative impact on habitat from scour events. Index values greater than or equal to 0.08 and less than 0.11 have a very high scour effect risk (rank of 1; very high risk), I_s values greater than or equal to 0.06 and less than 0.08 represent a moderate scour effect risk (rank of 2; moderate risk) within the natural distribution. Index values greater than or equal to 0.03 and less than 0.06 have a low potential for land-use scour risk (score = 3; low risk), and values less than this have a very low potential (score = 4; minimal or close to base conditions).

Table J-4. Bed scour potential and percentile ranks by stream length. Based on natural scenario bed scour values (I_s) (base flood discharge) for streams > 1 km^2 within the extent of the current and historical range for winter steelhead. Percentile ranges are indicated in column headings. FishEye scores apply equally to all species. Distribution of values was used to derive relative ranks. The highest value is the maximum bed scour value (I_s) possible for that segment.

Base Bed Scour (modified)	Base (0-20th)	Low (20-40th)	Moderate (40-60th)	High (60-80th)	Very High (Max)
I_S (baseflood discharge) range	0 - 0.0243	0.0243 - 0.0606	0.0606 - 0.0835	0.0835 - 0.1182	> 0.1182
FishEye Score	4	3	2	1	0

Integration of Variables

Scores from all variable components were combined to provide a rating to reflect the habitat quality for the stream reach. Habitat variables that incorporate the three natural suitability scores were combined to create a unique code that reflects the base intrinsic potential of the reach. Riparian, fine sediment and modified scour scores vary by land use conditions, upslope land cover, and riparian habitat quality. These scores were not incorporated into the natural suitability rating, since they reflect anthropogenic changes in current and restored watershed conditions. A binary combination method was used to allow discernment of unique combinations of variables throughout, and final qualitative, descriptive ratings were determined from these.

Combining Habitat Scores

The natural suitability component incorporates preference differences between species. This means that each species may have a different natural suitability code for the same stream segment. The Results section includes a description of case limitations by species. The multiple values were converted to a single binary score in an equation, by making the multiplier equal to the maximum number of scores within each of the incorporated variables, raised to the power of the number of variables (initiated with a power of 0), minus one for each variable in order. The variable with the most weight occupies the first position in the equation, and the variable in the last position has the least weight, since it is raised to a power of 0.

The binary code for natural suitability was calculated with width as the highest weight, since this variable was primarily a limiting factor rather than a quality indicator. The code was calculated as follows (Eq. 1):

Equation 1

$$W(4^2) + G(4^1) + HS(4^0) = \text{Natural Suitability Code}$$

Where:

W = width score

G = gradient score

HS = hydrologic scour potential score

The description for scores by species can be tracked back to original values in Table J-1 and Table J-2. The natural suitability variable combinations and codes are in Table J-1. These codes were used specifically to allow various conditions to be easily grouped into ratings of good, fair, and poor. Below is an example of the unique binary codes derived from Equation 1:

A BP of “21” = $1(4^2)+1(4^1)+1(4^0)$

A BP of “42” = $2(4^2)+2(4^1)+2(4^0)$

A BP of “58” = $3(4^2)+2(4^1)+2(4^0)$

Riparian condition code was calculated with a similar approach (Table J-1). In this binary code equation, the multiplier is 3 since there is a possibility of a maximum of 3 unique codes among the 3 variables, raised to the power of the number of variables (minus 1) (Eq. 2):

Equation 2

$$S(3^2) + LWD(3^1) + PFC(3^0) = \text{Riparian Condition Code}$$

Where:

S = shade score

LWD = large woody debris score

PFC = pool forming conifer score

Ranks for fine sediment and modified scour were combined into a binary code representing the modified physical function of the stream, or the instream habitat (Table J-1). This code also represents anthropogenically modified habitat. The multiplier is 5 since there is a possibility of a maximum of 5 unique scores from modified scour and fine sediment variables (Eq. 3).

Equation 3

$$MS(51) + FS(50) = \text{Instream Habitat Code}$$

Where:

MS = modified scour potential score

FS = fine sediment score

Table J-1. Final binary scores for all combinations of natural suitability, riparian conditions, and instream habitat. Scores used for binary code calculation are included. The codes were used to determine qualitative rating interpretations for each variable. The ratings are: P = poor, VP = very poor (modified physical function intermediate rating only), F = Fair, M = marginal / moderate, G = good, VG = very good (modified physical function intermediate rating only), and E = excellent (low or naturally low) (modified physical function final rating only).

Base Potential Variables (BP)					Riparian Function (RF)					Modified Physical Function (MPF)				
Wid.	Grad.	Hydro	Bin. Code	Rtg.	Shade	LWD	PFC	Bin. Code	Rtg.	Scour Pot.	Fine Sed	Bin. Code	Int. Code	Rtg.
0	0	0	0	P	0	0	0	0	P	0	0	0	VPP	P
0	0	1	1	P	0	0	1	1	P	0	1	1	VPM	P
0	0	2	2	P	0	0	2	2	P	0	2	2	VPG	F
0	1	0	4	P	0	1	0	3	P	1	0	5	PP	P
0	1	1	5	P	0	1	1	4	P	1	1	6	PM	F
0	1	2	6	P	0	1	2	5	P	1	2	7	PG	F
0	2	0	8	P	0	2	0	6	P	2	0	10	MP	F
0	2	1	9	P	0	2	1	7	M	2	1	11	MM	M
0	2	2	10	P	0	2	2	8	M	2	2	12	MG	M
0	3	0	12	P	1	0	0	9	M	3	0	15	GP	M
0	3	1	13	M	1	0	1	10	M	3	1	16	GM	G
0	3	2	14	M	1	0	2	11	M	3	2	17	GG	E
1	0	0	16	P	1	1	0	12	M	4	0	20	VGP	M
1	0	1	17	P	1	1	1	13	M	4	1	21	VGM	G
1	0	2	18	P	1	1	2	14	M	4	2	22	VGG	E
1	1	0	20	M	1	2	0	15	M	-	-	-	-	-
1	1	1	21	M	1	2	1	16	G	-	-	-	-	-
1	1	2	22	M	1	2	2	17	G	-	-	-	-	-
1	2	0	24	M	2	0	0	18	M	-	-	-	-	-
1	2	1	25	M	2	0	1	19	G	-	-	-	-	-
1	2	2	26	M	2	0	2	20	G	-	-	-	-	-
1	3	0	28	M	2	1	0	21	M	-	-	-	-	-
1	3	1	29	M	2	1	1	22	G	-	-	-	-	-
1	3	2	30	G	2	1	2	23	G	-	-	-	-	-
2	0	0	32	P	2	2	0	24	G	-	-	-	-	-
2	0	1	33	P	2	2	1	25	G	-	-	-	-	-
2	0	2	34	P	2	2	2	26	G	-	-	-	-	-
2	1	0	36	M	-	-	-	-	-	-	-	-	-	-
2	1	1	37	M	-	-	-	-	-	-	-	-	-	-
2	1	2	38	M	-	-	-	-	-	-	-	-	-	-
2	2	0	40	M	-	-	-	-	-	-	-	-	-	-
2	2	1	41	G	-	-	-	-	-	-	-	-	-	-
2	2	2	42	G	-	-	-	-	-	-	-	-	-	-
2	3	0	44	G	-	-	-	-	-	-	-	-	-	-
2	3	1	45	G	-	-	-	-	-	-	-	-	-	-
2	3	2	46	G	-	-	-	-	-	-	-	-	-	-

FishEye Final Ratings

Results of the FishEye approach are generated at two levels – the individual habitat variable ratings, and the overall observed suitability rankings. The habitat ratings can be used to indicate the source of habitat limiting factors by stream segment – from either natural intrinsic features, or anthropogenic impacts (Table J-1 and Table J-1). The observed suitability ranking is meant to estimate the overall habitat quality (Table J-1).

Habitat-specific Rating Process

The binary codes for natural suitability and the anthropogenic components are designed to provide continuous gradations in variable scores (Table J-1). Since variable thresholds differ by weight, influence, source, and sensitivity, their associated binary codes are distinct from each other. This design provides flexibility and transparency in the ranking process, and allows the discernment of variables per segment reach. Secondarily, the combinatorial framework is designed to bin final scores into general qualitative ratings The ordered binary codes were used to determine more descriptive qualitative ratings, relative to the distribution of conditions in the Lewis River. Individual cases were reviewed and the qualitative ratings in Table J-1 were adjusted where necessary.

Natural suitability, instream habitat, and riparian condition ratings were reviewed by project personnel for accuracy, and adjustments were made based on this process. We determined certain rules to the assignment of qualitative ratings for consistency. Width is a limiting factor, so a 0 for width means the habitat can never be considered good, and are generally considered to be poor habitat. The limiting factor for natural suitability where width is greater than 0 is dependent on stream gradient. Gradient is the second limiting factor, so a 0 here automatically means the habitat is either moderate or poor. A score of 0 for both gradient and width means that it is highly likely the habitat is not accessible to fish, so the stream segment habitat is identified as poor. Of the three natural suitability variables, hydrologic scour potential has the least influence on the binary score weight, since this is dependent on the quality of gradient and width. The highest quality score is 2, which represents the lowest risk for fish. Where hydrologic scour has a positive rank (2), the habitat rating is at least moderate or good unless there is a 0 gradient or width. Since there are only two possible hydrologic scour potential scores for chum, there were a number of score combinations possible for other species that were not possible for chum. The moderate physical function codes and ratings differ in order (see Table J-1), as modified scour and fine sediment are strongly correlated and have a similar impact on physical habitat quality.

Table J-1. Example of FishEye habitat ratings. Summarized habitat descriptors and limiting factors for three stream segments.

Segment ID	Natural Suitability	Riparian Condition	Modified Physical Function	Habitat Quality Limited by?
331	M	G	P	Sediment
332	G	P	G	Riparian
333	M	G	P	Sediment and BP

FishEye Observed Suitability

The final FishEye Observed Suitability rankings were assigned using a matrix of anthropogenic variable(s) and natural suitability potential (Table J-1). The final FishEye ranks provide a generalized qualitative measure of the natural intrinsic suitability in relation to the anthropogenic features (riparian condition and sediment). Spatial distributions of ranks are useful for indicating stream regions with extremes of either type of limiting factor. The categorical nature of the thresholds and ranking scheme used in FishEye means that extremes (good or poor) have a higher source certainty of habitat quality than moderate or fair descriptors.

The ranking process for FishEye variables incorporates this data effect, as streams with a mid-level rank are considered to have a more neutral habitat influence. If the natural suitability is poor, the best the overall rank can be is moderate, and can only be moderate if the other two variables (modified physical process and riparian) are good or excellent. If natural suitability is fair, the overall FishEye ranking can be good, but only if both riparian and physical process conditions are at least good. Where natural suitability is good, the overall rank is good if just one of the other two variables is also good.

Table J-1. Observed suitability rankings as a function of natural suitability, riparian condition, and physical process condition. Code indicates the final quality rank assigned to the stream segment. P = Poor Quality, F= Fair Quality, A = Adequate/Moderate Quality, and G = Good Quality.

Natural Suitability		**Modified Physical Function**				
		Poor	**Fair**	**Moderate**	**Good**	**Excellent**
Riparian Condition	**Natural Suitability = Poor**					
	Poor	**P**	**P**	**P**	F	F
	Moderate	**P**	**P**	F	F	F
	Good	**P**	F	F	A	A
	Natural Suitability = Fair					
	Poor	**P**	F	F	F	F
	Moderate	**P**	F	F	A	A
	Good	F	F	A	G	G
	Natural Suitability = Good					
	Poor	F	F	F	F	G
	Moderate	F	A	A	G	G
	Good	F	A	G	G	G

Results

Intermediate results from each FishEye component and final FishEye suitability ratings for multiple species are included in Figure J-1 through Figure J-8.

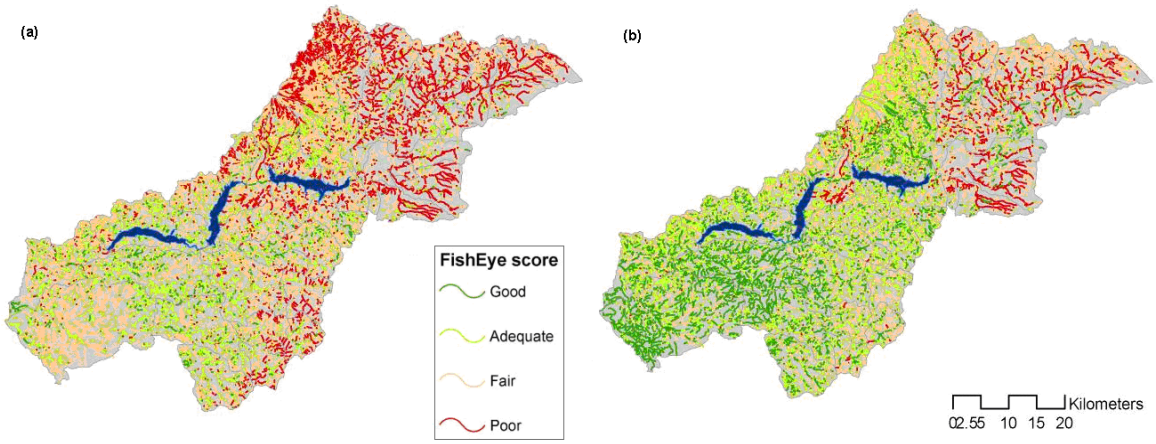

Figure J-1. FishEye Observed suitability rankings for current conditions (a) and pre-development conditions (b) for spring chinook salmon.

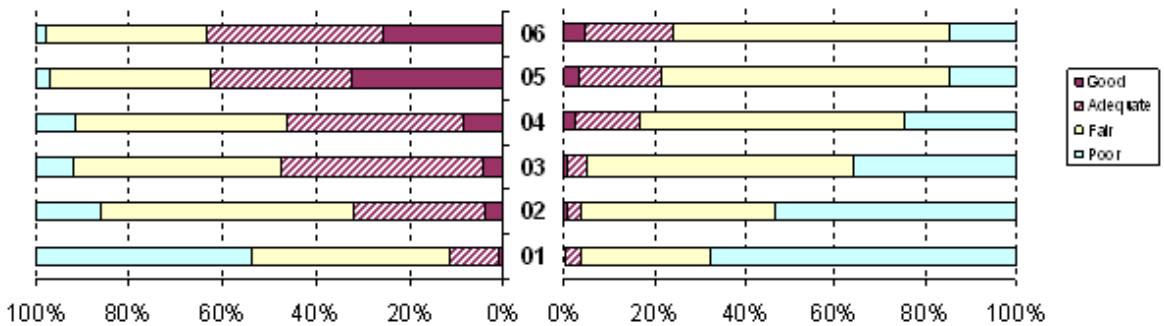

Figure J-2. FishEye Observed suitability rankings by percent for spring chinook for historical (left) and current (right) conditions. Results are grouped by the abbreviated 5th field hydrologic unit code on the *y*-axis.

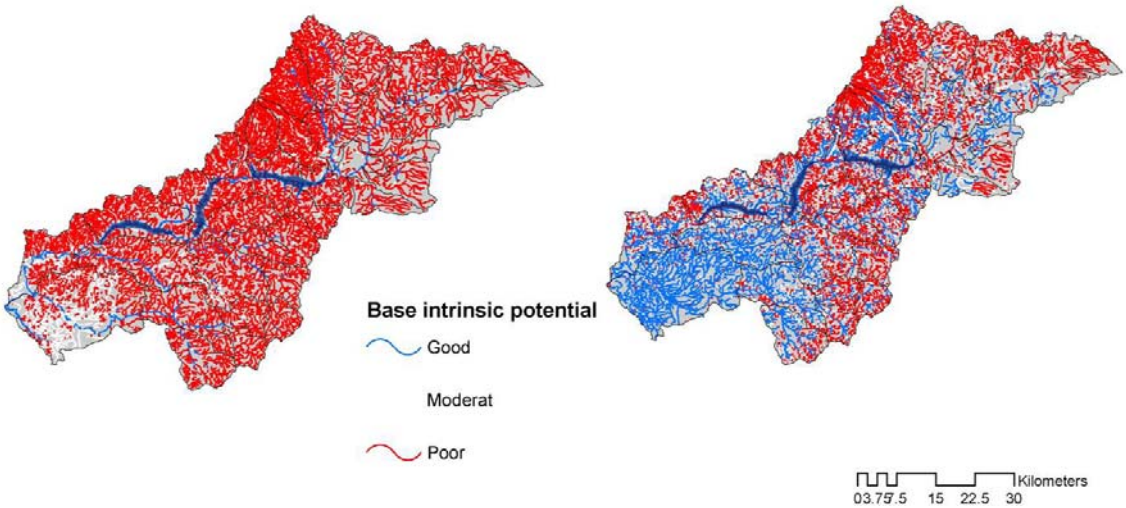

Figure J-3. Natural suitability, or base intrinsic potential for (a) chum salmon and (b) winter steelhead for all streams in the watershed. This includes areas upstream of natural and manmade barriers. A "moderate" potential is represented by white streams, though not clearly distinguished in the legend.

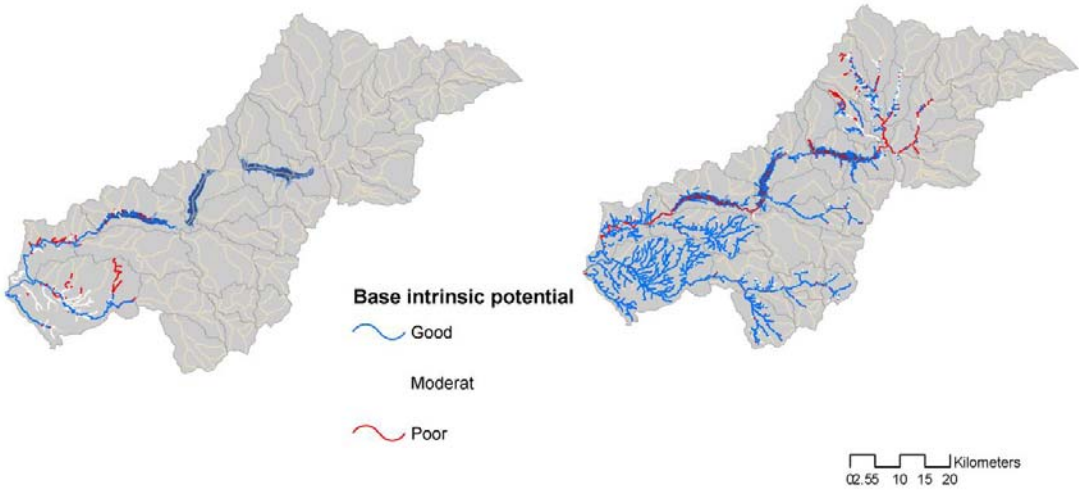

Figure J-4. Natural suitability, or base intrinsic potential for (a) chum salmon and (b) winter steelhead including only historical (pre-dam) distribution areas. The base intrinsic potential includes physical parameters only, so indicates the natural topographic limits to fish distribution and potential habitat. See Appendix A for a description of how fish distribution was determined. A "moderate" potential is represented by white streams, though not clearly distinguished in the legend.

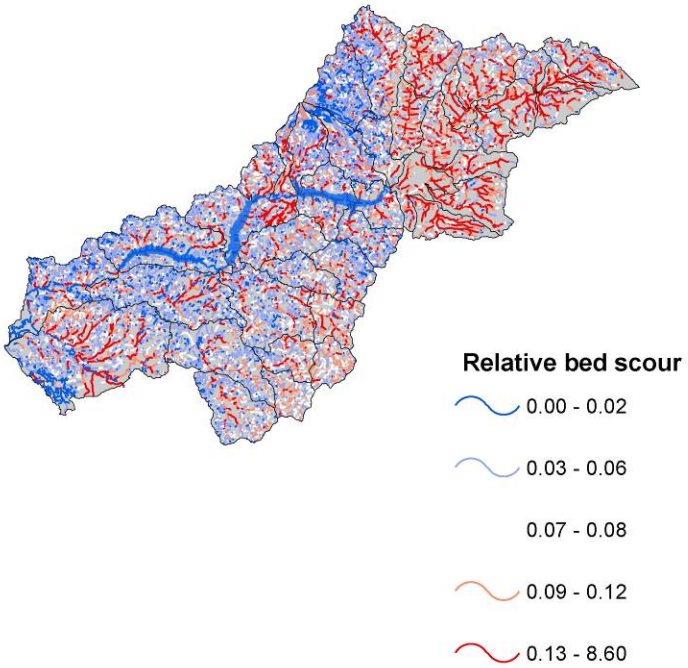

Figure J-5. Modified scour conditions for current conditions. Values > 0.08 are considered to be very high. See Table J-4 for details.

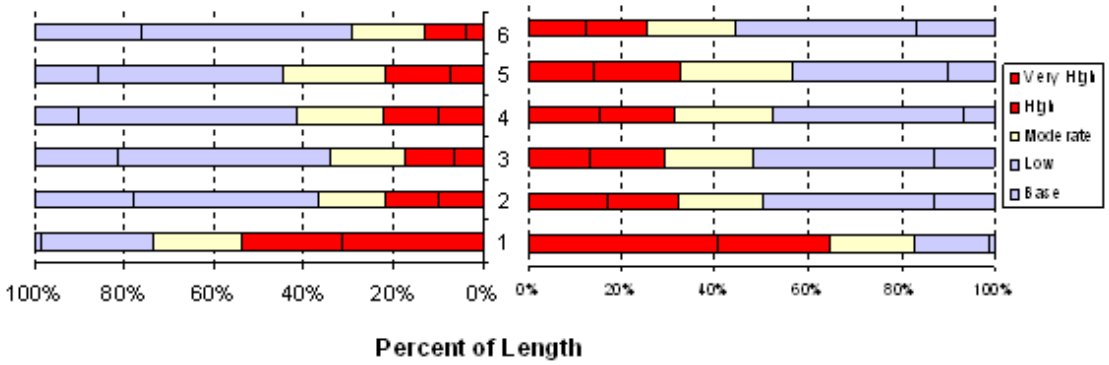

Figure J-6. Difference in high modified bed scour condition rankings for historical (left) and current (right) landscape conditions. Abbreviated hydrologic unit numbers are on the *y*-axis. More intensive bed scour is indicated by either "high" or "very high" categories (in purple). This is the second parameter in the modified physical function score in FishEye.

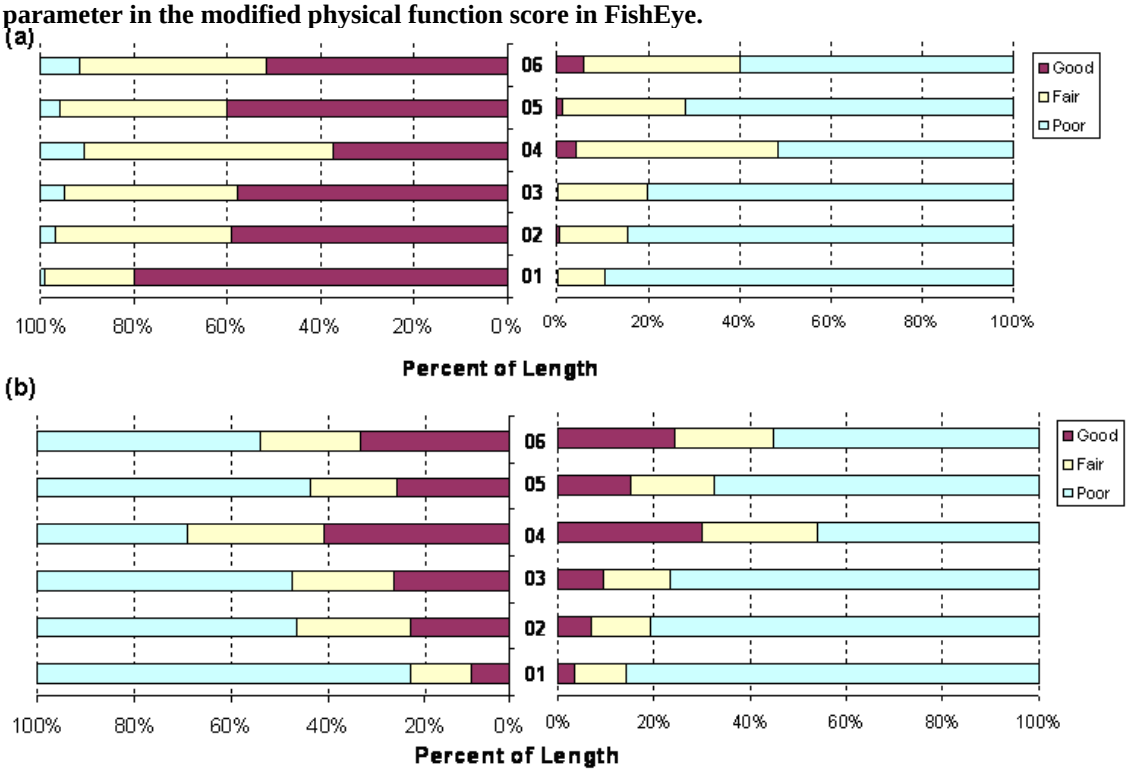

Figure J-7. Percent fines results for (a) juvenile steelhead and (b) juvenile chinook salmon for historical (left) and current (right) templates. Abbreviated hydrologic unit numbers are on the *y*-axis. See Table J-3 for more information. Percent fines were incorporated into the modified physical function component of FishEye.

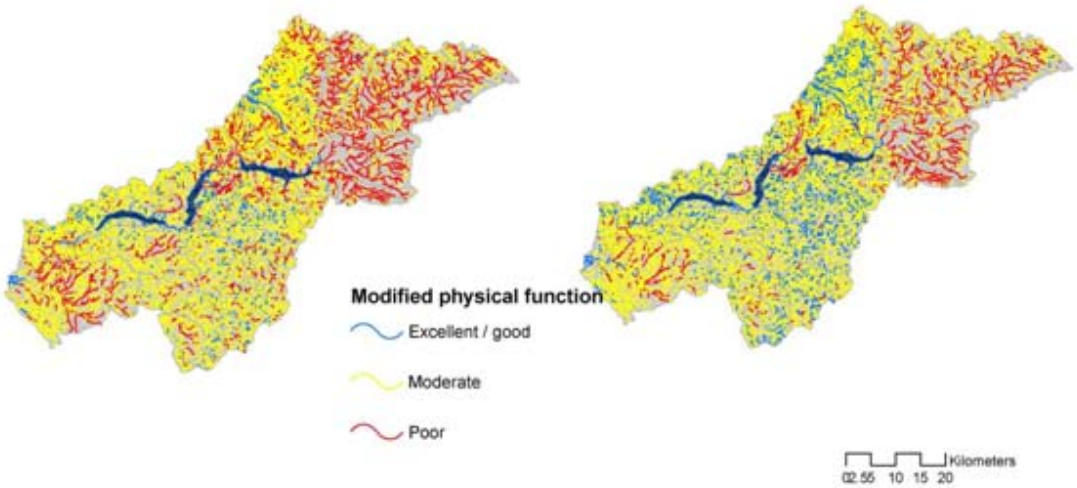

Figure J-8. Modified physical function ratings for FishEye for (a) current conditions and (b) historical conditions. Modified physical function includes modified bed scour and fine sediment.

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Appendix K: Functional Relationships

May 2007

Introduction

A literature review was used to develop empirically based functional relationships between key habitat characteristics and fish abundance or survival at various life stages. Relationships were used in the Lewis River Case Study (Lewis River Case Study Final Report) to understand the impact of fine sediments on juvenile salmonids, both in the FishEye Model (Appendix J) and as their own habitat evaluation model. These functional relationships can be used to estimate the effects of changes in habitat conditions on juvenile survival for particular species. This method of estimating restoration effects on fish populations is the least complex method we could develop. It has clear limitations in that effects on multiple habitat conditions are not linked and there is no life-cycle model incorporated. The strength of this approach is that it is based solely on empirical data. Comparing the restoration priorities between this limited but simple and transparent method to restoration prioritization results from a complex model such as EDT will help identify strengths and limitations of the two approaches as well as to indicate knowledge gaps. Restoration actions that change habitat in a way that is estimated to have a strong fish response using both methods might be rated particularly high.

Methods

Five measures of substrate composition were used in the studies of the relationship between sediment and egg survival included in this analysis: percent fines less than 0.85 mm, percent fines less than 3.3 mm, percent fines less than 4.7 mm, percent fines less than 6.4 mm and the geometric mean of particle size. The actual sieve size used in the various studies varied somewhat but within a size class all were within 0.2 mm of each other. The sediment model that will be used to estimate the changes in sediment resulting from the implementation of the various scenarios used the sediment size classes from Table A-3. The closest correspondence to the sediment sizes in the literature was percent fines < 0.85mm, hence only the relationships developed for that size class will be presented here.

Statistical Analysis

Logistic regression models that included sediment as a covariate were fit to the data from all appropriate studies located in the literature search. In this model, each egg has a specific probability of hatching, with this probability depending on sediment composition. Further, each trial in an experiment or each redd is a binomial experiment with N trials, where N is the number of eggs in the redd. Thus, we model the probability of hatching, and from this derive the proportion of eggs expected to hatch. In most of the studies used here, the number of eggs planted, or the estimated eggs produced by each female were given. In those studies without this information, the average fecundity from other studies with that species was used. In a few of the studies used here, the data given were means of 2-6 trials. These data were weighted to reflect the fact that there is less variability associated with means than with individual values.

An initial fitting of logistic regression models to the individual studies used here indicated that nearly all of the data are over-dispersed; the observed variation exceeds that expected under a binomial model, and therefore quasi-likelihood procedures were

used to fit models to these data (McCullagh & Nelder 1989). Williams' method (Williams 1982) was used to estimate the overdispersion parameter, which is subsequently used to adjust the standard errors of the regression parameter estimates.

The first stage of modeling checked whether the slope of the relationship between percent fines and survival differed. A common slope model was then fit to the data and used to estimate the change in the odds of survival when the percentage of fines in the substrate changes. We needed an estimate of the intercept in order to estimate survival or the change in survival. If there was more than one intercept for a given species, a weighted average of the intercepts was calculated with the slope held constant at the value determined in the previous step. The slope and intercept were used to estimate the change in mean survival resulting from the various scenarios evaluated under this study.

We used a bootstrap procedure (Efron and Gong 1983) to estimate the covariance matrix for the slope and intercept. A bootstrap sample was drawn from each of the groups having a common slope. We fit a parallel lines model to the bootstrap sample and used the slope of the lines as an offset in a subsequent model to estimate the intercept. We repeated this 5000 times, and then used results to estimate the variances of the intercept and slope, and the covariance between them. These estimates were then used to construct 95% Wald confidence intervals for the estimated mean survival.

Results

Egg-to-fry Survival vs. Percent fines < 0.85 mm

The studies utilizing percent fines less than 0.85 mm as the sediment metric included experiments and field studies with four salmon species: chinook, coho and chum salmon and steelhead. Only one paper reported results of an experiment with chum salmon (Hall 1986). There was convincing evidence for a negative relationship between sediment and survival ($p < 0.001$), even though there is considerable overdispersion (deviance=17.22 on 4 df, $p < 0.002$). The modeled relationship is:

$$\text{logit(survival)} = \underset{(0.316)}{0.144} - \underset{(0.029)}{0.146} * \text{fines}$$

Among the papers reporting results from chinook (Bennett et al 2003, Tappel & Bjornn 1983, Hall 1986, Reiser & White 1988, Reiser & White 1990), survival was monitored from the green egg stage in three and from the eyed egg stage in two (Table K-1). Reiser and White (1990) conducted experiments on both, but because the relationship between survival and sediment was not significant for the eyed egg data ($p>0.11$), these data were not included in further analyses. Within each of these groups, the slope of the regression line was not different in the various studies ($p>0.56$ for eyed egg survival; $p > 0.75$ for green egg survival). The intercept did differ among the studies of eyed egg survival ($p<0.006$). Within the studies of green egg survival, two (Bennet et al. 2003 and Reiser and White 1988) were not statistically different ($p< 0.13$), but the third did have a different intercept ($p< 0.0001$).

Table K-1. Chinook salmon and steelhead studies evaluated, including the life stage at which monitoring began, sediment metric and whether the study was conducted in an artificial (lab) or natural (field) environment.

Species	Life Stage	Sediment Metric	Source	Type
chinook	green egg-to-fry	% fines < 0.84mm	Reiser & White 1988	lab
chinook	green egg-to-fry	% fines < 0.84mm	Reiser & White 1990	lab
chinook	green egg-to-fry	% fines < 0.85mm	Bennett et al. 2003	lab
chinook	eyed egg-to-fry	% fines < 0.84mm	Reiser & White 1990	lab
chinook	eyed egg-to-fry	% fines <0.8mm	Hall 1986	lab
chinook	eyed egg-to-fry	% fines < 0.85mm	Tappel & Bjornn 1983	lab
steelhead	eyed egg to yolk absorption	% fines < 0.841mm	Cederholm & Lestelle 1974	field
steelhead	eyed egg-to-fry	% fines < .85mm	Tappel & Bjornn 1983	lab

For the steelhead data (Table K-1), there was no linear relationship in the Cederholm and Lestelle (1974) data ($p > 0.62$). The Tappel and Bjornn (1983) steelhead data monitored survival from the eyed egg stage, and there was no evidence that steelhead survival differed from chinook survival in those data ($p > 0.17$). Thus, the steelhead data were combined with the eyed chinook data. Within these studies, there was no evidence that the slope of the relationship was different among the studies ($p > 0.78$), but the intercept was different in the Hall data ($p < 0.007$).

These results suggest a model for chinook and steelhead survival that adjusts survival depending on the egg stage at which monitoring begins. Since there were no studies found that monitored steelhead survival from the green egg stage, it could not be determined whether steelhead and chinook survival would be the same from the green egg stage. The survival data for each life stage were combined and a single model was fit to estimate the slope of the relationship, although survival from the eyed egg stage was not included in the final functional relationships or the Lewis River DSS.

In these models, the estimated odds of emergence from the green egg stage decreases by 12.1% for each 1% increase in fine sediment (95% Wald confidence interval 8.9% to 15.1%) and the estimated odds for eyed eggs decreases by 25.5% for each 1% increase in fine sediment (95% Wald confidence interval 21.5% to 29.3%).

The final functional relationship models for chinook salmon and steelhead are:

green egg survival (Chinook): $\text{logit(survival)} = \underset{(0.154)}{0.237} - \underset{(0.018)}{0.129} * \text{fines}$

eyed egg survival (Chinook & Steelhead): $\text{logit(survival)} = \underset{(0.266)}{3.54} - \underset{(0.027)}{0.294} * \text{fines}$

Plots of these relationships and 95% confidence intervals for estimated mean survival are included below (Figure K-1 and Figure K-2).

Four coho studies were analyzed (Hall and Lantz 1969, Cederholm and Salo 1979, Tagart 1984, and Hall 1986; see Table K-2). Each of these studies except Hall's monitored survival from green eggs, the latter monitored from the eyed egg stage. There was no evidence that the slope of the relationship between survival and sediment differed in the

two groups ($p > 0.28$), so egg stage was not considered a factor in subsequent analyses. There was moderate evidence suggesting that the slope of the relationship differed among the four studies ($p < 0.05$), however pairwise comparisons did not find any differences so a common slope was assumed for the four data sets.

There is at best moderate evidence that the intercept differed between the data of Cederholm & Salo, Hall, and Tagart ($p > 0.05$), but convincing evidence that the intercept differs in the Hall and Lantz data ($p < 0.0001$). In this model, the odds of coho survival is estimated to decrease 13.3% for every 1% increase in the percentage of fine sediment (95% Wald confidence interval: 9.6% to 16.9%).

Table K-2. Coho salmon evaluated, including the life stage at which monitoring began, sediment metric and whether the study was conducted in an artificial (lab) or natural (field) environment.

Species	Egg Stage	Sediment Metric	Source	Type
coho	egg-to-fry	% fines < 0.83mm	Hall & Lantz 1969	field
coho	eyed egg-to-fry	% fines <0.8mm	Hall 1986	lab
coho	green egg-to-fry	% fines < 0.85mm	Cederholm & Salo 1979	lab
coho	egg-to-fry	% fines < 0.85mm	Tagart 1984	field

The final functional relationship model for coho salmon is:

logit(survival) = 1.72 – 0.143*fines

(0.329) (0.021)

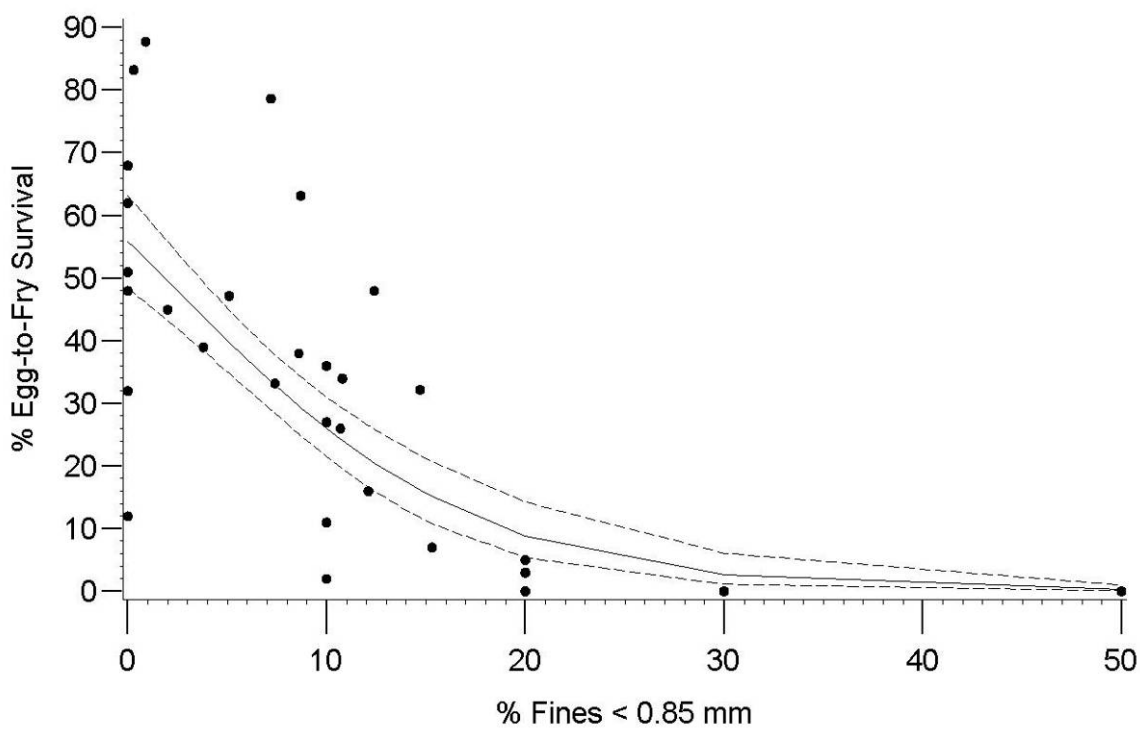

Figure K-1. Relationship between green egg-to-fry survival of chinook salmon and the percentage of fine sediment. The points are from studies used to estimate the relationship. The solid line is the estimated mean survival; the dashed lines are 95% confidence intervals for the mean.

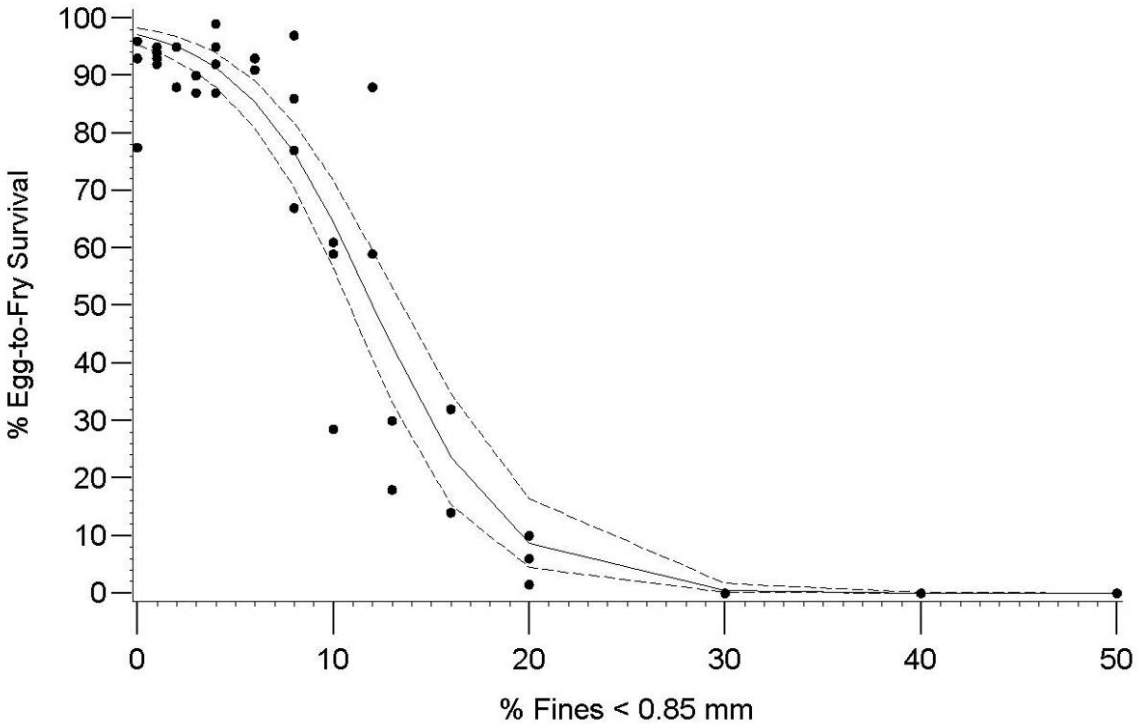

Figure K-2. Relationship between eyed egg-to-fry survival of chinook salmon and steelhead and the percentage of fine sediment. The points are from studies used to estimate the relationship. The solid line is the estimated mean survival; the dashed lines are 95% confidence intervals for the mean.

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Appendix L: Models used to coordinate with Ecosystem Diagnosis and Treatment Model (EDT)

May 2007

Introduction

We used the EDT model (Mobrand Biometrics 2004) in two different ways in this analysis. The first use of the EDT model was to generate a restoration strategy based on the reach level restoration and preservation prioritization output from EDT (Table L-1 and Table L-2). The second use of the EDT model was to evaluate the future landscapes. Using EDT to evaluate future landscapes required a translation between restoration actions and EDT input data (Table L-3).

Table L-1. The prioritization system for allocating funds to EDT reaches based on EDT output. 50% of available funds were designated for restoration and 50% for protection. The same reach-level prioritization system was used to allocate funds independently for restoration and for protection.

Basis for Prioritization	Prioritization Notes
EDT model rankings for restoration or protection benefit	Select the reaches with the highest EDT restoration or protection benefit ranking If funds remain after treating all reaches identified as high priority, move to the reaches identified as intermediate priority.
Reach type: Spawning vs. non-spawning reaches. Mainstem vs. tributary reaches	Start with the spawning reaches. If funds remain after all high priority spawning reaches are treated, move to high priority mainstem reaches
Reach location	Within the high priority spawning (or migration) reaches, select the most upstream reach first.

Table L-2. Translation from EDT model output for current conditions within each reach prioritized for restoration or preservation to the EDT watershed management strategy. 50% of the funds were spent on restoration actions. Habitat attributes identified by EDT, by reach, as the most important were "fixed" first. Numbers in each cell represent the prioritization of restoration actions within each row. If two habitat attributes were most limiting, we started with the least expensive problem to fix. All protection funds were spent on riparian protection or restoration. If the current riparian condition was good (as rated by the remotely-sensed riparian model in Table 4 of the main report), riparian conditions were protected; if the current condition was fair or poor, riparian conditions were restored.

EDT Habitat Attribute	Restoration Actions				
	Restore Riparian	Decommission Roads	Remove Barriers	Restore for Spawning	Restore Floodplain[a]
Key Habitat				1	2
Temperature	1				
Sediment Load	2	1			
Obstructions			1		
Habitat Diversity					1
Food	1				
Flow	1	2			
Chemicals	1[b]				
Channel Stability	1	2			

[a] Only areas that historically had floodplains could be treated with floodplain restoration.
[b] If the habitat element was chemicals, riparian areas were only treated if the uplands were currently classified as agricultural or urban land-use.

Table L-3. Model used to translate conservation actions in management strategies into data in a format ready to be used as inputs by the EDT model. All actions were subject to 4 constraints: 1) the proportion of each EDT reach affected by a strategy was equal to the proportion of affected SSHIAP reaches comprising an EDT reach; 2) new EDT scores affected by conservation actions were constrained between patient and template scores and trended toward the template; 3) actions only affected scores if there was a potential for change (i.e., patient - template ≠ 0); and 4) if >1 actions each changed EDT scores, only the largest was registered if effects were in the same direction but the sum of effects was registered if effects of actions had different directions. Abbreviations used are as follows: pΔ = potential for change; p(reach) = proportion of the EDT reach affected; ↑ = improve score. Conditions: †1 if any part of riparian area was originally urban and at least 50% of the reach is protected/restored; †2 also improve the next downstream reach in the same way; †3 if LWD or PFC function improves.

EDT Attribute	Decommission Roads	Protect or Restore Riparian	Restore Floodplain Connectivity	Restore Spawning Habitat
Bed Scour	Scour Depth is estimated directly from the modeled 2.3 year flood flow as Depth = 10*sqrt(flood discharge/bankfull width) (from Emmett and Leopold 1965), then converted to EDT ratings.			
Embeddedness	↑ score by p(reach) where roads are restored * Δ in % covered (as estimated based on road density).	↑ score by p(reach) restored.		New score is the p(reach) restored/protected * pΔ.
Diel Variation in Flow		↑ score by ½ p(reach) where riparian area was urban * pΔ.†1		
Fine Sediment Deposited	↑ score by p(reach) where roads are restored * Δ in % fines (as estimated based on road density)* 1.34.			
High Flow	High Flow was calculated as the %Δ in modeled 2.3 year flood flow from historical, and then converted to EDT ratings.			
Large Woody Debris Recruited		New score is the p(reach) restored/protected * pΔ.†3		New score is the p(reach) restored/protected * pΔ.†3
Miscellaneous Toxic Wastes		↑ score by p(reach) where riparian area was urban * pΔ.†1		
Monthly Max Temperature		New score is the p(reach) restored/protected * pΔ.†2		

Nutrient Enrichment		↑ score by p(reach) where riparian area was agriculture * pΔ.[†1]		
Channel Confinement resulting from hydrological modifications			New score is the p(reach) restored/protected * pΔ.	
Off-Channel Habitat			↑ score by p(reach) where floodplains were restored * pΔ.	
Riparian Functions		New score is the p(reach) restored/protected * pΔ.	New score is the p(reach) restored/protected * pΔ.	
Small Cobble-Dominated Habitat				New score is the p(reach) where spawning habitat is restored * pΔ.
Turbidity	↑ score by p(reach) where roads are restored * 0.3 * Δ in road density.	↑ score by p(reach) restored * 0.3.		

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Appendix M: Historical Floodplain and Channel Reconstruction

May 2007

Introduction

Channel change analyses were conducted on the historical floodplains of sections of both the North and East Fork Lewis. The historical analysis was divided into two projects – historical river morphology for the three reservoirs of the Lewis (Merwin, Yale, and Swift reservoirs), and a channel change analysis for a 23-km section of the lower East Fork River (Figure M-1).

The river morphology project was based on a 1938 U.S. Geologic Survey planar survey of the East Fork (to Lucia Falls) and the North Fork Lewis River scanned map series. This analysis included three sections of the upper North Fork Lewis River currently flooded by Merwin (15 km), Yale (18 km), and Swift (13 km) reservoirs. The purpose of this assessment was to provide a general understanding of historical stream conditions and historically available habitat. The East Fork channel change analysis was done for a twenty-three (river) kilometer section of the East Fork Lewis River, based on multiple years of aerial photo interpretation data. The purpose of this change analysis was to assess natural and modified stream channel characteristics in a floodplain environment. We were unable to combine the various sources for use in both projects because of inconsistencies in alignment and geographic transformation.

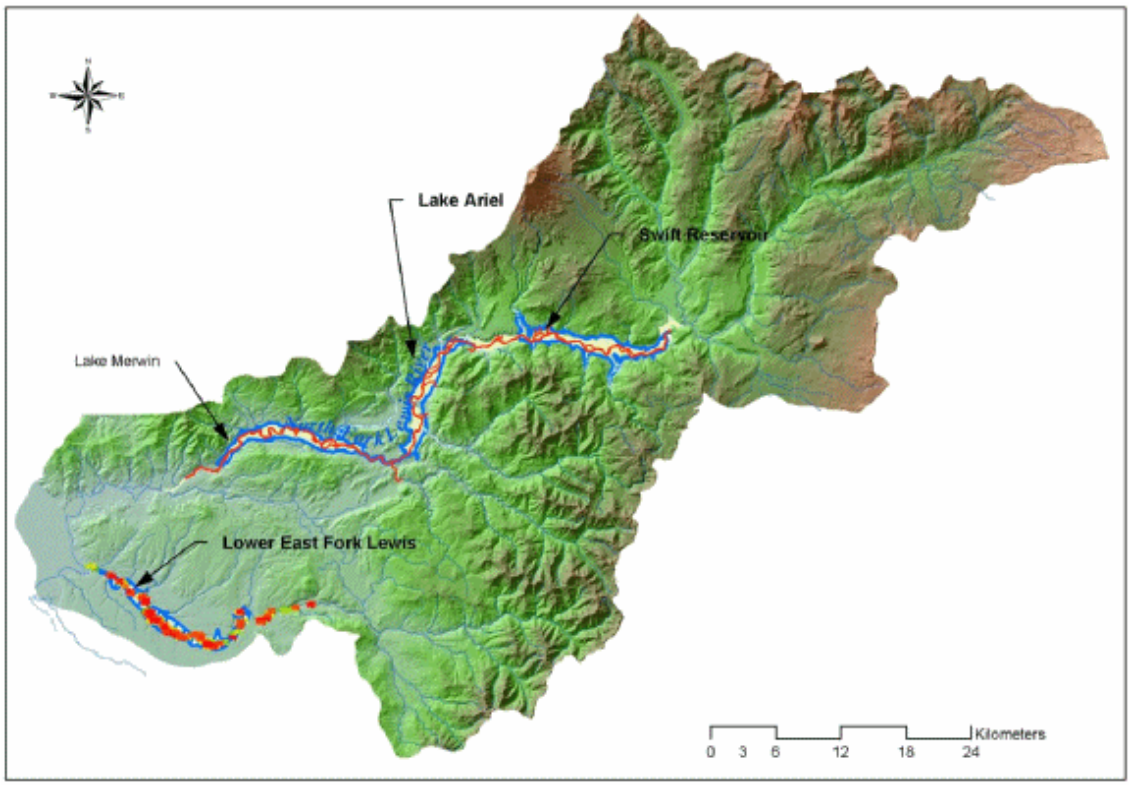

Figure M-1: The four historical data sites in the Lewis watershed, three reservoirs and one floodplain river section. Red indicates historical channel data from Mylars, aerial photographs, or maps.

North Fork

The primary base map reference for the historical channel analysis were 1938 survey maps from the U.S. Geologic Survey (USGS). These maps were scanned and digitized, and the main channels, side channels, islands, tributaries, and contours were converted to GIS data layers using a scanning trace function (by private consultant) (Figure M-1). The maps depict the channels as double-banked channels, and the edge was determined from the location of the wetted shoreline.

Contours crossed the channel at every five-foot gain in elevation, though some steeper areas had contours every 1-2 feet. Each contour crossing was used to divide the river into a longitudinal series of channel sections. Main channels and side channels were broken into sections using this method, and the starting and ending elevations were identified for each section. Gradient was calculated by dividing change in elevation by the thalweg length of that channel section. For each section, perpendicular transects were created at three locations, and intersected with the channel section polygon to identify location and obtain bank to bank wetted width measurements from the U.S. Geological Survey maps. The three measurements were averaged to obtain wetted width for the section. Each section was identified as a main channel or secondary side channel. Some sections were further split to delineate the correct channel type.

Historical and aerial photos (1928, 1946-48) and narrative descriptions of historical conditions (Rice 1996; Cadastral survey notes, BLM 2000) were used to determine the size and composition of the vegetation at various locations along the river that is currently inundated by the three reservoirs. Based on these sources and descriptions, six categories of bank riparian vegetation were determined and identified for each stream section (Table M-1). Categories were extrapolated to similar sites and sections for the remainder of the reservoir. Data were mapped and summarized to represent historical channel and vegetated conditions for the river (Figure M-1).

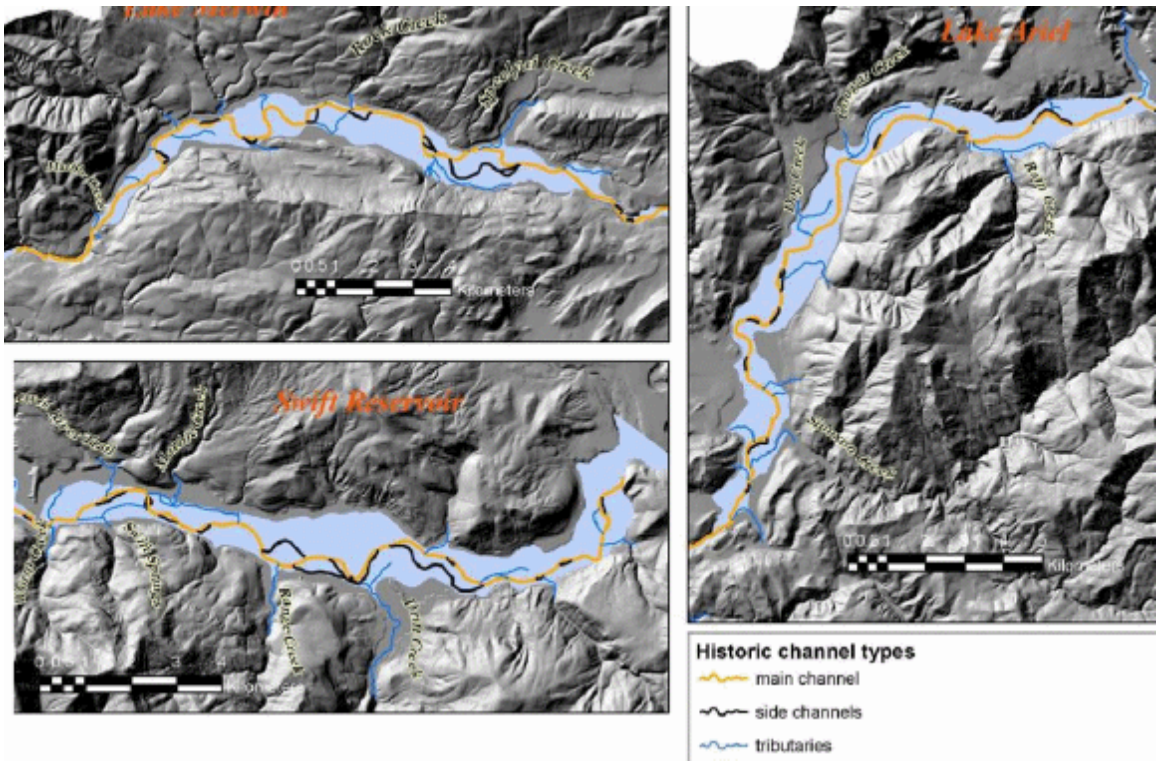

Figure M-1. Historical channels in the North Fork Lewis River. The blue background indicates the extent of the historical floodplain, which coincides in most places with the lateral extent of the current reservoir. Main channels are in orange, side channels (both single and double side channels) are indicated in black.

Table M-1. Translation table by which historical riparian forest survey notes were used to estimate historical vegetation. The riparian identifier was used to classify each stream section with a particular pre-inundation riparian vegetation type.

Riparian ID	Location Description	Photo year	Seral Stage	(%)			Dominant Conifer (%)	Shade	Large Woody Debris	Pool Forming Conifers	Comments
				Conifer	Deciduous	Bare nonforest					
A & B	Upstream end of Merwin	1948	Late	75%	25%		>=20" (50% con)	Good	Good	Good	
	DS Canyon Creek	1948	Late	75%	25%		>=20" (50% con)	Good	Good	Good	
C		1948	Early	30%	60%	10%, herbs	>= 20"	Poor	Good	Fair	
D	1/4 mile DS Siouxon & near Siouxon, lower Yale,	1948	Other (non-Con)	0%	40%	60%, burn, grass		Poor	Fair	Poor	
E	US Siouxon, near confluence, lower 1/2 of Yale	1948	Mid	70%	30%		>= 20"	Good	Good	Good	Straight reaches, inside bends of meanders
E2	US Siouxon, near confluence, lower 1/2 of Yale	1948	Other (non-Con)		90%	10%		Fair	Fair	Poor	Floodplain surfaces (within migration zone)
F	US Siouxon - upper end Yale Lake	1948	Late	70%	30%		>=20" (50% con)	Good	Good	Good	Left, upslope of floodplain
A2	US town of Cougar	1954, 1957	Mid	70%	30%		>=20" (50% con)	Good	Good	Good	Both sides, upslope of floodplain
A3	US town of Cougar	1954, 1957	Other (non-Con)	50%	50%		10-20" (70% con)	Good	Good	Fair	Floodplain surfaces (within migraion zone)
B	Tribs of Swift Reservoir	1954, 1957	Late / Mixed	70%	30%		>=20" (50% con)	Good	Good	Good	Tributaries only - upslope of floodplain

East Fork

Aerial photographs from multiple time periods (1854, 1928, 1946, 1948, 1984, 1976, 1955, 1990) were available for sections of the Lewis. The base maps used were USGS 1:24,000 topographic quadrangle maps (1990). Tributaries, main channels, secondary channels, point bars and other features were identified on non-georeferenced aerial photos, and transferred onto Mylar sheets overlays referenced to USGS quad sheets, using primarily manual methods. Stereo pairs were used where possible. Only channel sections that differed from the most recent time period (1990) river channel maps were transferred to the color-coded Mylar sheets. Gaps in delineated channels were filled with the 1990 stream channel to delineate a complete length of channel for each time period. Stream channels were broken into sections by channel type (main, lateral side, slough).

Information for all years was converted to GIS format by scanning and digitizing the Mylars and attributing all lines with source year and other attributes. Four years of Mylar data were chosen (1854, 1928, 1955, and 1990) to map channel migration patterns in the lower East Fork Lewis River. These years were chosen based on completeness of aerial photo coverage. Analysis of section-by-section changes in main channel position and associated channel features (e.g., secondary channels, wetlands) was done by segmenting the floodplain into one-kilometer lateral slices, following methods of Oetter et al. (2004). Channel types and lengths within each lateral floodplain slice were summarized by year to obtain channel change results (Table M-1, Figure M-1, Figure M-2).

Table M-1. Summary lengths (m) of historical channel characteristics for the lower East Fork Lewis River.

Section ID	MAIN CHANNEL				SIDE CHANNELS				BLIND CHANNELS			
	1990	1955	1928	1854	1990	1955	1928	1854	1990	1955	1928	1854
1	1183	1183	1183	1037	0	0	90	1254	2187	0	1159	0
2	1126	1126	1126	950	0	0	0	978	158	0	304	0
3	1353	1353	1400	1268	0	0	0	0	640	0	0	0
4	1060	1060	1215	1060	0	0	0	0	140	0	0	0
5	1105	1105	1488	1136	0	235	0	0	31	0	0	0
6	1540	1626	1491	1592	0	603	0	0	590	0	0	43
7	1546	1727	1273	1486	251	589	926	2337	0	0	0	0
8	1610	1214	1417	1180	1107	1032	41	4042	0	0	211	0
9	1043	1128	1335	953	204	820	0	303	0	0	0	0
10	1334	1428	1372	1363	202	427	0	0	0	0	102	0
11	1290	1279	1319	1358	649	1275	600	1509	1281	0	0	0
12	1381	1450	1483	1056	850	873	1036	534	135	0	45	0
13	1023	1005	1011	864	344	943	797	0	0	0	0	0
14	1007	1007	1007	1007	0	147	0	0	0	0	0	0
15	1317	1338	1366	1191	524	743	123	0	54	0	0	0
16	1359	1359	1344	1332	0	226	0	0	0	0	0	0
17	1637	1637	1582	1637	64	131	0	64	0	0	0	0
18	989	989	989	989	347	523	0	347	0	0	0	0

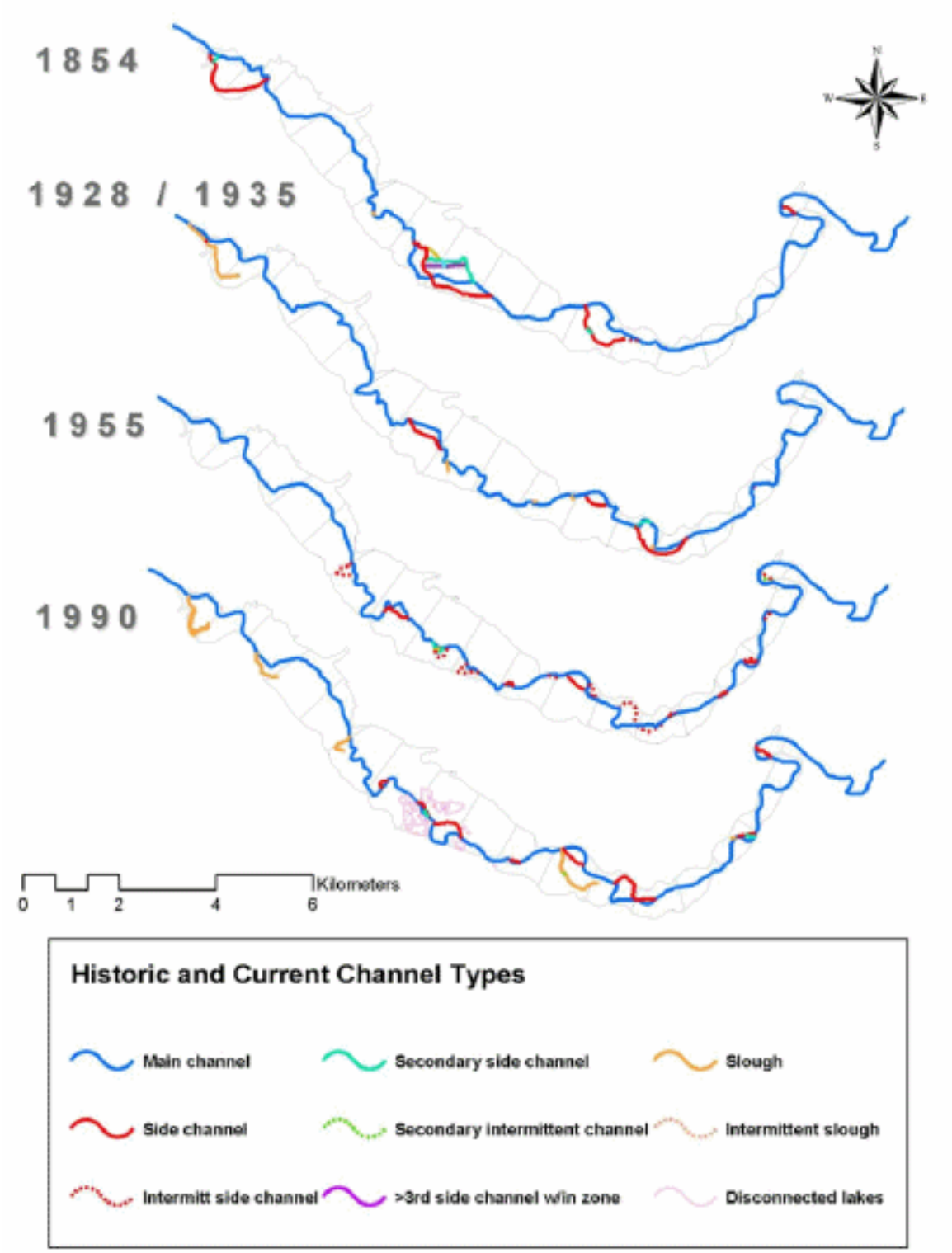

Figure M-1. Historical channels for the lower section of the East Fork Lewis River. Four time periods are presented, and colors indicate channel characteristics for each. Lateral cross-sections on the maps indicate section area delineations used to measure differences in channel length between the years. The right side of the graphic(s) is upstream.

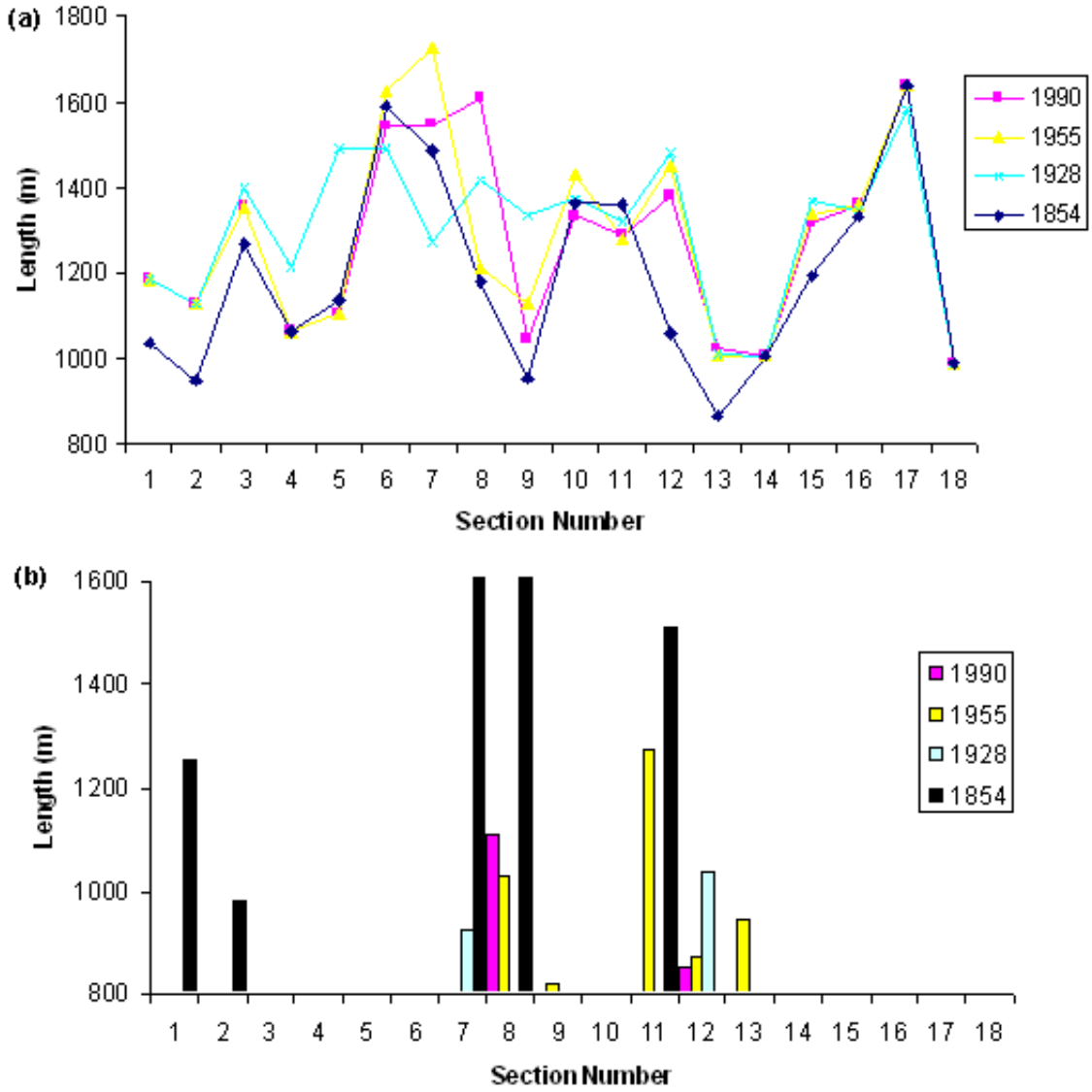

Figure M-2: Channel length for historical channels in the lower section of the East Fork Lewis River. Graph (a) indicates channel length of the main channel by section, (b) presents the total length of side channels by section and year. Sections are numbered from downstream to upstream. Total length was 23 kilometers (based on 1990 main channel length for study area).

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